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Effects of demand-side restrictions on high-deforestation palm oil in Europe on deforestation and emissions in Indonesia

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E-mail: jbusch@conservation.org**Keywords:** climate policy, forest-risk commodities, land-use change, oil palm, supply chains, teleconnections, tradeSupplementary material for this article is available [online](#)**Abstract**

Demand-side restrictions on high-deforestation commodities are expanding as a climate policy, but their impact on reducing tropical deforestation and emissions has yet to be quantified. Here we model the effects of demand-side restrictions on high-deforestation palm oil in Europe on deforestation and emissions in Indonesia. We do so by integrating a model of global trade with a spatially explicit model of land-use change in Indonesia. We estimate a European ban on high-deforestation palm oil from 2000 to 2015 would have led to a 8.9% global price premium on low-deforestation palm oil, resulting in 21 374 ha yr⁻¹ (1.60%) less deforestation and 21.1 million tCO₂ yr⁻¹ (1.91%) less emissions from deforestation in Indonesia relative to what occurred. A hypothetical Indonesia-wide carbon price would have achieved equivalent emission reductions at \$0.81/tCO₂. Impacts of a ban are small because: 52% of Europe's imports of high-deforestation palm oil would have shifted to non-participating countries; the price elasticity of supply of high-deforestation oil palm cropland is small (0.13); and conversion to oil palm was responsible for only 32% of deforestation in Indonesia. If demand-side restrictions succeed in substantially reducing deforestation, it is likely to be through non-price pathways.

1. Introduction

Tropical deforestation and forest degradation emit 1.8–11.4 GtCO₂ yr⁻¹ (Olsson *et al* 2019). The largest direct cause of tropical deforestation is the conversion of forested land to produce globally traded 'forest-risk commodities,' e.g. palm oil, beef, soy, and paper (Curtis *et al* 2018, Pendrill *et al* 2019).

Consumer countries are increasingly attempting to combat tropical deforestation and climate change by distinguishing whether forest-risk commodities were produced with high or low levels of recent deforestation, and restricting demand for commodities associated with high deforestation. Public demand-side restrictions on high-deforestation commodities have been enacted in the United Kingdom

(DEFRA 2015), Netherlands (PIANOo 2017), Norway (Norwegian Parliament 2018, Rainforest Foundation Norway 2018), and France (République Française 2018), introduced in the United Kingdom (BBC 2020), vetoed in California (California Legislature 2020, Office of the Governor 2021), and proposed in the European Union (European Parliament 2020; Bager *et al* 2021) and United States (United States Congress 2021) (table S11 available online at stacks.iop.org/ERL/17/014035/mmedia).

The recent public policy focus on restricting demand for high-deforestation commodities represents a shift in approach from positive incentives to keep tropical forests standing through international carbon payments via markets or funds (i.e. REDD+; Seymour and Busch 2016). It potentially

complements private corporate commitments to eliminate deforestation from commodity supply chains (Lambin *et al* 2018, NYDF Assessment Partners 2019, Hargita *et al* 2020).

The goals of governments that restrict demand for high-deforestation commodities include both ending domestic consumption of products associated with deforestation as well as contributing to reducing deforestation abroad (e.g. République Française 2018). Meanwhile governments of tropical commodity-producing countries have accused consumer countries of using climate concerns as thinly guised protectionism for their own farmers (e.g. Reuters 2019). However, despite the controversial and potentially central role of demand-side restrictions on high-deforestation commodities as a climate policy, their impact on reducing tropical deforestation and emissions has yet to be quantified.

Here we model the effects of one important bloc of consumer countries (Europe) restricting demand for the high-deforestation type of one important forest-risk commodity (palm oil) on deforestation and emissions in one important tropical country (Indonesia). We do so by integrating a global trade model (GTAP-BIO; Taheripour *et al* 2019) with a spatially explicit model of land-use change in Indonesia (OSIRIS; Busch *et al* 2015). We use spatial data on oil palm expansion (Austin *et al* 2017) and forest cover loss in Indonesia from 2000 to 2015 (Hansen *et al* 2013) to distinguish whether oil palm expansion occurred on land deforested after 2000 ('high deforestation') or before 2000 ('low deforestation'). We estimate how much larger impacts would be if other countries outside Europe also enacted demand-side restrictions. And we compare the impacts of demand-side restrictions to the impacts of a benchmark alternative climate policy: a hypothetical domestic carbon price on emissions from deforestation in Indonesia.

Previous studies have examined the distal impacts of demand-side policies on palm oil production (e.g. Britz and Hertel 2011, Elliott 2015, Jafari *et al* 2017, Taheripour *et al* 2019, Hsiao 2021). However, these studies considered the effects of policies on palm oil in aggregate; they did not differentiate between high-deforestation and low-deforestation palm oil. Disaggregating the palm oil market into high-deforestation and low-deforestation segments, as we do here using spatial data on the location of oil palm expansion, lets us better represent the recent wave of public policies, and is the principal innovation of this paper.

Our paper advances several other strands of literature as well. Previous studies have investigated the environmental impacts of bans, e.g. on shark fin (Ferretti *et al* 2020), ivory (Sosnowski *et al* 2019), and rhino horn (Crookes and Blignaut 2015). But due to the illicit nature of these markets, reliable data on production, trade, and consumption is scarce, which

hampers the calibration of sophisticated policy models, as we undertake here using readily available data on the international palm oil market.

Previous studies have researched the effects of eco-labels for products that are interchangeable in consumption but vary in the environmental friendliness of their production, e.g. dolphin-safe tuna (Kirby *et al* 2014), Forest Stewardship Council (FSC) certified timber (Blackman *et al* 2018), and Roundtable on Sustainable Palm Oil (RSPO) certified palm oil (Carlson *et al* 2018). But in the case of eco-labels, price premiums are driven by voluntary consumer preferences and voluntary producer adoption. We are aware of only one previous study of mandatory national restrictions on products that do not comply with eco-certification, for tropical timber (Cole *et al* 2021).

Finally, our study contributes to literature estimating the effectiveness of initiatives to reduce deforestation and emissions in Indonesia, e.g. through protected areas (Brun *et al* 2015, Shah and Baylis 2015), debt-for-nature swaps (Cassimon *et al* 2011), tenure security and REDD+ projects (Resosudarmo *et al* 2014), reduced-impact logging (Griscom *et al* 2019), FSC certification (Miteva *et al* 2015), community forestry (Santika *et al* 2017, 2019), community land titling (Kraus *et al* 2021), multifunctional landscapes (Santika *et al* 2015), and a moratorium on concessions for oil palm, timber, and logging (Busch *et al* 2015, Chen *et al* 2019). It also contributes to a growing effort to integrate global economic models with fine-scale analysis of land-use change (Hertel *et al* 2019).

2. Methods

We modeled the effects of restrictions on demand for high-deforestation palm oil in consumer countries on deforestation and emissions in Indonesia. We integrated the GTAP-BIO global trade model and the OSIRIS model of land-use change in Indonesia in a three-step process (figure SI1). First, we developed the land-use change model to estimate how the conversion of forests to oil palm plantations (which produced 'high-deforestation' palm oil) and the conversion of non-forests to oil palm plantations (which produced 'low-deforestation' palm oil) responded to changes in prices (i.e. 'price elasticities of supply'). Then, using these price elasticities implemented in the GTAP-BIO model, we modeled how prices, quantities, and trade flows of high-deforestation and low-deforestation palm oil would change globally if consumer countries restricted demand for high-deforestation palm oil. Finally, we mapped global price changes back to spatially explicit changes in land use and emissions in Indonesia. We selected palm oil as the commodity of interest, Indonesia as the study region, Europe as the policy focus, and 2000–2015 as the study period for reasons elaborated in the supplementary information (scope of inquiry).

We modeled the conversion of forested and non-forested land to oil palm plantation from 2000 to 2015 as a function of palm oil price and other geographic factors (slope, elevation, distance from roads and provincial capitals, logging, timber, and oil palm concessions, and strict and multiple-use protected areas) using the OSIRIS land-use change model, as elaborated in the supplementary information (determinants of oil palm expansion). OSIRIS has been used to model policy decisions including reference levels for REDD+ (Busch *et al* 2009), incentives to reduce deforestation in Indonesia (Busch *et al* 2012), and carbon prices for pan-tropical reforestation (Busch *et al* 2019).

We defined palm oil as ‘high-deforestation’ if it was produced on land that was converted from forest after the year 2000, and ‘low-deforestation’ if it was produced on land that was converted from forest before the year 2000. We calculated the price elasticity of supply of high-deforestation and low-deforestation oil palm cropland using the observed relationship between palm oil price and conversion, as elaborated in the supplementary information (price elasticity of supply of oil palm).

We modeled global demand and supply of palm oil using GTAP-BIO, as elaborated in the supplementary information (a global market for two types of palm oil). GTAP is a widely used global Computable General Equilibrium model (Hertel 1997). For this study we used the GTAP 9 data base which represents the global economy in 2011. This data base disaggregates global economic activity and bilateral trade into 57 sectors and 140 regions (Aguilar *et al* 2016). We used the GTAP-BIO model (Taheripour *et al* 2017), which further disaggregates oilseeds into four subsectors (oil palm, soybeans, rapeseed, and other oil crops), as well as four oilseed crushing industries, in order to capture the potential for demand-side substitution among these products. GTAP-BIO aggregates the full data base to seven regions (Indonesia and Malaysia; European Union; United States; China; Brazil; Other South America; Rest of World) for computational and analytical tractability. GTAP-BIO models the market for palm oil in three successive, vertically integrated stages: oil palm cropland; palm fruit; and palm oil. Our main extension and improvement of the GTAP-BIO model for this paper was to disaggregate demand for and supply of palm oil into two distinct commodities, high-deforestation palm oil and low-deforestation palm oil, allowing different prices, quantities, and trade flows for each.

In our base scenario, all Indonesian palm oil exports were treated equally by consumers and therefore exhibited a single global price. In our primary policy scenario, consumer countries in Europe imposed a ban on the use of high-deforestation palm oil, as elaborated in the supplementary information (restricting demand for high-deforestation palm oil).

In secondary policy scenarios, we sequentially expanded the breadth of participation across regions. Starting with Europe (the destination for 12.0% of palm oil produced in Indonesia and Malaysia, in GTAP-BIO), we expanded to Europe plus the United States (13.7%), Europe plus the United States plus China (23.1%), and all countries outside Indonesia and Malaysia (56.0%).

After running the base and policy scenarios in GTAP-BIO to generate changes in prices of oil palm cropland, we used the Indonesia land-use change model described above to calculate corresponding changes in quantities of conversion. We calculated greenhouse gas emissions due to conversion of forest to oil palm plantation by multiplying the area of conversion of forest to oil palm in each cell by the cell-specific emission factors used in Busch *et al* (2015), as elaborated in the supplementary information (land-use change and greenhouse gas emissions).

We used cost curves from Busch *et al* (2015) to identify the hypothetical carbon price at which an equivalent reduction in emissions could be achieved, using either a mandatory carbon price (payments for emission reductions plus taxes on emission increases) or a voluntary carbon price (payments for emission reductions only).

To assess uncertainties associated with model parameterization, we generated 95% confidence intervals around trade volumes and producer prices for each scenario, using a systematic sensitivity analysis in GTAP-BIO, as elaborated in the supplementary information (sensitivity analysis). To assess uncertainties associated with emissions from conversion, we tested alternative forest carbon data sets and emission factors for peat and mineral soil.

3. Results

Of the 11.8 million hectares of oil palm plantation in 2015, we find that 6.4 million hectares (54%) was planted after 2000 on land that had been forested in 2000. A 1.0 million hectares (8.7%) was planted on land that was not forested in 2000. 4.4 million hectares (37%) was planted before 2000; the forested status of this land before 2000 could not be determined from our data (figure SI2). Use of an alternative map corresponding to the Indonesia Ministry of Forestry’s primary and secondary forest cover types in Indonesia (Global Forest Watch 2019) suggests that just 2.1 million hectares of oil palm (18%) was planted after 2000 on land that had been forested circa 2000. Our estimate of the share of post-2000 oil palm expansion that occurred on forested land, 86%, is higher than estimates of previous analyses (at least 56% from 1990 to 2005, Koh and Wilcove 2008; 21% from 2000 to 2015, Austin *et al* 2017; 54% from 1989 to 2013, Vijay *et al* 2016; 36.5% from 1990 to 2010, Gunarso *et al* 2013; 36% from 2000 to 2019,

Gaveau *et al* 2021; and 38% in a multi-period meta-study, Meijaard *et al* 2020). This is because our forest cover base map (Hansen *et al* 2013) included more areas of tree cover (e.g. secondary forests; non-forest tree cover) than forest cover base maps used in other studies (e.g. Margono *et al* 2014, Ministry of Environment and Forestry 2015), and because our definition of conversion allowed for a longer time increment between forest cover and plantation establishment than some other studies (up to 15 years in our study rather than up to 10 or 5 years in Gunarso *et al* 2013, Austin *et al* 2017).

More conversion of forest and non-forest to oil palm plantation between 2000 and 2015 occurred, all else equal, on land that was flatter, lower, nearer to roads, nearer to provincial capitals, and had greater estimated potential revenue from palm oil (tables SI2 and SI3). Conversion to oil palm was less likely within strict and multiple-use protected areas. Conversion to oil palm was more likely within lands designated as oil palm concession (*kebun*) or timber concession (HTI) circa 2010. Within lands designated as logging concessions (HPH), forest-to-palm conversion was more likely, but non-forest-to-palm conversion was less likely. Our finding that lower slope, lower elevation, and less distance to large cities were determinants of oil palm expansion is consistent with Shevade and Loboda (2019).

We estimated the price elasticity of forest-to-palm conversion to be 0.13, meaning that every 1% increase in the price of palm oil was associated with a 0.13% increase in forest-to-palm conversion. Meanwhile price elasticity of non-forest-to-palm conversion was 0.17, meaning that every 1% increase in the price of palm oil was associated with a 0.17% increase in non-forest-to-palm conversion (table SI4). These numbers compare with an estimate by Gaveau *et al* (2021) that a 1% decrease in the price of palm oil was associated with a 1.08% decrease in industrial oil palm expansion in Indonesia from 2001 to 2019.

The price elasticity of supply for high-deforestation oil palm cropland was equal to the price elasticity of forest-to-palm conversion, i.e. 0.13, meaning that every 1% increase in the price of palm oil was associated with a 0.13% increase in high-deforestation oil palm cropland. The price elasticity of supply for low-deforestation oil palm cropland was smaller than the price elasticity of non-forest-to-palm conversion because a substantial area of low-deforestation oil palm cropland had already been established by the year 2000; this area was unaffected by later price changes. As a result, we estimated the price elasticity of supply of low-deforestation oil palm cropland to be 0.033, meaning that every 1% increase in the price of palm oil was associated with a 0.033% increase in low-deforestation oil palm cropland. Combined, the price elasticity of supply for all Indonesian oil palm cropland was 0.086,

meaning that every 1% increase in the price of palm oil was associated with a 0.086% increase in oil palm cropland.

A European ban on consuming high-deforestation palm oil decreased exports of high-deforestation palm oil from Indonesia and Malaysia to Europe by 99% by design of our experiment. We simulated that a European ban would have increased exports of low-deforestation palm oil from Indonesia and Malaysia to Europe by 31.1% (table 1). About half (52%) of the high deforestation trade flow that would otherwise have been exported to Europe shifted to other regions (figure 1), resulting in a 9.5% decrease in the volume of high-deforestation palm oil exported from Indonesia and Malaysia and a 1.4% increase in the volume of low-deforestation palm oil exported from Indonesia and Malaysia (table 1). The global share of palm oil exports from Indonesia and Malaysia consumed by Europe would have fallen from 21% to 14% (table SI5).

A European ban on consuming high-deforestation palm oil would have produced an 8.9% price premium for low-deforestation palm oil, composed of a 7.5% price decrease for high-deforestation palm oil and a 0.7% price increase for low-deforestation palm oil (table 1). The quantity of high-deforestation palm oil produced in Indonesia would have decreased by 6.3%, while the quantity of low-deforestation palm oil would have increased by 0.3% (table SI6). Global production of palm oil would have decreased by 1.32%, with shifts in production among oilseeds resulting in a decrease in global production of edible oils of 0.44% (table SI7).

These changes in the market for palm oil would have resulted in 320 609 hectares (5.03%) less conversion of forest to oil palm plantations in Indonesia from 2000 to 2015 (21 374 ha yr⁻¹), with a resulting reduction in associated emissions of 315.9 million tCO₂ (21.1 million tCO₂ yr⁻¹; 5.86%) (table 2). This would have reduced Indonesia's total deforestation of 20.0 million hectares from 2000 to 2015 by 1.60%, and its total emissions from deforestation of 16.6 billion tCO₂ by 1.91% (figure 2). Considering 11.2 billion tCO₂ in emissions from sectors other than land-use change and forestry (Climate Watch 2020), this would have reduced Indonesia's emissions from all sources by 1.14%. The deforestation emission reductions of 1.91% from a European ban would have been equivalent to a hypothetical mandatory Indonesia-wide price on carbon emissions from deforestation of \$0.81/tCO₂, or a voluntary price of \$1.57/tCO₂ in 2013 USD (table 2).

A ban on high-deforestation palm oil by a combined bloc of Europe and the United States would have reduced deforestation by 365 015 hectares (24 334 ha yr⁻¹; 5.72% of deforestation associated with conversion of forest to oil palm plantation; and 1.83% of all deforestation) and reduced emissions

Table 1. Effects of demand-side restrictions on importing high-deforestation palm oil on exports of palm oil from Indonesia and Malaysia and producer prices. Values in parentheses represent endpoints of the 95% confidence interval of a joint sensitivity analysis to seven key model parameters in GIAP-BIO.

| Scope of ban on high-deforestation palm oil | Type of palm oil | Change in palm oil exports from Indonesia and Malaysia to regions (%) | | | | | | | | | | Change in producer price of palm oil in Indonesia and Malaysia (%) | Price premium for low-deforestation palm oil (%) |
|---|--------------------|---|-------------------------|----------------------|----------------------|-------------------------|-------------------------|-------------------------|-------------------------|----------------------|--|--|--|
| | | EU27 | USA | China | Brazil | Other South America | Other | World ^b | | | | | |
| EU27 | High-deforestation | -99.0 ^a | 14.4 (13.2, 15.6) | 12.8 (11.2, 14.4) | 53.3 (43.1, 63.5) | 59.4 (45.4, 73.4) | 13.3 (12.3, 14.3) | -9.5 (-10.3, -8.7) | -7.5 (-8.1, -6.9) | 8.9 (8.8, 8.9) | | | |
| | Low-deforestation | 31.1 (25.1, 37.1) | -2.1 (-3.3, -0.9) | -7.0 (-8.4, -5.6) | 1.5 (1.3, 1.7) | 1.3 (-0.1;2.7) | -6.6 (-7.8, -5.4) | 1.4 (1.0, 1.8) | 0.7 (0.1-1.3) | | | | |
| | High-deforestation | -99.0 ^a | -99.0 ^a | 14.7 (12.9, 16.5) | 56.5 (45.9, 67.1) | 67.0 (51.2, 82.8) | 15.5 (14.3, 16.7) | -11.3 (-12.3, -10.3) | -8.6 (-9.4, -7.8) | 10.0 (9.7, 10.3) | | | |
| EU27 + USA | Low-deforestation | 31.7 (25.7, 37.7) | 7.6 (4.8, 10.4) | -7.5 (-8.9, -6.1) | 1.5 (1.3, 1.7) | 2.0 (0.4, 3.6) | -7.0 (-8.2, -5.8) | 1.5 (1.1, 1.9) | 0.5 (-0.1, 1.1) | 24.3 (23.4, 25.1) | | | |
| | High-deforestation | -99.0 ^a | -99.0 ^a | -99.0 ^a | 74.7 (62.3, 87.1) | 100.5 (77.9, 123.1) | 30.3 (28.3, 32.3) | -21.1 (-23.8, -18.4) | -12.7 (-11.7-13.7) | | | | |
| | Low-deforestation | 14.4 | -4.7 (-8.5, -0.9) | 87.0 (80.0, 94.0) | -6.0 (-6.4, -5.6) | -17.4 (-21, -13.8) | -20.2 (-22.0, -18.4) | 5.1 (4.7, 5.5) | 8.5 (6.5-10.5) | | | | |
| All except Indonesia and Malaysia | High-deforestation | -99.0 ^a | -99.0 ^a | -99.0 ^a | -99.0 ^a | -99.0 ^a | -99.0 ^a | -94.9 (-95.1, -94.7) | -19.8 (-20.4, -19.2) | 80.7 (69.0, 92.2) | | | |
| | Low-deforestation | -23.5 (-27.9, -19.1) | -34.0 (-38.6, -29.4) | 50.3 (41.9, 58.7) | 11.5 (5.3, 17.7) | -44.4 (-50.0, -38.8) | 22.2 (17.6, 26.8) | 12.8 (11.5, 14.3) | 44.9 (34.5, 55.3) | | | | |

^a 99% reduction in imports of high-deforestation palm oil is by construction as a fixed input to the trade model.

^b Includes exports between Indonesia and Malaysia.

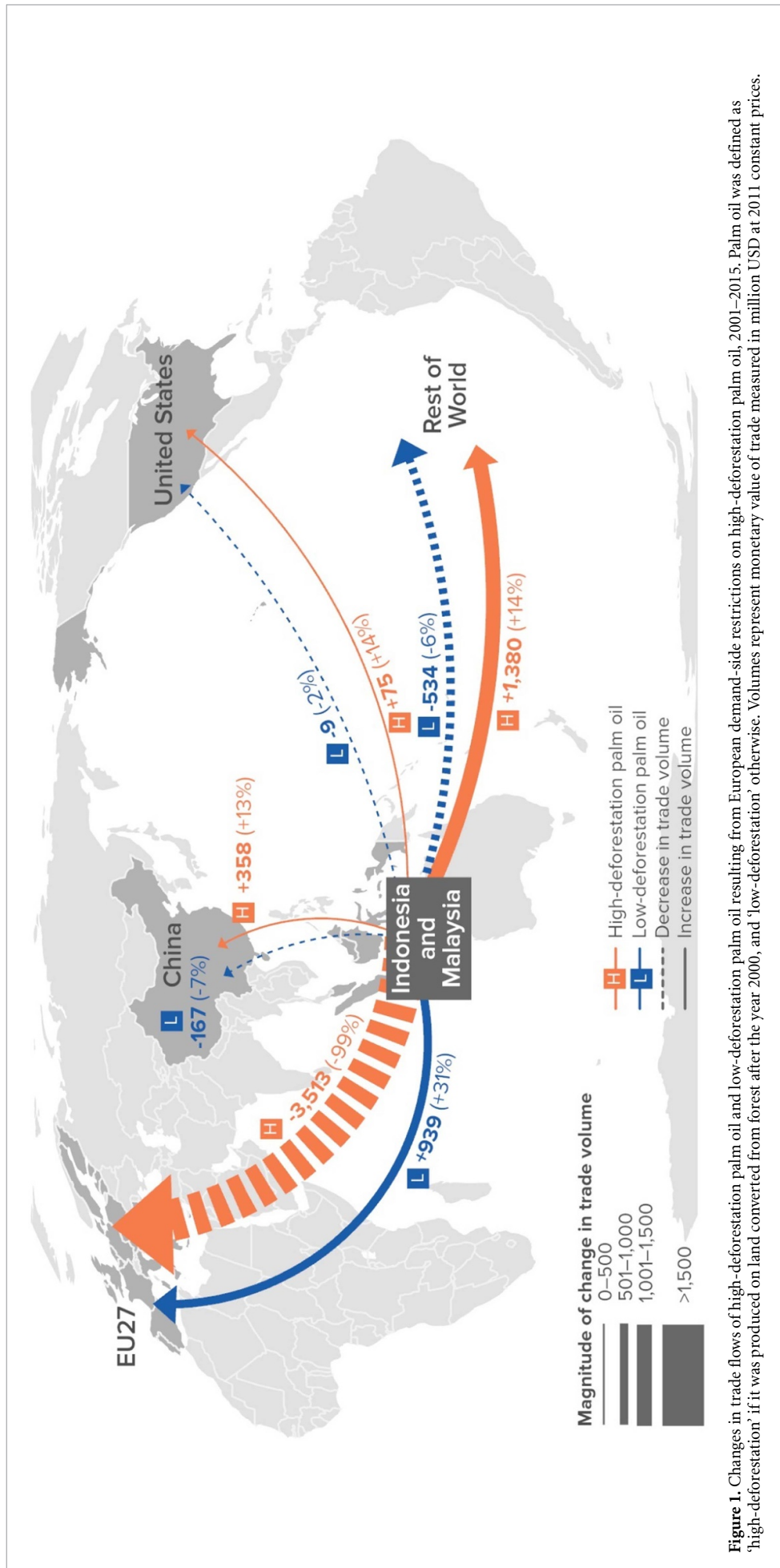


Figure 1. Changes in trade flows of high-deforestation palm oil and low-deforestation palm oil resulting from European demand-side restrictions on high-deforestation palm oil, 2001–2015. Palm oil was defined as ‘high-deforestation’ if it was produced on land converted from forest after the year 2000, and ‘low-deforestation’ otherwise. Volumes represent monetary value of trade measured in million USD at 2011 constant prices.

Table 2. Effects of demand-side restrictions on high-deforestation palm oil on land-use change and emissions in Indonesia, 2000–2015. Values in parentheses represent endpoints of the 95% confidence interval of a joint sensitivity analysis to seven key model parameters in GTAP-BIO.

| Scope of ban on high-deforestation palm oil | Change in oil palm price (land rental rate) (%) | Change in conversion of forest to oil palm (%) | Change in conversion of non-forest to oil palm (%) | Change in deforestation (%) | Change in emissions from conversion of forest to oil palm (%) | Change in total emissions from deforestation (%) | Equivalent mandatory price (2013 USD/tCO ₂) | Equivalent voluntary carbon price (2013 USD/tCO ₂) |
|---|---|--|--|-----------------------------|---|--|---|--|
| EU27 | -39.8 (-41.6, -36.3) | -5.03 (-5.25, -4.60) | 0.69 (0.16, 1.24) | -1.60 (-1.67, -1.47) | -5.86 (-6.11, -5.36) | -1.91 (-1.99, -1.75) | 0.81 (0.74, 0.85) | 1.57 (1.49, 1.61) |
| EU27 + USA | -45.6 (-47.7, -41.5) | -5.72 (-5.97, -5.23) | 0.50 (-0.09, 1.08) | -1.83 (-1.90, -1.67) | -6.67 (-6.96, -6.10) | -2.17 (-2.27, -1.99) | 0.93 (0.85, 0.97) | 1.71 (1.61, 1.76) |
| EU27 + USA + China | -68.5 (-71.0, -63.6) | -8.38 (-8.66, -7.82) | 8.03 (5.98, 10.13) | -2.67 (-2.76, -2.49) | -9.76 (-10.09, -9.11) | -3.18 (-3.29, -2.97) | 1.36 (1.27, 1.40) | 2.27 (2.15, 2.33) |
| All except Indonesia and Malaysia | -99.97 | -11.84 | 57.03 (40.64, 76.15) | -3.77 | -13.79 | -4.49 | 1.92 | 2.90 |

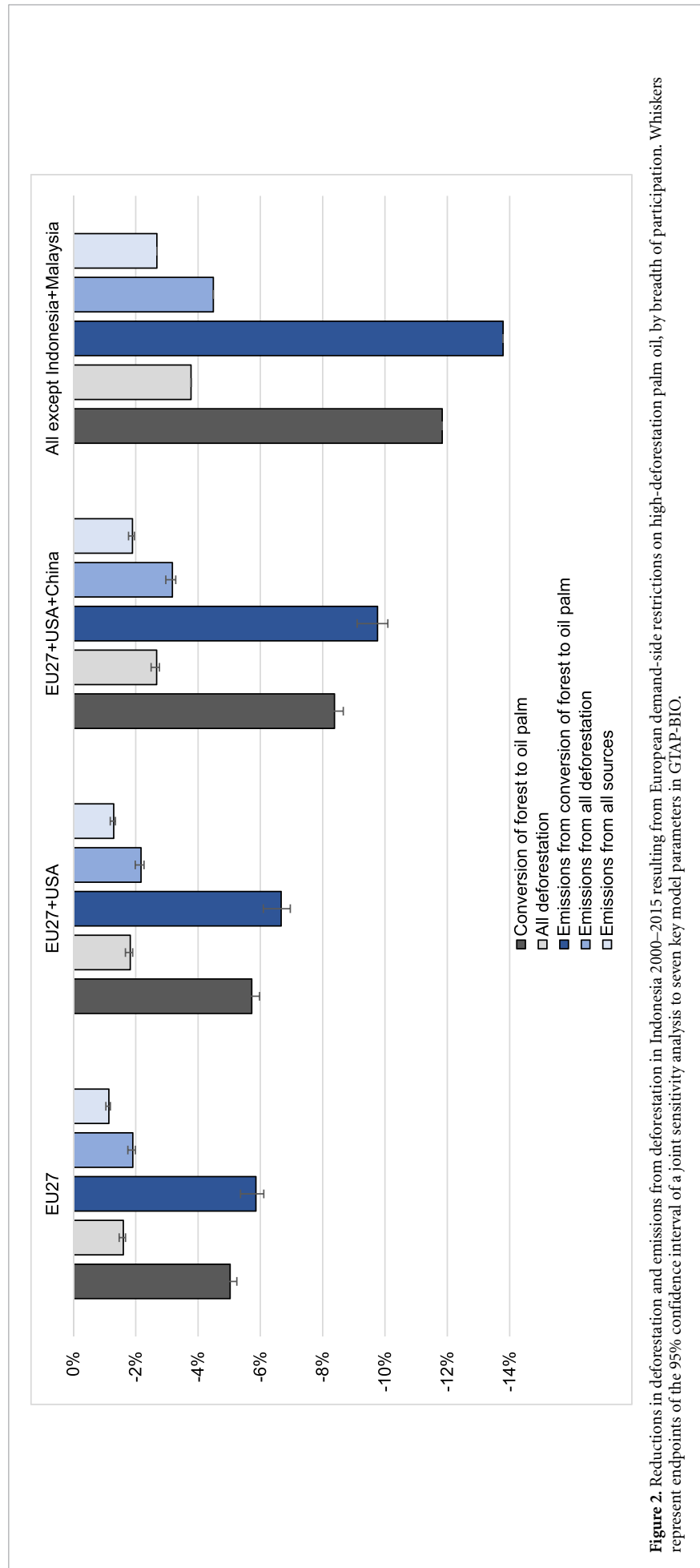


Figure 2. Reductions in deforestation and emissions from deforestation in Indonesia 2000–2015 resulting from European demand-side restrictions on high-deforestation palm oil, by breadth of participation. Whiskers represent endpoints of the 95% confidence interval of a joint sensitivity analysis to seven key model parameters in GIAP-BIO.

by 359.6 million tCO₂ from 2000 to 2015 (24.0 million tCO₂ yr⁻¹; 6.67% of emissions associated with conversion of forest to oil palm plantation; 2.17% of all emissions from deforestation; 1.30% of emissions from all sources). A ban by Europe, the United States, and China would have reduced deforestation by 534 158 hectares (35 611 ha yr⁻¹; 8.38%; 2.67%) and reduced emissions by 526.3 million tCO₂ from 2000 to 2015 (35.1 million tCO₂ yr⁻¹; 9.76%; 3.18%; 1.90%). A ban by all countries outside Indonesia and Malaysia would have reduced deforestation by 754 731 hectares (50 315 ha yr⁻¹; 11.84%; 3.77%) and reduced emissions by 743.7 million tCO₂ from 2000 to 2015 (49.6 million tCO₂ yr⁻¹; 13.79%; 4.49%; 2.68%) (table 2). Alternative emission factors related to biomass and soil changed the estimates of percent reductions in emissions by just 2%–7% (table SI8).

4. Discussion

European demand-side restrictions on high-deforestation palm oil from 2000 to 2015 would have resulted in a moderate price premium for low-deforestation palm oil (8.9%), but only minor impacts on deforestation (–1.60%) and emissions from deforestation (–1.91%) in Indonesia (table 2). The impact of demand-side restrictions would be small because: about half (52%) of the high-deforestation palm oil that would otherwise have been exported to Europe would be absorbed by increased consumption in non-participating countries, including domestic consumption in Indonesia; the price elasticity of supply of high-deforestation oil palm cropland is small (0.13), meaning relatively large changes in price result in relatively small changes in quantity; and conversion to oil palm was responsible for only 32% of deforestation in Indonesia.

A benchmark policy of a mandatory Indonesia-wide price on deforestation emissions would have been as effective at \$0.81/tCO₂ in 2013 USD (table 2). This price is one-to-two orders of magnitude lower than carbon market prices circa April 2021 of \$9/tCO₂ in the Regional Greenhouse Gas Initiative, \$16/tCO₂ in Korea, \$18/tCO₂ in California and Quebec, \$26/tCO₂ in New Zealand, and \$50/tCO₂ in the European Union (World Bank 2021).

More consumer countries implementing demand-side restrictions would result in greater emission reductions. By following Europe's lead, other countries would avoid first-mover costs of designing policies. However, even under very optimistic assumptions about participation by consumer countries, demand-side restrictions on high-deforestation commodities would still have only modest effects on deforestation and emissions; e.g. a ban on high-deforestation palm oil by all countries outside Indonesia and Malaysia from 2000 to 2015 would have reduced deforestation in Indonesia

by just 3.77% and emissions from deforestation in Indonesia by just 4.49% (table 2).

As modest as our estimates are, they likely overstate the impacts of demand-side restrictions on palm oil for several reasons. Actual demand-side policies have been narrower in scope than we modeled, covering only the portion of palm oil used in biofuels (e.g. Norway), or only the portion of palm oil used in public procurement (e.g. Netherlands, France), or only the portion of commodity-associated deforestation that is illegal (e.g. US, UK; Dos Reis *et al* 2021). On the supply side, the definition of deforestation used by RSPO is also narrower in scope than we modeled, covering deforestation only after 2005 rather than 2000, and only primary forest rather than all forest. Importantly, we assumed that restrictions on palm oil use would be fully effective. We did not consider, for example, the ability of producers of high-deforestation commodities to circumvent restrictions by disguising their product as low-deforestation (e.g. Reuters 2020). We also did not consider implementation or transaction costs of verifying or certifying palm oil as low-deforestation, which can be considerable and can pose disproportionate challenges to smallholders (Brandi *et al* 2015).

Conversely, there are also reasons our estimates could understate policy impacts. Many of the demand-side policies that have been passed or are under consideration cover multiple forest-risk commodities, and thus would have an increased effect on deforestation. Furthermore, the presence of 'stickiness' in commodity supply chains (Dos Reis *et al* 2020) could result in smaller geographical shifts in consumption, and thus larger changes in prices and greater reductions in deforestation.

Several limitations with our data and model introduce uncertainty to our estimates of reductions in deforestation and emissions from deforestation. As elaborated in the supplementary information, these include the inclusion of tree cover other than forest in our land-cover change data; the exclusion of many smallholder plantations from our oil palm coverage data; and the inability to disaggregate Indonesia from Malaysia in our global trade model. Future data improvements could lead to more accurate estimates. However, we do not expect that improving these data would alter our central finding that the effects of trade restrictions would be small due to the three factors we have identified.

The effects of demand-side policies on deforestation in other regions, commodities, and time periods will depend on the same three factors: share of global consumption comprised by the countries participating in a ban; price elasticity of supply for the high-deforestation variety of the commodity; and share of deforestation comprised by the commodity. For example, the effects of European demand-side restrictions on oil palm-driven deforestation in Indonesia would likely be smaller in the present

(e.g. 2021–2030) than for the earlier 2000–2015 time period that we analyzed. This is because two of the three factors influencing impact have decreased: the European share of global palm oil consumption has declined (USDA 2019), and the share of Indonesia's deforestation due to oil palm expansion has diminished (Austin *et al* 2019, Gaveau *et al* 2019).

If demand-side restrictions are to succeed in substantially reducing deforestation, therefore, it would have to be through non-price pathways that are beyond the scope of our analytical model. These could include social pathways (e.g. raising consumer awareness, imposing stigma, changing social norms); industrial organization pathways (e.g. commodity producers or supply chain intermediaries applying European standards to exports to all markets and not just exports to Europe); or synergistic effects with other domestic, bilateral, or corporate forest conservation actions. Conversely, there are also non-price pathways that could increase deforestation. For example, we did not consider whether import restrictions constitute protectionism for farmers in wealthy countries in violation of World Trade Organization rules. Backlash to perceived protectionism has resulted in some tropical countries enacting compensatory domestic price support policies, e.g. biofuel mandates in Indonesia (Elliott 2015). Such non-price effects, both positive and negative, could have a larger impact on land-use changes than direct price effects.

Our findings imply that demand-side restrictions on high-deforestation commodities alone are insufficient to substantially counteract distal deforestation. Consumer countries seeking to reduce deforestation in commodity-producing countries should support more direct forest and climate protection policies such as carbon payments, which even at a very low price could meet or exceed the effects of demand-side restrictions.

Data availability statement


The data that support the findings of this study are available upon reasonable request from the authors.

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