Cerulogy

Soy, land use change and ILUC-risk

A review

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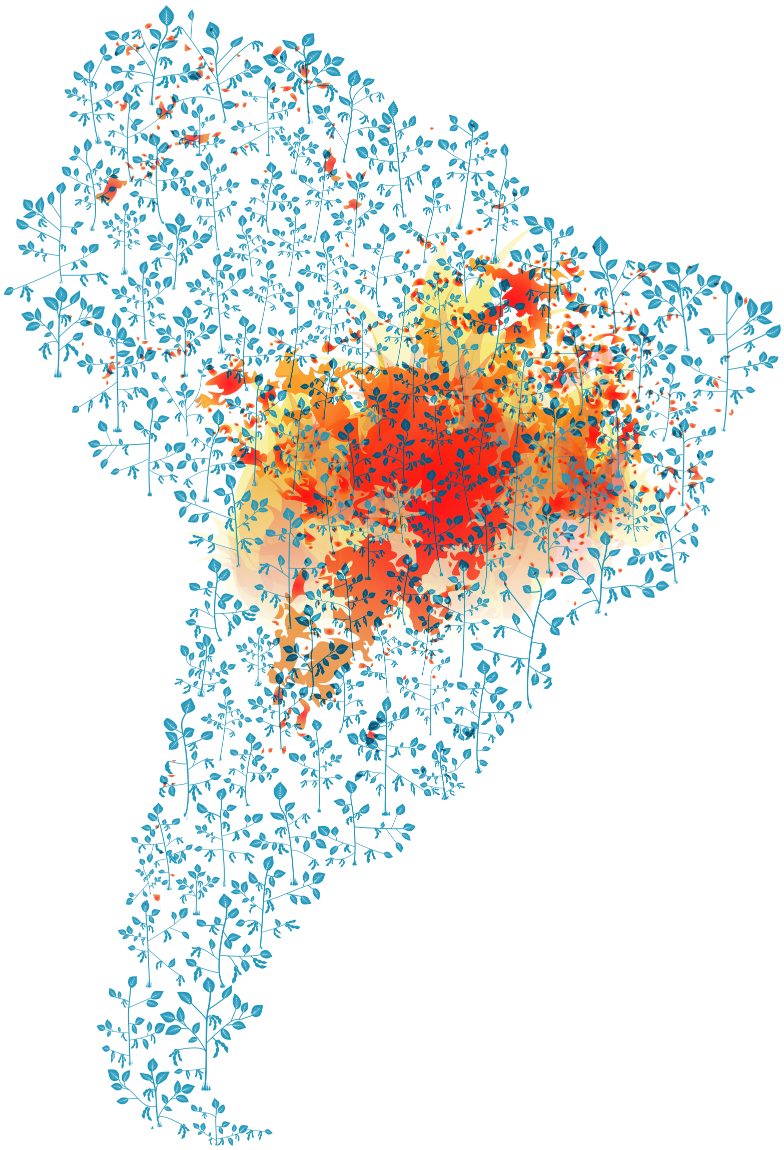
November 2020



Acknowledgements

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Executive Summary

Over the past 15 years, biofuel policy in Europe has created an additional source of soy oil demand, with significant volumes of material imported either in raw form or processed into biodiesel. As of 2019, something like 2 billion litres a year of soy oil biofuels were being consumed in the EU. Through this biofuel demand, as well as through imports of soy meal as livestock feed, the EU has contributed to an export business that, in South America in particular, has long been identified as a major driver of deforestation. The EU's recast Renewable Energy Directive (RED II) continues to offer support for food-based biofuels including soy oil biodiesel and renewable diesel[1], but introduces a new category of 'high ILUC-risk' biofuel feedstocks, for which support will be gradually eliminated between 2023 and 2030. Palm oil has been labelled as high ILUC-risk, but while soy oil was identified as the biofuel feedstock second most strongly associated with conversion of high carbon stock areas the initial EU assessment found that it was below the threshold for action.

Between 2004 and 2010 of a range of measures were introduced to manage tropical forest loss in South America, including in the major soy producing countries (Brazil, Argentina and Paraguay). In Brazil the combination of increased enforcement of the forest code and the introduction of a moratorium on deforestation for soy expansion in the Amazon were heralded as major landmarks in the fight to reduce deforestation. Official government satellite deforestation monitoring showed impressive reductions from 2004 onwards (blue line on Figure 1), but by 2012 the downward trend was reversing and the last two years have seen a deforestation uptick that many commentators have associated with a new presidential administration with less sympathy for environmental protection. Other satellite tools, such as the University of Maryland's Global Forest Change (GFC) database, suggest that tree cover loss rates are higher than shown by the government data, which could suggest that deforesters have been operating in ways that take advantage of limitations in the government detection system (red line on Figure 1).

Beyond the Amazon, deforestation remains a threat to the Brazilian Cerrado, to the Chaco in Argentina, Brazil and Paraguay, to the Atlantic forest in Paraguay and to forest systems across South America. Increased forest protection in some ecosystems may have contributed to accelerated losses in less protected ones by shifting patterns of agricultural expansion (Figure 2).

1 'Biodiesel' refers to fatty acid methyl ester (FAME), which is produced by reacting vegetable oil with methanol and can be used in diesel engines, normally in low blends with conventional diesel. 'Renewable diesel' refers to hydrotreated vegetable oil (HVO), which is produced by reacting vegetable oil with hydrogen and which can be used in diesel engines either pure or blended at any rate with conventional diesel.

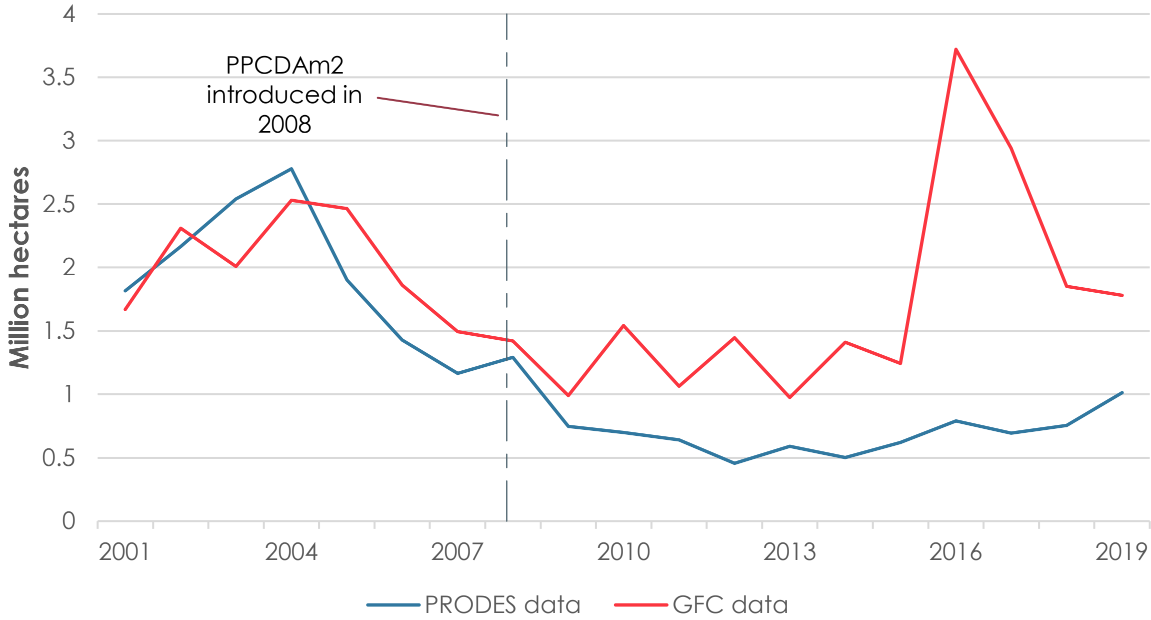

Figure 1. Amazon deforestation as monitored by PRODES and tree cover loss as monitored by Global Forest Change (GFC)

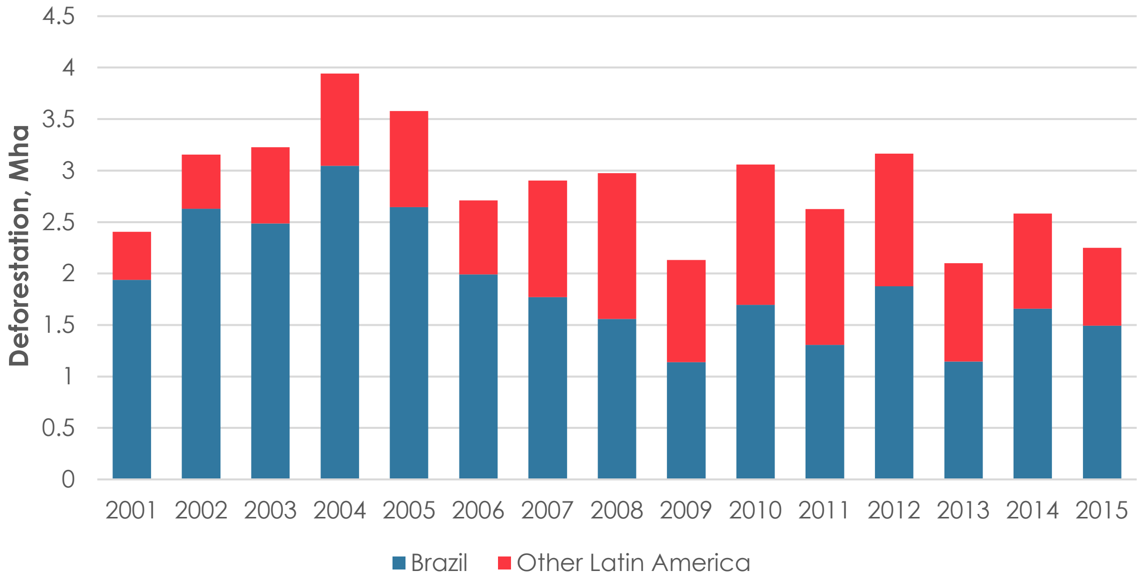

Figure 2. Commodity associated forest loss in South America

Source: Curtis et al. (2018)

In short, deforestation remains a major threat to biodiversity and driver of climate change. In particular, the most recent available analysis shows that deforestation for agricultural expansion has continued in South America's major soybean producing countries across a number of biodiversity and carbon rich biomes. Soybean expansion is not the primary direct driver of deforestation in these areas. It is not always simple to identify proximate causes of tree cover loss, especially to the level of distinguishing the impact of individual crops, but in the areas of most interest (the Amazon, the Cerrado, the Chaco) we can conclude that pasture expansion remains the dominant proximate driver of deforestation, with cropland expansion more likely to occur on previously cleared pasture land more than on newly cleared forest areas. Even where soy expansion does not intrude on recently deforested land in the years immediately after clearance, the soy industry could still be indirectly contributing to forest loss. The profitability of the soy industry can be a source of capital for pasture expansion – selling pasture land away from the forest frontier to soy farmers could be a way to raise funds to expand land holdings into the forest, and some authors argue that cattle ranching and soy farming must be understood as fundamentally coupled industries. Still, analysis of the impacts of the Amazon soy moratorium has failed to provide a 'smoking gun' to prove that forest loss has simply leaked out of the soy industry into the livestock industry, suggesting that taking action on deforestation in a single market can be at least partly effective.

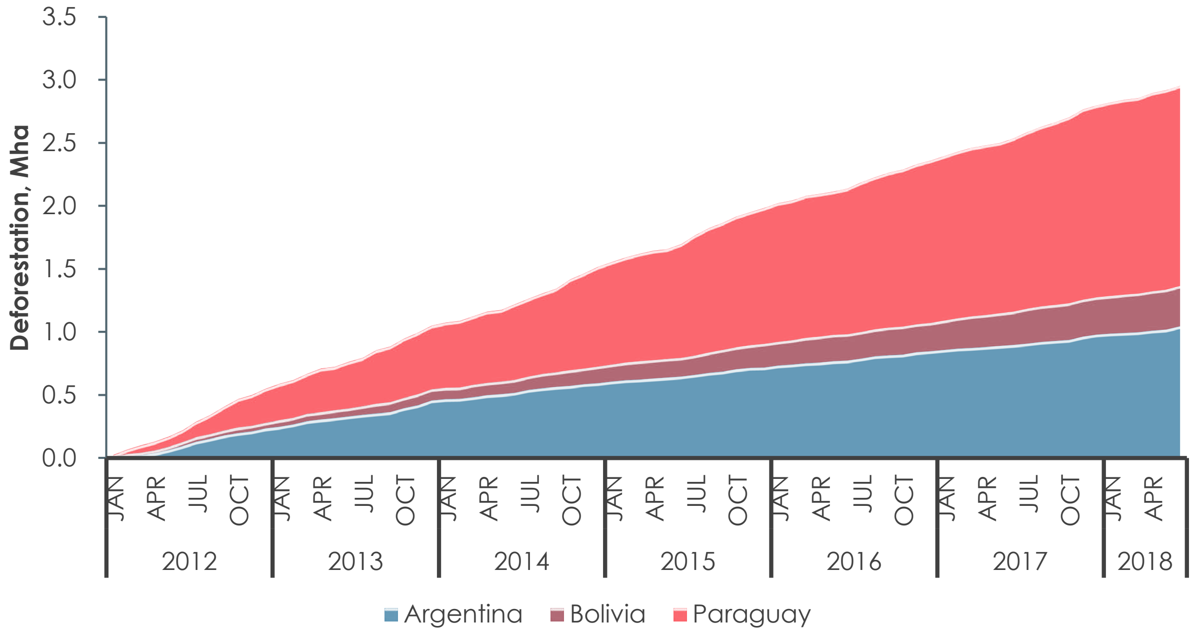

Figure 3. Chaco deforestation January 2012 – June 2018 by country: a) monthly reported; b) cumulative

Soy may not be the proximate driver of as much deforestation in South America as the livestock industry is, but the available analysis suggests that a significant fraction of cropland expansion in general and soy expansion in particular in these regions continues to occur at the expense of forests. Deforestation rates in the Brazilian Cerrado have been relatively steady as Amazon deforestation has been reported as decreasing. Since the European Commission's assessment of high ILUC-risk feedstocks in 2019 (European Commission, 2019a), we calculate[2] that an increased fraction of global soy expansion has occurred in the Brazilian Cerrado, and there is evidence (Noojipady et al., 2017)that a larger fraction of cropland expansion in the

2 Based on Brazilian national agricultural statistics and data from FAOstat.

Cerrado may have occurred on high carbon stock land than was assumed in the previous assessment. Deforestation rates in the Chaco also remain relatively steady despite nominal efforts to address deforestation in both Argentina and Paraguay (Figure 3).

In this report we have revisited the calculation of the fraction of soy expansion affecting high carbon stock areas that was undertaken by European Commission (2019c). Integrating the more recent evidence we estimate it as 10.5%, which is above the threshold for categorisation as a high ILUC-risk biofuel. If ongoing research for the Commission confirms this result, then soy oil biofuels will need to be phased out from support under the EU's RED II by 2030.

We estimate that in the absence of a high ILUC-risk classification between 3 and 6 billion litres of soy oil would be used to produce biofuels for the EU in 2030. A high-ILUC risk categorisation would eliminate that source of demand. Avoiding the ILUC emissions associated with that volume could potentially reduce net emissions by tens of millions of tonnes of CO_2e, although the size of that benefit is highly sensitive to the estimated size of ILUC emissions. While the magnitude of the potential benefit is subject to the same uncertainty as all efforts to estimate ILUC emissions, significantly better climate outcomes could be delivered the transition to advanced biofuels or other low carbon transport alternatives could be accelerated.

While the soy oil biofuel market in the EU is considerable, the EU's demand for soy biofuels is secondary in terms of development of the global soy market to continued expected growth in demand of soy as livestock feed. Just as characterising palm oil as high ILUC-risk will reduce the growth rate of global palm oil demand rather than leading to any absolute reduction, the soy market will be expected to continue to grow through the coming decade requiring increased area even given forecast productivity gains – any use of soy oil for EU biofuels adds to that existing pressure. China's expanding appetite for soybeans to feed its livestock sector has been the most important driver of soy industry expansion for the last 15 years, and while this rate of expansion looks set to reduce beyond 2020 it is still a potential threat to forests.

Contents

Glossary of abbreviations

AEZ	Agro-ecological zone
DETER	Real Time System for Detection of Deforestation in the Brazilian Amazon
FAO	Food and Agriculture Organisation of the United Nations
FAS	Foreign Agricultural Service of the U.S. Department of Agriculture
FGP	First gathering point
FSA	Farm Service Agency of the U.S. Department of Agriculture
GAIN	Global Agricultural Information Network of the USDA FAS
GFC	Global Forest Change dataset of the University of Maryland
GHG	Greenhouse gas
IBGE	Brazilian Institute of Geography and Statistics
ILUC	Indirect land use change
INPE	Brazilian National Institute of Space Research
ISCC	International Sustainability and Carbon Certification
LCA	Lifecycle Analysis
LCFS	Low carbon fuel standard
MATOPIBA	Maranhão, Tocantins, Piauí and Bahia
MODIS	Moderate Resolution Imaging Spectroradiometer
NASA	U.S. National Aeronautical and Space Administration
NGO	Non-governmental organisation
NRCS	Natural Resource Conservation Service of the U.S. Department of Agriculture
OECD	Organisation for Economic Cooperation and Development
PRODES	Program for Monitoring Deforestation of the Amazon by Satellite
RED	Renewable Energy Directive (EU)
RFS	Renewable Fuel Standard (U.S.)
RSB	Roundtable on Sustainable Biomaterials
SAD	Deforestation Alert System of the Imazon (sic) institute
SIDRA	IBGE Automatic Recovery System
SSAP	Soybean Sustainability Assurance Protocol
USDA	U.S. Department of Agriculture

1. Introduction

Soy farming has increased dramatically in both output and area farmed over the past fifty years, and is a major export crop for several South American countries. Soy produces two products, soy meal for livestock feed and soy oil for human consumption and (in recent decades) biofuel production. It is the third most used food oil for biodiesel and renewable diesel production in the EU. In South America soy has become strongly associated with deforestation, notably in the Amazon rainforest but also in other forest biomes.

Soybeans are farmed primarily for three markets – for human consumption, for animal feed and for vegetable oil. Vegetable oil is extracted if the beans are crushed in mills instead of being fed directly to livestock. Crushing results in two co-products – a larger fraction of soy meal (about 80% by mass), which is used as a high protein animal feed, and a smaller fraction of soy oil (about 20% by mass), which is primarily sold as a cooking oil or as a biofuel feedstock, although there are also oleochemical and animal feed uses for which soy oil would be appropriate.

Soybean production has increase rapidly in the past 50 years from less than 50 million tonnes globally in 1970 to 350 million tonnes in 2018, and is now heavily concentrated in just a few countries. As shown in Figure 4, 95% of the world's soy is produced by the USA, China, India, Canada and the nations of South America, primarily Brazil and Argentina.

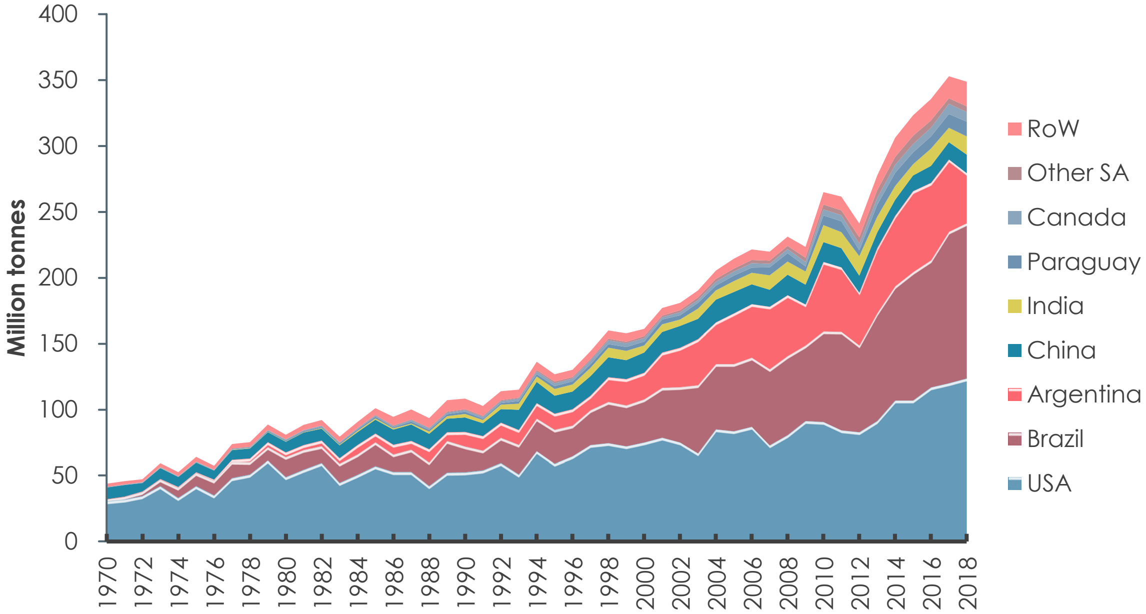

Figure 4. Global soybean production 1970 - 2018

Source: FAOstat

Accompanying this expansion of production has been a great growth in the area planted with soybeans. The need for increased area has been offset to a considerable extent by increases in productivity – for instance, between 1970 and now average yields reported by FAOstat[3] for the U.S. soy crop have nearly doubled while average yields in Brazil have nearly trebled. Still, total soybean harvested area has increased by 95 million hectares since 1970, a factor of four (Figure 5). That increase is about the same area as the total arable land of Brazil and Argentina combined.

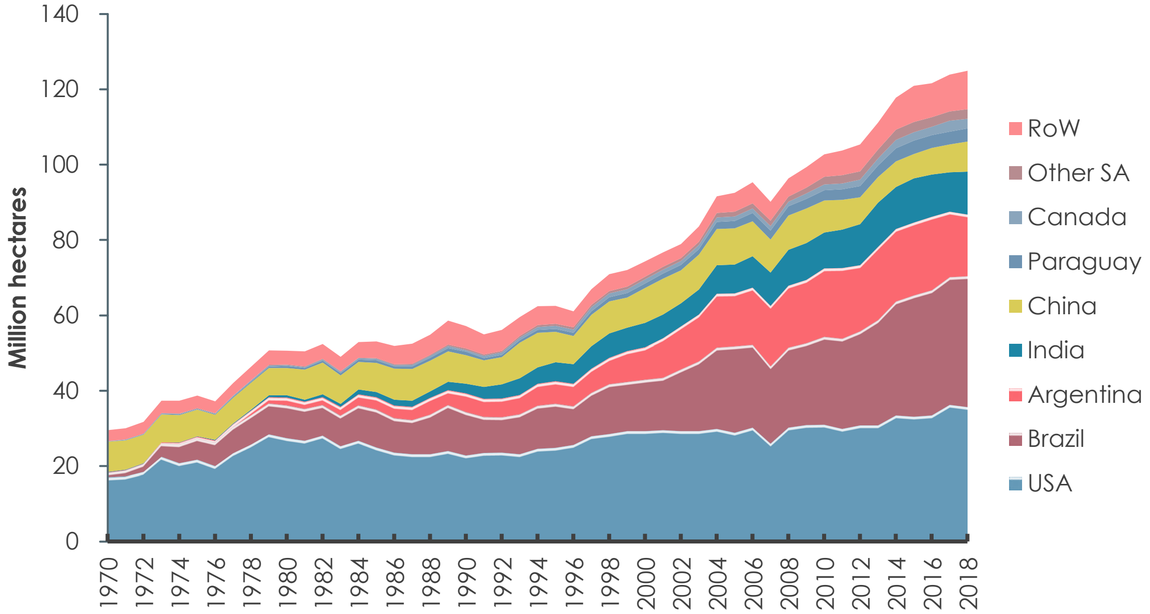

Figure 5. Global soybean harvested area 1970 - 2018

Source: FAOstat

The negative side of this great increase in production is that soybean agriculture is widely recognised as one of the world's most important drivers of tropical deforestation. A recent review of the link between EU commodity demand and deforestation for the European Commission (Ecofys et al., 2018) found that soy is consistently identified in the literature on forest-risk commodities as being strongly linked to deforestation (Brack, 2015; GEF Secretariat, 2014; Henders et al., 2015; Lammerant et al., 2014; Rautner et al., 2013; UNEP, 2015; Walker et al., 2013). The United Nations Food and Agriculture Organisation's 2020 "State of the World's Forests" report states that, "Large-scale commercial agriculture (primarily cattle ranching and cultivation of soya bean and oil palm) accounted for 40 percent of tropical deforestation between 2000 and 2010." Relatedly, modelling of indirect land use change emissions expected due to the use of soy oil as a biofuel feedstock has generally found that land use changes are likely to significantly reduce or eliminate any net climate benefit[4] from use of such biofuels (cf. Laborde, 2011; Malins, 2019; Valin et al., 2015; Woltjer et al., 2017).

Given these concerns about the impact on tropical forests of further increases in soy demand,

3 Agricultural statistics published by the Food and Agriculture Organisation of the United Nations, http://www.fao.org/faostat/en/

4 Over a 20 year time frame.

it is unsurprising that the use of soy oil as a biofuel feedstock remains the subject of much interest and some controversy in the academic and stakeholder community.

1.1. EU policy context

In the period from 2003 until about 2014, EU biofuel policy aimed to support continuing increase in the use of biofuels produced from crop commodities (primarily sugars, starchy grains and vegetable oils), albeit subject to some basic sustainability oversight[5]. However, from 2008 growing concerns relating to the impact of ever-increasing biofuel demand on land use change and food availability led to a policy debate, spanning several years, in which a number of options were considered to manage the indirect impacts of EU biofuel consumption.

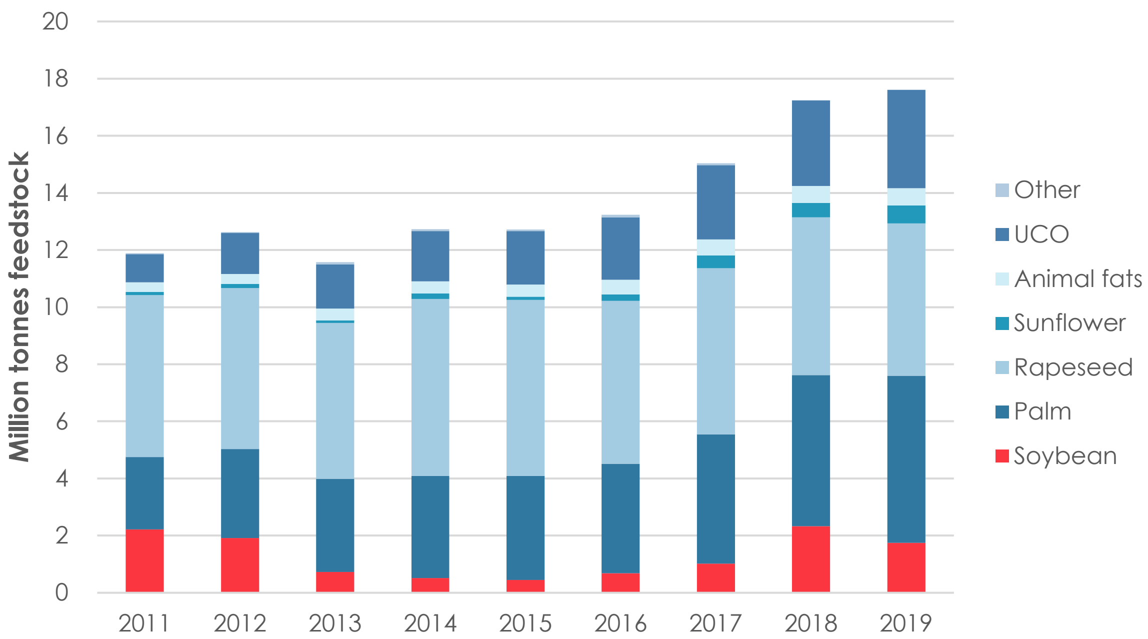

Figure 6. Feedstocks for EU biodiesel and renewable diesel consumption, 2011-2019

Source: Cerulogy estimate based on OilWorld (2020) (via Transport and Environment, 2020) and Comext[6] trade statistics.[7]

Note: OilWorld include estimated PFAD use under palm oil

5 Prohibition on sourcing biofuel feedstock from specified recently converted areas that previously had high carbon stocks or high biodiversity value, and a maximum threshold on the GHG intensity of the farming and fuel production process.

6 http://epp.eurostat.ec.europa.eu/newxtweb/

7 For EU processed biofuel the feedstock estimates is taken from OilWorld. For imports, feedstock assumption is based on country of origin. Biodiesel from South America is treated as soy based, biodiesel from the rest of the EU as rapeseed based, biodiesel from China, India and Pakistan as UCO based and other biodiesel from Southeast Asia as palm oil based. Imports of renewable diesel are not identified in Comext data and have not been included.

The legal framework changed formally in 2015 with the introduction of the 'ILUC Directive'[8], with a cap brought in on the contribution made by crop-based biofuels to meeting EU targets. As shown in Figure 6, there has been some continued growth in the use of food-grade vegetable oils as biodiesel feedstock for EU consumption since the ILUC Directive came into force, but also increased focus on the supply of used cooking oil, animal fats and other lower grade oils.

The recast of the Renewable Energy Directive ("RED II")[9] for the period from 2021 to 2030 allows Member States to further limit support for food-commodity-based fuels, and introduces a new category of 'high ILUC-risk' biofuels, support for which must be phased out by 2030. Whereas biofuel policies in the United States (RFS and LCFS) include terms for ILUC emissions in the associated lifecycle analysis (LCA) methodologies, ILUC emissions are considered outside the 'system boundary' for GHG intensity assessment under the RED. The high-ILUC-risk approach is intended to identify the biofuels for which the risk of ILUC emissions is considered greatest.

High ILUC-risk fuels are defined as fuels from feedstocks for which, "significant expansion of the production into land with high-carbon stock is observed". The threshold for significance was established in a subsequent delegated regulation[10] - a feedstock would be considered high ILUC-risk if on average 10%[11] or more of new area for that feedstock globally is established on land meeting the high carbon stock definitions. Note that this assessment considers only the direct impact on high carbon stock land of each feedstock – it is specifically designed to work around the difficulties associated with assessing indirect effects.

Analysis by the Commission (European Commission, 2019c) identified palm oil as the feedstock most strongly associated with the conversion of high carbon stock ecosystems, with an estimated 45% of palm oil expansion in the period 2008-17 being identified as directly associated with deforestation, and 23% directly associated with peat drainage. Palm oil will be subject to the high ILUC-risk provisions requiring a phase out of support by 2030 unless this assessment or the high ILUC-risk rules are changed during review. The Commission is required to review data on feedstock expansion by 30 June 2021 and review the delegated regulation by 1 September 2023.

Soy was identified in this analysis as the feedstock the second most strongly associated with deforestation, with 8% of expansion occurring on previously high carbon stock land, but this falls below the 10% threshold set in the delegated regulation for a 'significant' amount of

8 Directive (EU) 2015/1513 of the European Parliament and of the Council of 9 September 2015 amending Directive 98/70/EC relating to the quality of petrol and diesel fuels and amending Directive 2009/28/EC on the promotion of the use of energy from renewable sources.

9 Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources.

10 Commission delegated regulation (EU) 2019/807 of 13 March 2019 supplementing Directive (EU) 2018/2001 of the European Parliament and of the Council as regards the determination of high indirect land-use change-risk feedstock for which a significant expansion of the production area into land with high carbon stock is observed and the certification of low indirect land-use change-risk biofuels, bioliquids and biomass fuels.

11 The equation for significance includes modifying terms for high-productivity feedstocks, and reflecting that peatland conversion leads to more CO_2 emissions than forest conversion. These are both important for palm oil, but are not relevant for soy (which is considered to have 'normal' productivity and which is not normally planted in peat areas).

expansion onto high carbon stock areas. With soy being relatively close to the threshold, it remains possible that if the rules were to change, if deforestation rates increase before the feedstock expansion data is reviewed, or if new evidence shows that deforestation rates were previously underestimated, soy could be placed on the list of high ILUC-risk feedstocks.

1.2. Soy as a driver of forest loss

The underlying reason for a link between soy farming and deforestation is simple. Soy grows well in tropical climates, and many areas that are naturally forested are suitable for soy production (using varieties specifically developed for these areas over the last 60 years). Clearing high carbon stock and biodiversity rich tropical forest to make way for soy farming (sometimes with an intermediate period of cattle ranching) can be a good business model, even if the cost to nature and the climate is severe. Prior to 1970, there was very little soy farming in South America, but In the decades that followed development of new varieties and cultivation technologies coupled with buoyant international demand and strong government support have turned soy into one of South America's most important export crops (Kaimowitz & Smith, 2001). By 2017, the value of net exports of soybeans, soy meal and soy oil from South America had reached $50 billion according to FAOstat reporting. The main exporter nations are Brazil, Argentina and (to a lesser extent) Paraguay, as shown in Figure 7.

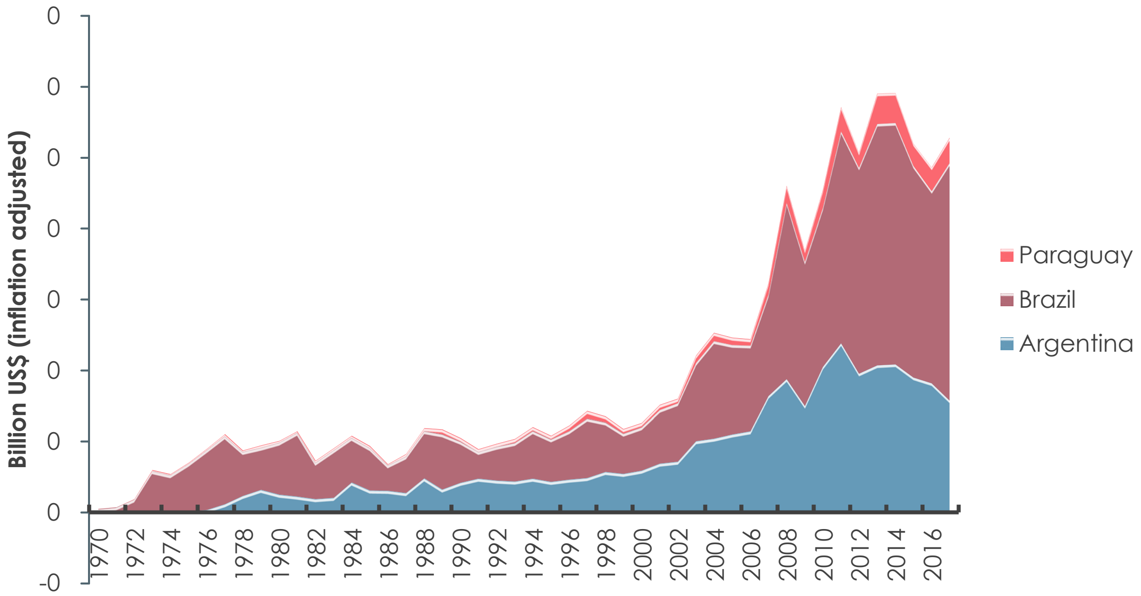

Figure 7. Export revenue from soy products (beans, meal, oil) 1970-2017 (inflation adjusted to 2020 dollars)

Source: FAOstat

Identifying the link between soy farming and deforestation is complicated by the fact that soy is often not planted directly after a deforestation event. It is common for deforestation to be followed by a period of cattle pasturing and then by one or two years of dry rice cultivation to prepare the land before soy is planted as a long-term arable crop (sometimes accompanied by a second-season corn crop), such that five years or longer could pass between deforestation and the establishment of the soy crop (Berkum & Bindraban, 2008). Conclusions about the extent to which soy should be considered a direct (or proximate) driver of deforestation can therefore be sensitive to the length of time that is considered following initial deforestation. These delays also complicate any attempt to identify correlation or causation between changes in soy demand or prices and specific episodes of deforestation.

Pendrill et al. (2019) estimates that in the period 2010-14, the expansion of oilseeds in South America (dominated by soybean expansion) was associated with 150 $MtCO_2e$ of deforestation emissions. Arima et al. (2014) provides evidence that demand for soy and beef has been a driver of historical Amazon deforestation by analysing the correlation between lagged soy and cattle prices and rates of deforestation in the period 1995 to 2007, finding that the combination of these prices could explain 75% of the variation in forest loss rates through this period. This relationship is seen to weaken after 2007, with the global food price spikes seen in 2007/08 and again in 2010-12 not reversing the downwards deforestation trend for the Amazon region. Arima et al. (2014) goes on to assess the roll of enhanced enforcement of the 2004 Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDAm)[12], in the period 2009-11. It concludes that the evidence supports a hypothesis that enhanced enforcement of deforestation prevention measures contributed to reduced levels of deforestation.

The historical relationship between soy and deforestation goes beyond the direct process of clearing forest and planting soybeans. For example, Kaimowitz & Smith (2001) argues that by offering a production model that required higher capital but less labour than other available crop choices, the growth of soybean production in existing agricultural areas in the south of Brazil resulted in job losses for rural labourers and the consolidation of small holder farms. This in turn created a pool of unemployed labour and the reduction in employment created an economic incentive to migrate to the agricultural frontier in search of jobs and opportunities. It is suggested that this migrating population displaced by the consolidation of industrial agriculture contributed to accelerated forest conversion (whether to grow crops or raise animals).

1.3. A note on deforestation data

It should be understood that estimates of annual deforestation rates can vary between sources depending on the input data considered, the spatial resolution of that data, forest definitions, handling of areas obscured by cloud cover, how temporary reductions in tree cover are handled and so forth. For instance, the estimate of 2019 Cerrado deforestation made by Azevedo et al. (2020) based on deforestation alerts from DETER[13] is only about 2/3 of the 2019 Cerrado deforestation that is recorded by INPE (2020) based on analysis with PRODES[14]. Part of the explanation in this case is that INPE (2020) extrapolate upwards to compensate for areas obscured by cloud cover in the satellite pictures used. As we will discuss at more length later in the report, there are also significant discrepancies between

12 http://redd.mma.gov.br/en/legal-and-public-policy-framework/ppcdam

13 Brazilian real time deforestation alert data.

14 Brazilian deforestation monitoring data based on a combination of satellite imagery and expert judgement.

deforestation rates reported by PRODES and by other datasets. Given these differences in data and data treatment caution should always be applied if considering making a direct comparison between numbers from different sources, years and regions. When considering classification of a biofuel feedstock as 'high ILUC-risk', the objective will be to find data that provides the best possible estimate of total replacement of high carbon stock ecosystems (as defined in RED II, i.e. wooded land with 10% or greater tree canopy cover) by the crop in question, but in the discursive sections of the report values are often quoted as given in the source material, which may use differing forest definitions. The reader may need to refer to those sources for additional context if interested in pursuing further reading or if seeking to understand apparent consistencies between different quoted values.

2. Making and breaking the link from soy to deforestation in the Amazon

Deforestation rates in the Amazon reached a peak of over 2.5 million hectares per year in 2004, driven primarily by expansion of livestock pasturing and soy farming. International campaigning led by Greenpeace led to the agreement in 2006 of a 'soy moratorium' whereby the soy industry committed to try to eliminate the conversion of forest land in the Amazon to soy farming. The moratorium was signed alongside Government action to increase enforcement of forest protection law, and its implementation coincided with significant reductions in deforestation. Some analysts have questioned the size of the contribution of the moratorium to reducing overall deforestation, and it is likely that some deforestation is going undetected, but it seems certain that soy expansion on forest land has been significantly reduced since the moratorium was agreed. It is likely that the moratorium has resulted in increased deforestation pressure in the cattle industry, in other parts of Brazil and South America more widely, but the evidence from studies of deforestation leakage seems consistent with a conclusion that the moratorium has contributed to net reductions in forest loss for South America as a whole. There is evidence of a more recent uptick in Amazon deforestation rates, which some have associated with a reduced focus on forest conservation by a new Brazilian administration, but it is not yet clear what role the soy industry may have played.

Between 1990 and 2015, 78 million hectares of native forest in the Brazilian Amazon was replaced with alternative land uses (Massoca et al., 2017). Most of this land was turned into extensive cattle pasture, but soybean agriculture has also become important in recent decades. Reviews of commodity linked deforestation routinely identify soy as a high deforestation risk crop (Malins, 2019).

As rates of soybean expansion in Brazil peaked in the early 2000s (cf. Figure 8), deforestation rates in the Amazon rainforest were also growing (Figure 9), causing a global outcry. Environmental non-governmental organisations (NGOs) led by Greenpeace identified soy production as a key driver of forest loss, and called upon major companies in the soy supply chain to commit not to buy soy from land cleared after 2006.[15]

15 Cf. https://www.greenpeace.org/usa/victories/amazon-rainforest-deforestation-soy-moratorium-success/

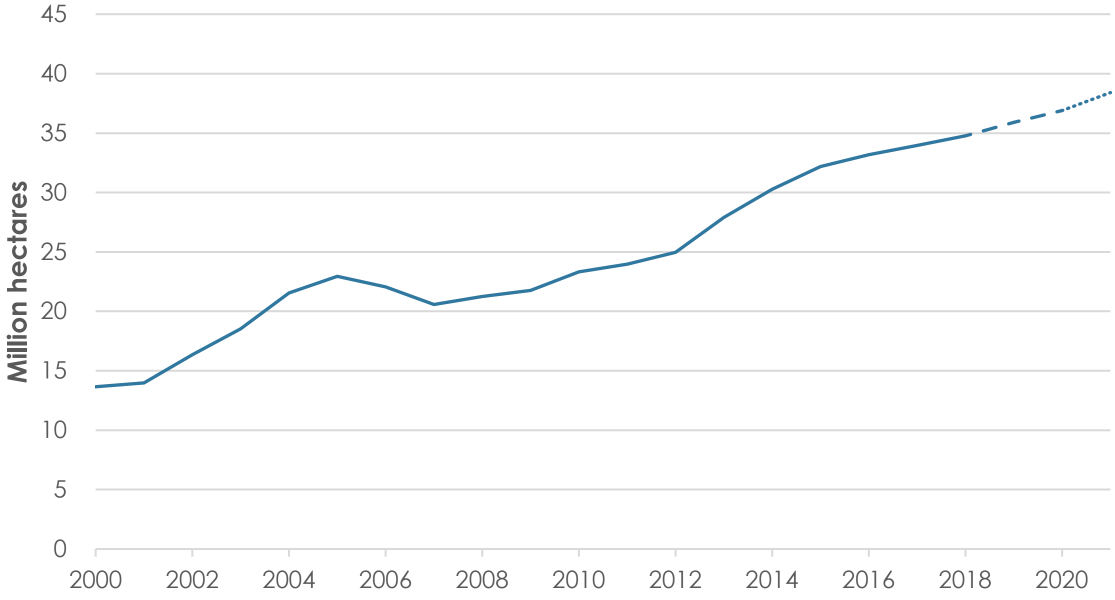

Figure 8. Harvested soy area in Brazil, 2000 to 2020

Source: 2000 to 2018 from FAOstat, 2019 and 2020 and projection for 2021 from Ustinova & Flake (2020)

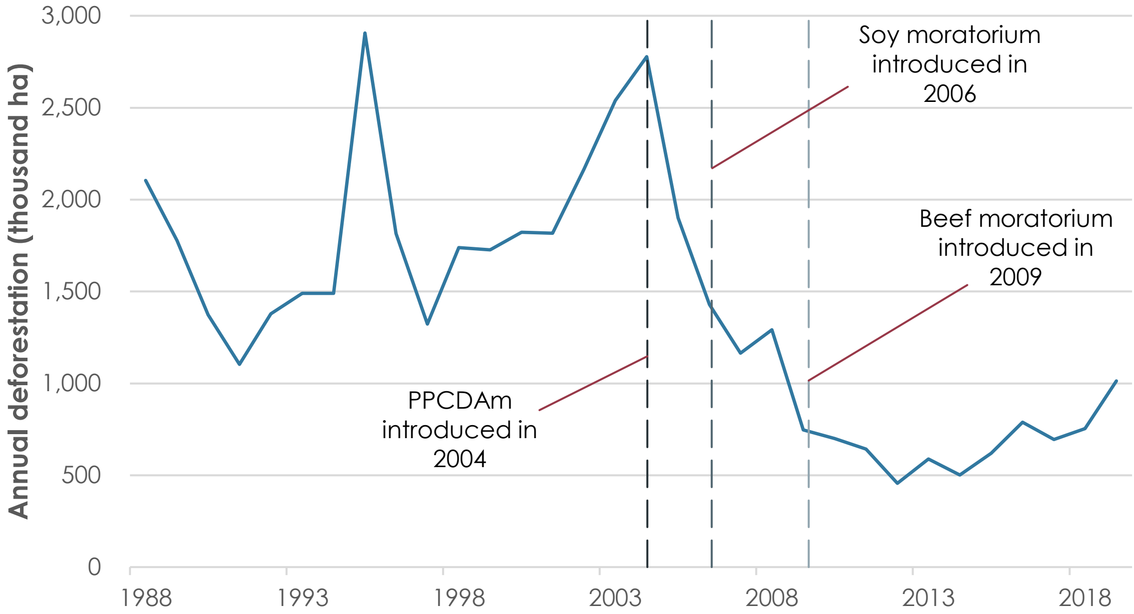

Figure 9. Estimated annual Amazon deforestation (1988-2019)

Source: Mongabay (2020), based on DETER and SAD

2.1. The Amazon soy-deforestation moratorium

The Amazon soy moratorium was signed in 2006 by the Brazilian Association of Cereal Exporters and the Brazilian Association of Vegetable Oil Industries, together representing 90% of all soy purchased in Brazil (Massoca et al., 2017), and monitoring and compliance are done by the Brazil Soybean Working Group. It committed soy producers not to clear additional forest area, and soy buyers not to purchase soy from deforested areas. After a series of extensions to the initial period, since 2016 it has been extended indefinitely.

The soy moratorium, coupled with contemporaneous government measures under PPCDAm, has been successful in terms of reducing the direct deforestation impact of soy expansion in the Amazon biome (Massoca et al., 2017). Whereas in the two years before the soy moratorium 30% of soy expansion in the Amazon biome was identified as being directly associated with deforestation (meaning that deforestation had occurred within three years prior to crop establishment), by 2014 this had reduced to 1% on one assessment (Gibbs et al., 2015). This was despite there being no reduction in overall rates of soybean area expansion in the Amazon in this period. Soy expansion had shifted from a roughly 30:70 mix of forestland and pastureland to almost exclusively targeting pastureland. Similarly, Macedo et al. (2012) assess changing deforestation dynamics in Mato Grosso (part of which lies within the Amazon biome), and conclude that in the period 2001 to 2005 26% of soy expansion occurred directly on forest, but that this reduced to 9% in the period 2005-2009.

Monitoring of the moratorium for the Soy Working Group reports only relatively small areas of non-compliance (Nassar et al., 2018), and non-compliant operators appear to be the exception rather than the norm. Silva Junior & Lima (2018) assessed transitions from forest to soy cropping in the Amazon parts of Mato Grosso state since implementation of the soy moratorium. From 2009 to 2016, 54 municipalities are identified as non-compliant with the moratorium, accounting for 60 thousand hectares of soy expansion. That is equivalent to about 5% of soy expansion in the Amazon part of Mato Grosso in that period.

Providing additional context for the reduction in soy related deforestation after 2004, Carvalho et al. (2019) notes that as well as implementation of the deforestation moratorium and new Forest Code the Brazilian Real experienced strengthened by more than 50% against the dollar in the period 2004-2007, which made agricultural exports less profitable. This is shown in Figure 10, which shows that the period in which the Real strengthened against the dollar coincided closely with the period of reduction in deforestation rate from 2004 to 2012. The Real weakened again from 2012 to 2019, in which period deforestation rates have crept up. The link between currency fluctuations and avoided deforestation is also explored by Richards (2012), which claims that a strengthening of the Real against the US Dollar after 2003 may have spared as much as 4 million hectares of Amazon land from deforestation (based on estimation of the elasticity of soy area to producer incentives). The weakening of the currency (and therefore increase in the profitability in local currency terms of export crops) may have contributed to the increased deforestation seen in the last few years.

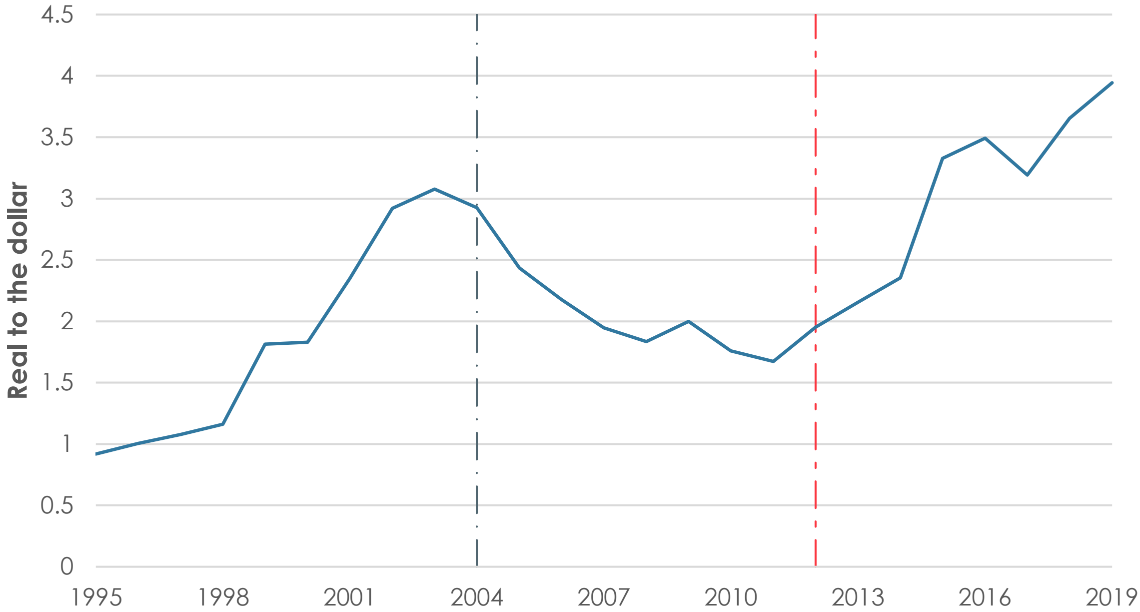

Figure 10. Exchange rate, Brazilian Real to U.S. Dollar (1995-2019)

Source: International Monetary Fund

Grey line at 2004 shows deforestation peak, red line at 2012 shows return to increasing reported deforestation.

Carvalho et al. (2019) also asks why rates of Amazon forest loss started to creep up again after 2012, discusses whether there are weaknesses in the soy and beef moratoria, and considers the potential impacts of the new political regime (a new presidential administration came to power on 1 January 2019). It is argued that anti-deforestation mechanisms including the soy moratorium have been 'subverted and bypassed', and that this explains the return to increasing deforestation rates since 2012. In the case of the soy moratorium leakage is emphasised – from the soy market into the cattle market, and from the Amazon to other regions (leakage is discussed further in the next section). While the concerns expressed are legitimate Carvalho et al. (2019) overstates the evidence base to support its concerns. In particular, while Richards et al. (2014) is quoted as evidence of the indirect link between conversion of pasture to soy farming and increased deforestation for cattle expansion, this paper actually shows a clear change from the period up to 2006 to the period from 2007 to 2011. After the soy moratorium was introduced there is a significant reduction in impact of crop expansion on deforestation, consistent with a hypothesis that the combination of the soy moratorium and other measures had been somewhat successful in weakening the link between soy expansion from even indirect deforestation.

Recognising that simultaneous changes in enforcement of the Forest Code and in exchange rates make it difficult to clearly identify the impact of the Amazon soy moratorium, Pede & Chibebe Nicolella (2020) set out to use the discontinuity in application of the moratorium at the edge of the Amazon biome as a way to analytically explore the effect on deforestation. By comparing deforestation data in Mato Grosso state (which is part of the legal Amazon but that straddles the Amazon and Cerrado biomes) this paper seeks to isolate the effect of the moratorium (PPCDAm applies to the whole legal Amazon). The paper concludes that the implementation of the moratorium did indeed contribute to reducing deforestation rates in the Amazon biome. It is unclear from this work whether higher forest cover loss rates after 2007 in the Cerrado are indicative of deforestation leakage from the Amazon. A similar approach is taken by Moffette & Gibbs (2021), which considers distance from the Amazon-Cerrado border in Mato Grosso as an explanatory variable. This study shows that soy expansion in Cerrado areas near the Amazon border accelerated after the introduction of the soy moratorium, which implies a spillover effect. Similarly, cattle population in Cerrado areas near the Amazon border increased after introduction of the zero-deforestation agreement for cattle. This study finds no statistically significant evidence of deforestation leakage due to the soy moratorium, but the zero-deforestation agreement for cattle was associated with a deforestation increase near the Amazon-Cerrado border.

Alongside leakage, another potential limitation in the soy moratorium is the possibility of 'soy laundering', whereby deforestation-associated soy could be sold as deforestation free by falsifying documentation. Rausch & Gibbs (2016) report that increasingly complex ownership and rental arrangements for soy farms in Mato-Grosso could create opportunities for false reporting, for instance in the case of mismatch between property ownership information recorded on the soy moratorium blacklist and actual land ownership. It is nevertheless highlighted by Rausch & Gibbs (2016) that monitoring of the soy moratorium has provided no evidence of largescale laundering.

2.2. Are deforesters avoiding detection?

Brazilian Government deforestation statistics show a considerable and persistent drop in deforestation rates since 2004 (Figure 9), but are they telling the whole story? An alternative explanation for reduced reported deforestation rates is provided by Richards et al. (2017). Since 2008, monitoring of deforestation under the PPCDAm is primarily undertaken using the PRODES (Program for the Estimation of Deforestation in the Brazilian Amazon) system. Richards et al. (2017) hypothesises that agents of deforestation have been able to take advantage of knowledge of the specifications and limitation of the PRODES system allowing significant areas of forest loss to escape detection. One piece of evidence is a discrepancy between Amazon deforestation rates as assessed by PRODES and deforestation rates reported in the Global Forest Change (GFC) dataset (Hansen et al., 2013). From 2002-06 it is reported that the deforestation rate reported by PRODES was similar to the tree cover loss identified in the GFC dataset. After 2006 deforestation rates recorded with PRODES fell rapidly to a quarter of previous levels but while the tree cover loss recorded by GFC also reduced it did not reduce by as much, such that from 2009-2013 GFC recorded twice as much deforestation as PRODES (see Figure 11). As noted in the introduction, it should not be surprising that forest cover loss assessments differ between tools. The magnitude of the discrepancy, however, and its emergence after PRODES became a regulatory tool suggest that there may be a bigger issue than analytical differences. Richards et al. (2017) also point to fire incidence data that tracks the GFC assessment more closely than the PRODES assessment. The three datasets are well correlated before 2008, but after 2008 the fire data and GFC data remained broadly consistent but became decoupled from the PRODES data.

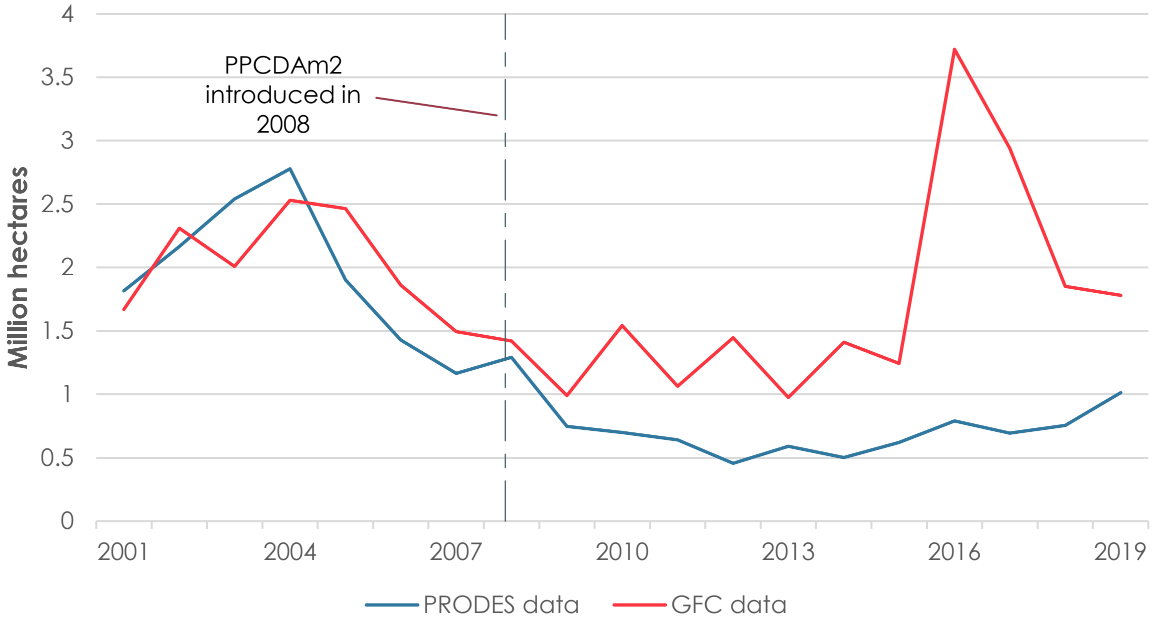

Figure 11. Comparison of PRODES deforestation data with tree cover loss (> 30% canopy cover) recorded in the GFC dataset

Source: PRODES and GFC. GFC data for tree cover loss with threshold set at > 30% canopy cover, assessed for Amazon biome (data accessed via polygon upload on www.globalforestwatch.org).

Note: 2016 was a record year for fire incidence which contributes to the spike in tree cover loss in the GFC data. Understory fires result in recorded tree cover loss in GFC, but are not recorded as deforestation in PRODES.

One potential explanation of differences between PRODES and GFC based deforestation estimates would be that the GFC assessment is undertaken at higher resolution. GFC identifies tree cover loss using 30m resolution Landsat data. PRODES on the other hand identifies only areas of deforestation that are 6.25 hectare or larger. It is possible that deforestation for agricultural expansion is purposefully being undertaken in smaller increments than PRODES identifies. Richards et al. (2017) directly consider this possibility by estimating the total area of tree cover loss in the GFC dataset occurring in units of less than 6.25 hectares, but do not find evidence that the sum of deforestation in these smaller units has increased over time. Another difference in approach is that GFC records all incidences of tree cover loss (including for instance harvesting of timber by clear cutting in managed plantations) whereas PRODES would exclude that type of tree cover loss. There is no obvious reason, however, for the discrepancy due to such differences in classification to suddenly grow after 2008.

A similar result is reported by Milodowski et al. (2017). This study compares deforestation estimates at two chosen sites from four satellite assessment tools – PRODES, GFC, FORMA (Forest Monitoring for Action) and a new assessment undertaken using higher resolution 5 metre resolution RapidEye[16] data. The sites were chosen to allow comparison of ability to detect a range of types of deforestation event, and to minimise the interference of cloud cover in the RapidEye data. This analysis highlights that there are significant discrepancies between the datasets both regarding identification of areas as forest in the baseline and

16 https://www.satimagingcorp.com/satellite-sensors/other-satellite-sensors/rapideye/

identification of subsequent forest loss. The RapidEye assessment found higher rates of forest loss than either GFC or PRODES, but was significantly closer to GFC. On one site, RapidEye analysis found 50% more deforestation than recorded by PRODES, on the other 15% more. Milodowski et al. (2017) also provides a characterisation of the cumulative forest loss at each site by size of area cleared, showing that a significant fraction of recorded deforestation has occurred at scales below the resolution of PRODES. The study suggests that even the higher resolution GFC may struggle to identify deforestation events smaller than two hectares – this may suggest an explanation of the failure by Richards et al. (2017) to demonstrate increases in the fraction of tree cover loss occurring in small increments.

The issues detailed by Milodowski et al. (2017) and Richards et al. (2017) are of great importance for understanding Amazon deforestation rates, but it is not clear what the full implication is for assessing the success of the soy moratorium in changing the role of soy expansion as a direct driver of deforestation. For example, the analysis in Gibbs et al. (2015) is based on PRODES data for identifying deforestation area, and thus if PRODES systematically understates forest loss then the result in Gibbs et al. (2015) for the estimated fraction of soy expanding onto cleared forest in the period after 2008 is likely to be an underestimate. Similarly, monitoring of the soy moratorium by the soy industry relies on PRODES data to identify deforested areas (Nassar et al., 2018). If the data discrepancy does indeed reflect active subversion of deforestation monitoring by the deforesting parties then it is at least possible that a disproportionate amount of this undetected deforestation is associated with soy expansion – avoiding detection of deforestation for soy would avoid both government intervention and exclusion from the soy supply chain. Richards et al. (2017) find that the largest discrepancies between the deforestation datasets are in regions with booming soy industries (Mato Grosso and Para).

2.3. Soy, cattle and forest loss on the frontier

One simple way to categorise episodes of deforestation is by identifying new land uses that replace lost forest – for instance cattle pasture, agriculture or urban expansion. This can sometimes be referred to as identifying the 'proximate causes' of deforestation (Geist & Lambin, 2002). The European Commission assessment of high ILUC-risk feedstock status (European Commission, 2019a) is based on this type of analysis, identifying areas where cropping follows within a few years[17] after deforestation (or peat drainage). There is always some uncertainty in this assessment – in general the more quickly crop establishment follows deforestation the more confident one can be that crop expansion was the cause of the deforestation event, but ignoring crop establishment taking place later could underestimate the real impact.

Simply identifying which land uses follow a deforestation event may not, however, provide a full picture of the underlying causes. It does not consider the value of timber extracted during deforestation, any relationships between agricultural systems or the role of infrastructure development. Geist & Lambin (2002) reviewed identified drivers of tropical deforestation (up to 1996) from 150 cases in the literature and concluded that "tropical deforestation is … best explained by multiple factors and drivers acting synergistically rather than by single-factor

17 The annexes to the Commission staff report rely on a number of assessments from the literature with different assessment periods and therefore there is not a single deforestation period considered in the Commission assessment.

causation". Assessing proximate causes of deforestation allows a greater degree of certainty than trying to assess underlying drivers – assessing what type of vegetation is currently grown on a given land area is a more tractable question than trying to assess what informed the decision to remove tree cover in the first place – but may miss important context that could affect our understanding of the success or failure of bioenergy policy.

Some analysts have suggested that soy expansion acts not only directly as a driver of deforestation, but indirectly by displacing cattle ranching. For example, Margulis (2003) argued that, "soybeans have been expanding rapidly in the Cerrado, pushing the expansion of the agricultural frontier into forest regions", and Rausch & Gibbs (2016) observes that, "it is possible that the soy sector's private governance could result in displacement of activities besides soy into areas that do not meet the traders' criteria, potentially resulting in deforestation in any case." Similarly, Barona et al. (2010) conclude that, "in Mato Grosso, an increase in soybeans occurred in regions previously used for pasture, which may have displaced pastures further north into the forested areas, causing indirect deforestation there. Therefore, soybean cultivation may still be one of the major underlying causes of deforestation in the Legal Amazon."

Gasparri & le Polain de Waroux (2015) discusses explicitly the development of a coupling between the soy and cattle sectors in South America, and argues that soy- and cattle-induced deforestation should be viewed as part of a single regional process. Conversion of pasture to cropland implies capital transfers from cropping to livestock farming, and this is proposed as a potential vector of land use change 'amplification', whereby the profitability of soy cropping makes accelerated conversion of forest to pasture possible. It is suggested that, given coupled industries, policies and voluntary interventions that target economic operators in only a single location (e.g. the Amazon) or single mode of production (e.g. soy farming) are likely to be subject to leakage.

Arima et al. (2011) aims to statistically demonstrate the association between soy expansion onto former pasture areas and pasture expansion on the forest frontier of the Brazilian Amazon. Three statistical models are presented, suggesting that for every hectare of soy expansion onto former pastureland in more settled areas of the Southeast Amazon there may have been between 1 and 7 hectares of indirect deforestation in areas further towards the forest frontier in the Northwest Amazon. This result implies that the deforestation impact of soy expansion by pasture displacement could potentially be greater (at least based on analysis of the period 2001-08) than the impact when soy expands directly on forestland. This is a strong result, which would suggest that the appreciation of land values in more settled regions where soy expansion is now concentrated is such that soy expansion is financing growth in the cattle industry at the expense of the forest. The range in those estimates is indicative of the difficulty in firmly identifying indirect effects from agricultural land use change (analogous to the inevitable uncertainty in modelling ILUC), and it is not clear whether those dynamics are still operative ten years later (in particular as the beef deforestation moratorium was signed only in 2009), but it is a reminder that focusing only on proximate deforestation drivers may not capture the full picture.

While the soy industry may be an indirect driver of deforestation associated with cattle ranching, the example of the soy moratorium provided the inspiration for the Amazon beef deforestation moratorium, which was signed in 2009. The beef moratorium applies to a much larger agricultural area (there is still ten times more area devoted to cattle than to soybeans in the Amazon biome) but is less stringent, in that it forbids only illegal deforestation whereas the soy moratorium prohibits any proximate deforestation. The impact of the beef moratorium is less clear than that of the soy moratorium – partly because it is difficult to unpick the impact of the moratorium on illegal deforestation from the impact of improved enforcement of the Forest Code, and partly because there is a possibility of cattle 'laundering' allowing cattle from non-compliant ranches to enter the supply chain by misidentifying the farm of origin (Massoca et al., 2017). For example, a recent investigation by Amnesty international (2020) found evidence of laundered beef entering the supply chain of the world's largest beef company, JBS. Still, Gibbs et al. (2016) shows that slaughterhouses committed to the moratorium have avoided purchasing deforestation-associated cattle, and that ranchers supplying these slaughterhouses have been more likely to register their property on a public environmental registry.

2.4. Has soy expansion been displaced from the Amazon to other high carbon-stock areas?

Just as the climate benefits of reducing soy expansion on recently deforested land in the Amazon could be undermined if there is a 'leakage' of deforestation into the livestock industry, they could also be undermined if soy expansion is displaced to other areas. In Brazil, the other main area of soy-related deforestation is the Cerrado, while outside Brazil the Gran Chaco spanning Bolivia, Paraguay and Argentina has become a focus of deforestation concern. Leakage could take the form of 'activity leakage' or 'market leakage' (le Polain de Waroux et al., 2017). Activity leakage refers to the case that capital is displaced away from the regulated area when the cost of production is increased by regulation (e.g. through increasing prices for land where production is not prohibited by the regulation) and used to invest in land clearance in other less regulated areas where returns may be better. Market leakage refers to the case that suppression of production in the regulated area causes marginal increases in affected commodity prices, creating an incentive for additional expansion in other areas.

The Cerrado is a tropical savanna region in Brazil sprawling across a dozen states. The Brazilian Cerrado is very biodiversity rich[18], and supports a range of vegetation types. It can be divided into forest (cerradão), shrubland *(sensu stricto*) and grassland (*campo limpo* and *campo sujo*) areas. The forest areas have high levels of canopy cover (over 50%) but much of the Cerrado shrubland also qualifies as a high carbon stock area under the RED (this is discussed in more detail in section 3.2.2). Azevedo et al. (2020) note that, "The Amazon and Cerrado biomes together comprise 96% of all the alerts and 96.7% of the total [Brazilian] deforested area in 2019," making the Cerrado biome of key interest in considering the possibility of leakage of deforestation from the Amazon. Lima et al. (2019) argue that while the Amazon soy moratorium has successfully prevented further direct deforestation for soy expansion within the legal Amazon, the new deforestation frontier is in the Cerrado, in the MATOPIBA region (Maranhão, Tocantins, Piauí and Bahia). MATOPIBA contains the largest remaining continuous areas of the Cerrado. The Brazilian Forest Code requires less preservation of native vegetation in the Cerrado biome than in the Amazon – in the legal Amazon (the nine states[19] that span the Amazon basin) landowners are allowed to convert only 20% by area of native

18 Cf. https://wwf.panda.org/knowledge_hub/where_we_work/cerrado/

19 Acre, Pará, Amazonas, Roraima, Rondônia, Mato Grosso, Amapá and Tocantins as well as the region west of longitude 44° W in the state of Maranhão.

vegetation in their land holdings to productive use but up to 65% of shrubland and 80% of grassland, while outside the legal Amazon 80% of native vegetation on any holding may be converted regardless of vegetation type (Machado & Anderson, 2015).

Data for the Amazon and Cerrado from the PRODES deforestation monitoring system (INPE, 2020) suggests that reductions in Amazon deforestation have been achieved without strong leakage into the Brazilian Cerrado. Figure 12 shows that the reported reductions in Amazon deforestation rates after 2004 were paralleled by reductions in Cerrado deforestation (of course, if Richards et al. (2017) is correct in claiming that real deforestation rates have been higher than suggested by PRODES data, this could also explain the absence of an obvious leakage effect).

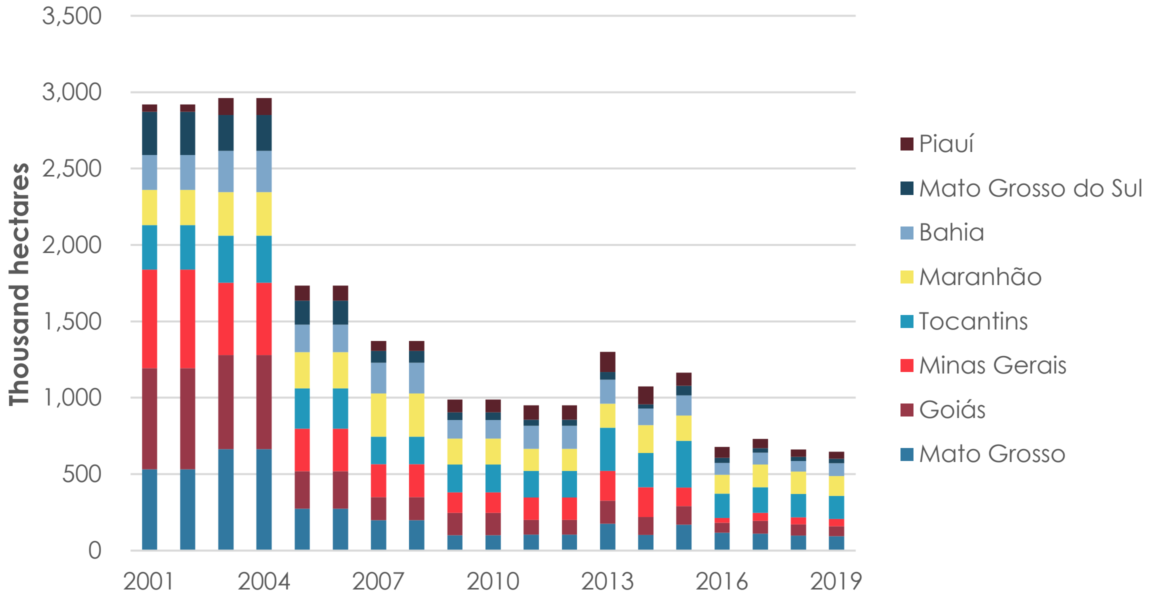

Figure 12. Reported annual deforestation in the Cerrado (2001-2019)

Source: INPE (2020)

Noojipady et al. (2017) adds to this picture by providing estimates of the area of Cerrado deforestation in the period 2003-13 that was associated with cropland expansion. As seen in Figure 13, Amazon deforestation for cropland expansion is found to have declined by a factor five in that period, but Cerrado deforestation for cropland was relatively steady. While there is no obvious uptick in Cerrado forest lost to match the considerable reduction in Amazon deforestation in the period, given that overall rates of Cerrado deforestation were also falling this does suggest that cropland was taking a larger role as a proximate driver of deforestation in the Cerrado. Comparing the Noojipady et al. (2017) results to overall deforestation numbers from INPE (2020)[20], cropland goes from being a proximate cause for 7% of deforestation on average in the three years 2003-2005 to being a proximate cause for 16% of deforestation in the three years 2011-2013.

20 Remembering that caution should be used in comparing results from different datasets, and therefore that the percentages we calculate here should be treated as indicative rather than precise.

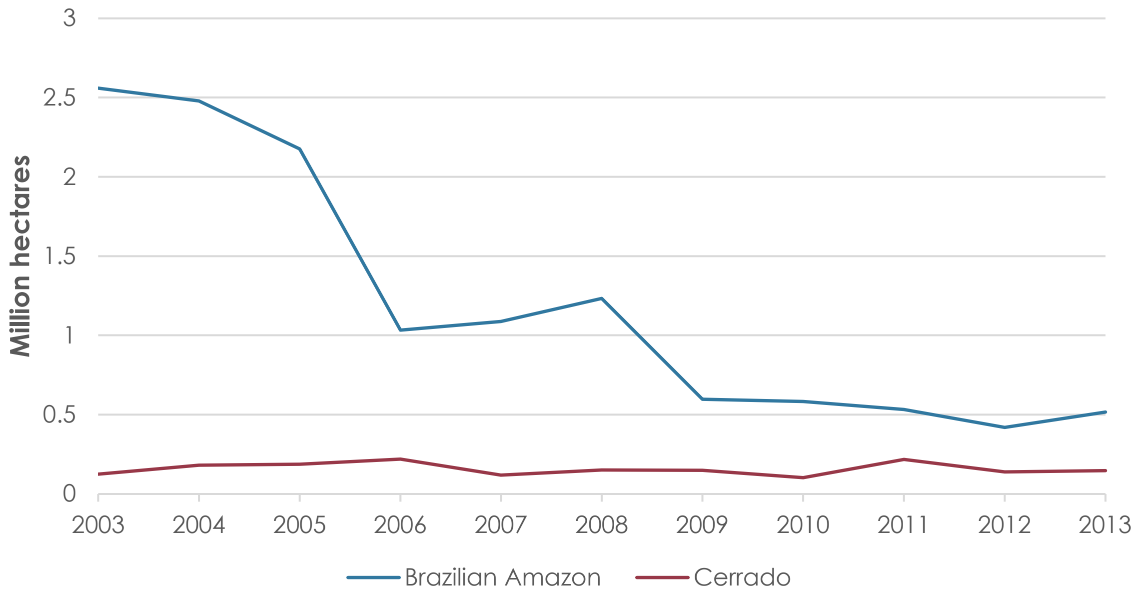

Figure 13. Annual forest lost for cropland expansion in the Brazilian Amazon and Cerrado

Source: Noojipady et al. (2017)

Malins (2019) showed that nearly half of soy planted area expansion in Brazil in the period 2013-18 had occurred in the Cerrado biome. Since then, Brazilian agricultural statistics (IBGE, 2020) show that the importance of the Cerrado as a location for soy expansion has increased further (Figure 14). As the Cerrado is believed to have a higher rate of deforestation for soy expansion than any other biome in Brazil (European Commission, 2019a; Malins, 2019) this could be consistent with a marginal increase in the fraction of soy expansion associated with deforestation.

While the Cerrado has accounted for a relatively high fraction of total Brazilian soy planted area expansion in recent years, total annual soy area growth has fallen since 2013, as shown in Figure 15, and while annual expansion has increased again in the years since the current administration came to power we are certainly not yet seeing unprecedented rates of new planting – although of course given the normal lag between forest clearance and soy establishment we would not yet expect to see evidence in the agricultural statistics of planting on land cleared in 2019.

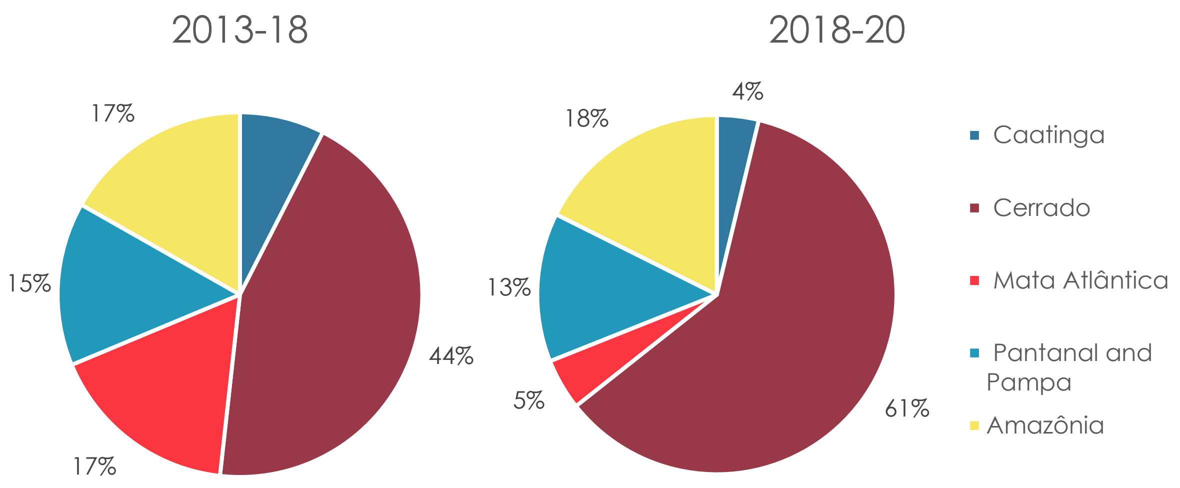

Figure 14. Soy area expansion in Brazil by biome

Source: own calculation on data from IBGE (2020). As in Malins (2019), data is grouped at the state level into the most characteristics biomes, with the exception of Mato Grosso for which one third of expansion is categorised as Amazon and two thirds Cerrado based on Macedo et al. (2012) and Maranhão for which one third of expansion is categorised as Amazon and two thirds Cerrado based on relative area in each biome.

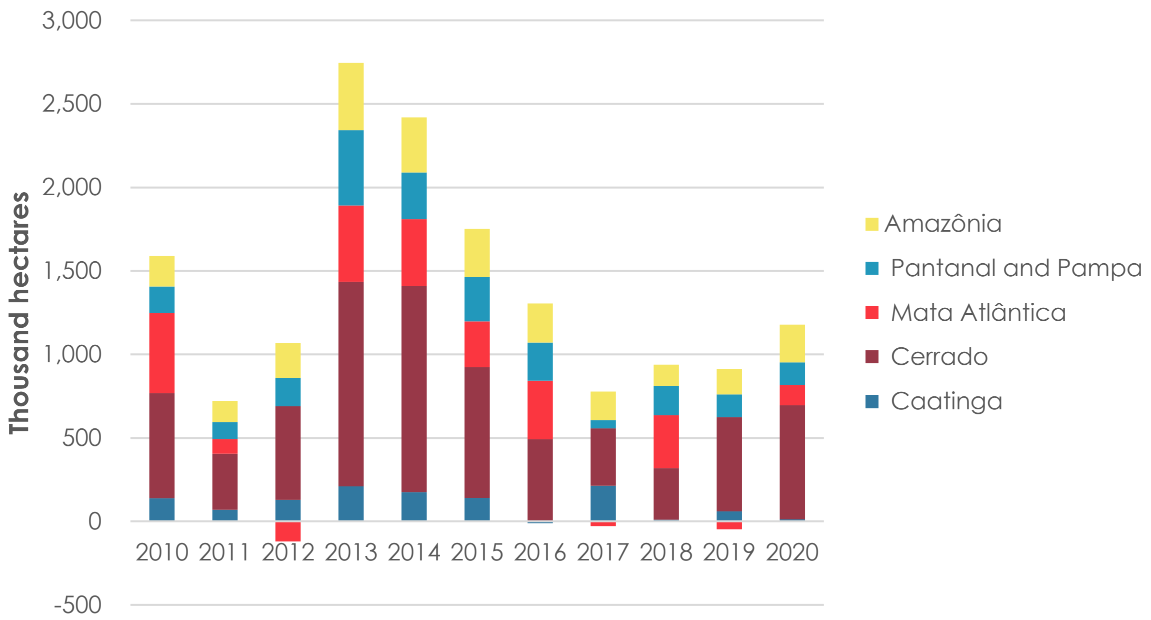

Figure 15. Annual soy area change by Brazilian biome, 2010-20

Source: own calculation on data from IBGE (2020), see note on Figure 14.

It is not only additional expansion in other Brazilian biomes that is of interest – there could also be leakage to the other major soy producers in other countries in the region. Curtis et al. (2018) shows that reductions in commodity-related forest loss in Brazil, and in particular the Amazon, after 2004 were partially offset by increasing deforestation rates in other parts of South America (Figure 4).

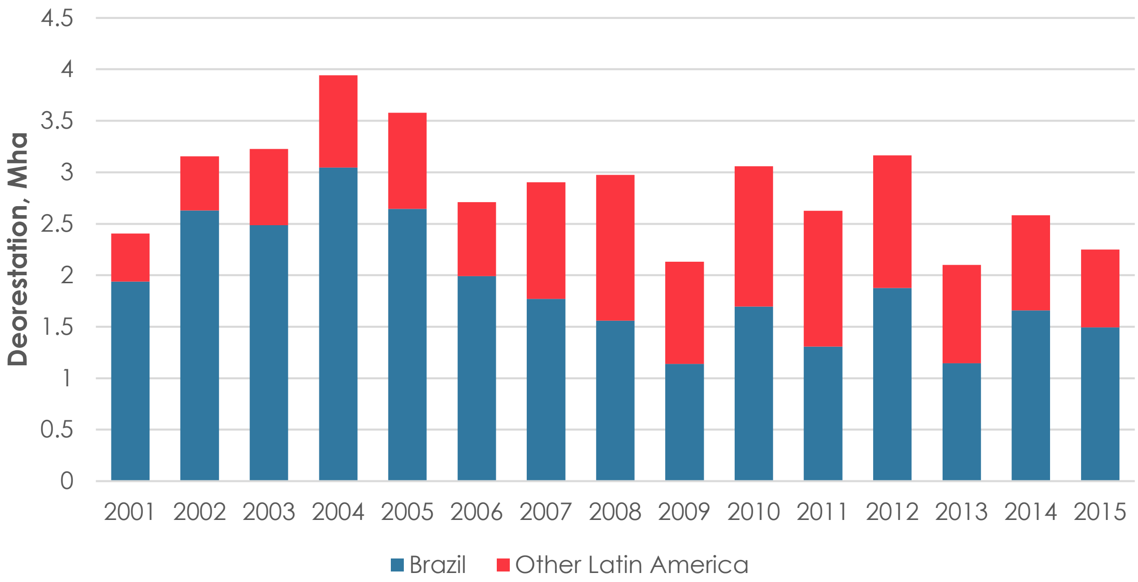

Figure 16. Commodity associated forest loss in South America

Source: Curtis et al. (2018)

le Polain de Waroux et al. (2017) seeks to identify whether restrictions on deforestation for soy and beef, especially in the Brazilian Amazon, have been accompanied by deforestation leakage to other areas of South America. The study presents econometric analysis that finds that there was a significant correlation between rates of soy expansion and rates of deforestation in most regions considered in the period 2001-2006, including all regions of Brazil, but that for the period after the soy moratorium (2007-2013) these correlations generally reduced and did not meet the standard for significance. There was no statistical evidence of increased regional regulation and enforcement reducing rates of soy expansion, suggesting that there has not been largescale leakage of soy expansion to other regions (i.e. that soy area was able to expand in more regulated areas despite restrictions on deforestation). Similarly, there was no major reductions in overall soy exports from countries with increased regulation (the study found that increased exports to Europe from more regulated areas largely compensated for reduced exports to other countries).

The study did however find a correlation between increased regulation of cattle in the Brazilian Amazon and reduced rates of pasture area expansion, and found that beef exports were reduced from countries with increased regulation. These results could be consistent with a hypothesis that the conversion of pasture to soy production in the Amazon contributed to increased deforestation for pasture expansion elsewhere, but this is not established by the analysis.

2.5. Is soy production a driver of recent (post-2018) increases in Amazon deforestation?

Data from the Brazilian National Institute for Space Research PRODES system shows that annual deforestation rates in the Brazilian legal Amazon, which trended down from 2004 to 2012, have started to rise again in the second half of the decade, and that 2019 saw the largest annual increase in deforestation rate since before 2004 (Figure 17). According to the PRODES data, the annual deforestation rate increased as much from 2018 to 2019 as in the whole period from 2014 to 2018. Real time monitoring data from the DETER tool shows that a further increase in deforestation rate may be expected from 2019 to 2020[21]. Data from GFC also shows an increase in tree cover loss in the last five years (Figure 11) but as discussed below this increase is identified earlier (2016) in the GFC data.

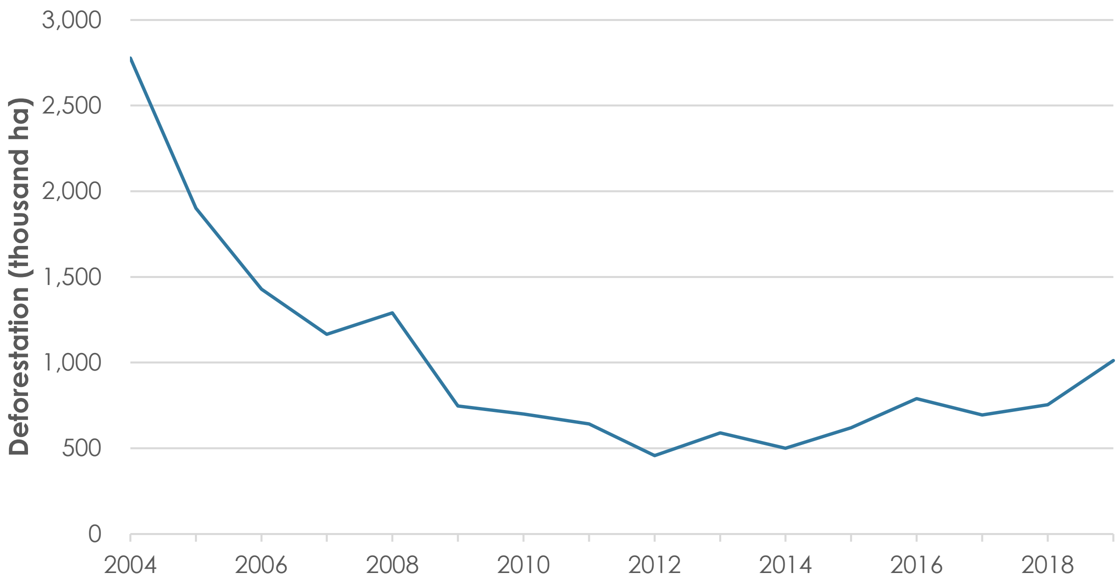

Figure 17. Annual deforestation in the Brazilian legal Amazon

Source: PRODES data (INPE, 2000)

The uptick in deforestation rates has been widely associated by commentators with "anti-environmental"[22] policies pursued by the administration of President Jair Bolsonaro, but it is less clear which mix of economic actors have been taking advantage of reduced enforcement of the Forest Code to increase deforestation. Data from Azevedo et al. (2020) based on non-PRODES data identifies that Amazon deforestation in 2019 was concentrated in three states: Amazonas, Pará and Rondônia. Of these, both Pará and Rondônia are recorded with significant expansion of soy area in recent years in Brazilian agricultural statistics from the SIDRA dataset (IBGE, 2020) (Table 1).

21 https://news.mongabay.com/2020/08/as-amazon-tree-loss-worsens-political-pressure-grows-and-brazil-hedges-critics/

22 Ibid.

Table 1. Recent soy planted area increase compared to deforestation rate in the wholly Amazon states

State	Total soy area, 2020	Increase in soy area, 2017-20	Deforestation in 2019
Amazônia total	1,023,604	209,047	630,307
Acre	1,651	1,513	57,891
Amapá	20,052	1,881	1,487
Amazonas	-	-	125,881
Pará	596,547	102,374	298,540
Rondônia	380,884	103,279	122,507
Roraima	24,470	-	24,001

While the PRODES data suggest an uptick in deforestation activity since 2018, the GFC data is complicated by the spike in tree cover loss associated with a rise in fire incidence in 2016 (Figure 11, Figure 18)[23]. GFC actually shows year-on-year reductions in tree cover loss since 2016. Fire is often associated with agricultural expansion, but when fires burn out of control the area affected does not necessarily represent a land use change.

Increases in recorded rates of deforestation in Brazil in the last few years were expected as a more agribusiness-friendly and less environmentally-concerned administration came to power. As well as 'real' increases in annual deforestation rates, if the new administration has created an expectation of relaxation in enforcement of the forest code the slight convergence of the PRODES and GFC datasets could reflect a feeling among agents of deforestation that it is now less important to evade detection. At present, there is not analysis available that would allow a confident statement either that the link between soy expansion and deforestation has increased since 2018 or that it is unchanged. Given that there are often several years between a deforestation event and the establishment of a soy crop, it would be difficult to identify the role of soy as a proximate driver of deforestation in 2018 and 2019 for several years. It will not be until at least 2024 that we could have access to analysis considering the subsequent five years of land use.

23 Cf. https://blog.globalforestwatch.org/data-and-research/what-to-know-about-2019-tree-cover-loss/

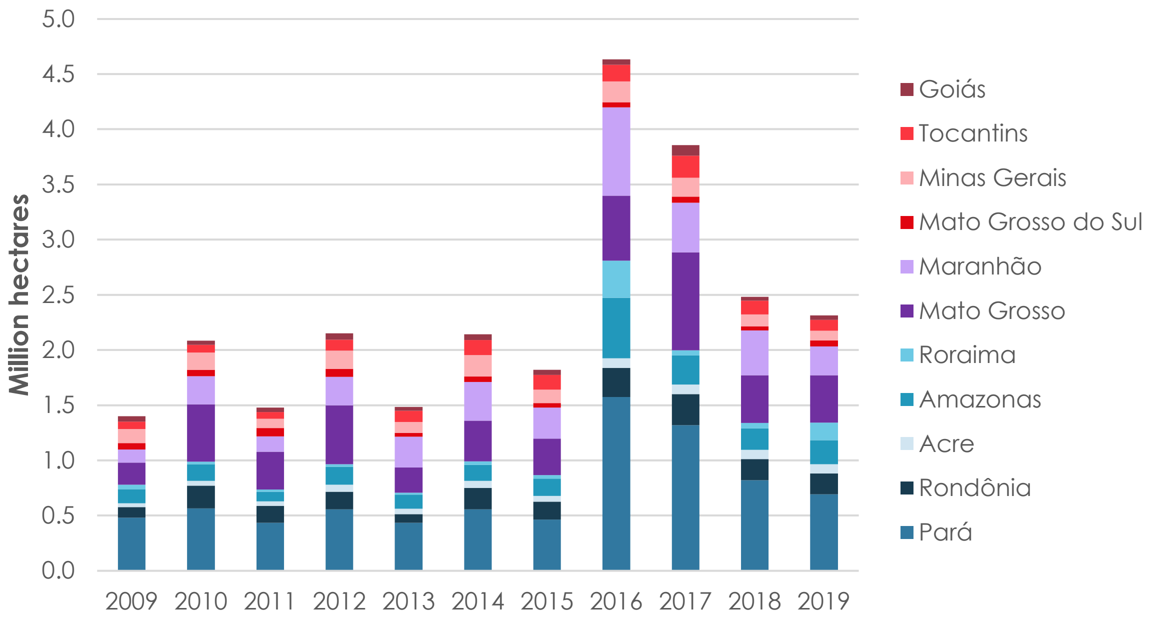

Figure 18. GFC reported tree cover loss (> 30% tree cover) for Amazon and Cerrado states from 2009 to 2019

Source: GFC data via www.globalforestwatch.org

Note: primarily Amazonian states in blue, primarily Cerrado states in red, mixed states in purple.

3. Quantifying the impact on carbon-rich ecosystems

The EU has set a threshold that if more than 10% of expansion of a given biofuel feedstock crop occurs at the expense of high carbon stock areas then that feedstock will be identified as high ILUC-risk. Soy expansion has been associated with clearance of wooded land across South America, in particular in the Brazilian Amazon and Cerrado and in the Gran Chaco, which spans Argentina, Paraguay and Bolivia. The most recent evidence from satellite assessment suggests that while the impact of soy expansion on deforestation has reduce in the Amazon since 2006, a significant impact continues in the other biomes, in particular the Cerrado. Outside of South America, the risk of soy-associated deforestation is in general much lower. Reviewing the evidence on the deforestation role of soy, we find that the fraction of global soy expansion occurring in the Cerrado and the fraction of expansion in the Cerrado occurring on forest land may both be higher than estimated in the European Commission's initial assessment of high ILUC-risk feedstocks. If those findings are confirmed by ongoing new analysis for the Commission, then soy oil would be moved into the high ILUC-risk feedstock category.

The delegated act on high and low ILUC-risk biofuels defines high carbon stock land as land meeting one of three definitions set in the recast Renewable Energy Directive: continuously forested areas with a tree canopy cover (or potential canopy cover from trees already in situ) of greater than 30%; areas with a tree canopy cover (or potential canopy cover from trees already in situ) of 10- 30%; and peatland areas. In this section, building on the analysis presented in European Commission (2019a) and Malins (2019), we present an update estimate of the fraction of soy expansion likely to occur on recently cleared land meeting these definitions, as well as discussing potential areas of expansion with high carbon stocks that would not be covered by the definitions.

3.1. Carbon loss from land clearing

When land is cleared for agricultural production, the primary carbon stock changes occur in aboveground biomass (the visible parts of trees and plants), belowground biomass (mainly roots) and soil. For forest systems, the largest carbon stock losses are likely to be from aboveground biomass, but for ecosystems such as grasslands the loss of soil carbon may well be much larger than even the combined loss of biomass carbon. The RED definition of high carbon stock land is based on soil type (peatland is defined as high carbon stock) and tree cover (land with > 10% canopy cover is defined as high carbon stock). European Commission (2019c) report that the expected average carbon stock loss from biofuel feedstock production on identified high carbon stock areas is 107 tonnes per hectare. If the resultant CO_2 emissions were allocated to soy biodiesel production on an energy basis[24] with the average yields given in BioGrace (2017) and assuming a reportable GHG emission reduction of 60% excluding land use change emissions, it would take 130 years of biofuel use to pay off this carbon debt and start to deliver a net CO_2 emission reduction.

24 This means that only about a third of the land use change emissions are allocated to the soy oil used to produce biodiesel, with two thirds being allocated to the soy meal co-product.

We will not consider peatlands here – while the carbon stocks in peatland are enormous, it would be very unusual for soy to be grown in peat soils (peatland is however very relevant to palm oil production, as discussed in Malins (2020)). It is useful, however, to briefly review the potential carbon costs of clearing woodland for soy cropping.

A range of estimates of carbon stock losses following land use change for cropping are provided in the 'Agro-Ecological Zone emission factor model' (AEZ-EF) (Plevin et al., 2014) which was developed for land use change analysis by the California Air Resources Board. For tropical forest conversion in the Amazon or Cerrado (generally AEZs 5 and 6), this model estimates typical carbon losses[25] of 230-250 tonnes per hectare. For forest conversion in the continental United States (AEZs 7 to 12), expected carbon losses would be lower, in the range 120 to 140 tonnes per hectare while on the Argentinian Gran Chaco losses of 95 to 150 tonnes per hectare are modelled. In some temperate regions the expected carbon loss per hectare of land converted is less than 100 tonnes per hectare. In reality, of course, there is considerable variation in per hectare carbon stocks even between areas with comparable agro-ecological conditions.

These low-end values for carbon loss on forest conversion can overlap the highest estimates for carbon loss after pasture[26] conversion, largely because of soil organic carbon loss. The RED implicitly ignores high levels of soil carbon storage in anything other than peat soils when making the high ILUC-risk assessment, and will thereby in at least some instances overlook land use changes with associated carbon dioxide emissions comparable with some forms of forest clearance.

3.2. Soy expansion in South America

There are five countries in South America identified by FAOstat as having a significant area of soybeans (more than 1 million hectares each) – Brazil, Argentina, Paraguay, Bolivia and Uruguay. All of these countries apart from Uruguay have lost millions of hectares of forest area in the past two decades – FAOstat reports over 40 million hectares of combined forest loss in the period 2000-2015 based on national reporting, while Hansen et al. (2013) reports a similar 39 million hectares of tree cover loss in the period 2000 to 2012 based on LandSat satellite data. Figure 19 shows areas of deforestation in South America in the period 2015-18 identified as commodity driven in GFC data, showing deforestation largely occurred within the Amazon, Cerrado and Chaco biomes.

25 This means that only about a third of the land use change emissions are allocated to the soy oil used to produce biodiesel, with two thirds being allocated to the soy meal co-product.

26 AEZ-EF includes only pasture, and does not explicitly consider native grasslands.

Figure 19. Commodity driven tree cover loss 2015-2018 (yellow), with location of the Amazon, Cerrado and humid and dry Chaco biomes

Source Global Forest Change dataset 1.7 via https://www.globalforestwatch.org/ *(Hansen et al., 2013)*

As noted in the introduction (section 1.3) soy is routinely identified as the main crop driving deforestation in South America. Aide et al. (2013) provides a characterisation of the regional variation in deforestation in South America in the period 2000-10. Concentrations of deforestation activity are apparent on the Amazon frontier in Brazil, in the Cerrado, in the Chaco region spanning Argentina, Paraguay and Bolivia, in the Atlantic forest in Eastern Paraguay and in the Uruguayan savanna. Graesser et al. (2015) provides a further characterisation of the extent to which deforestation activity in each country and in different biomes is associated with pasture expansion or cropland expansion, analysing land use changes in the period 2001-2013. Deforested land is identified as the source of over two millions of hectares of new cropland in the Brazilian Cerrado, over a million in the Atlantic forest in Paraguay and Brazil, nearly a million in the dry Chaco in Argentina, Paraguay and Bolivia, and 150 thousand hectares in the Southeast Amazon. These are the areas of most interest in assessing the deforestation impact of soy expansion. On the other hand, only limited expansion of cropland on deforested area is reported for the rest of the Amazon, the humid Chaco, the pampas and the Uruguayan savanna.

3.2.i) Amazon

As discussed above, the Amazon has previously been a major focus of environmental concern regarding the role of soy as both a proximate and indirect driver of deforestation, but since the adoption of the soy moratorium it is generally agreed that there has been a very significant reduction in the extent to which new soy area is directly replacing forest. European Commission (2019a) uses Gibbs et al. (2015) to estimate the fraction of soy expansion that replaces high carbon stock land in the Amazon, though noting that according to evidence from Richards et al. (2017) the real rate could be higher. European Commission (2019a) quotes 2.2% as the average expansion of soy onto previously high carbon stock land reported by Gibbs et al. (2015) for the period 2009-2013. Gibbs et al. (2015) considers only cases where soy was established within three years of deforestation – as it is reported that often soy establishment can take longer than this, Gibbs et al. (2015) may underestimate the true role of soy as a proximate driver of deforestation in this period.[27]

27 This applies equally to the Gibbs et al. (2015) estimates for Cerrado deforestation rates discussed in the next section.

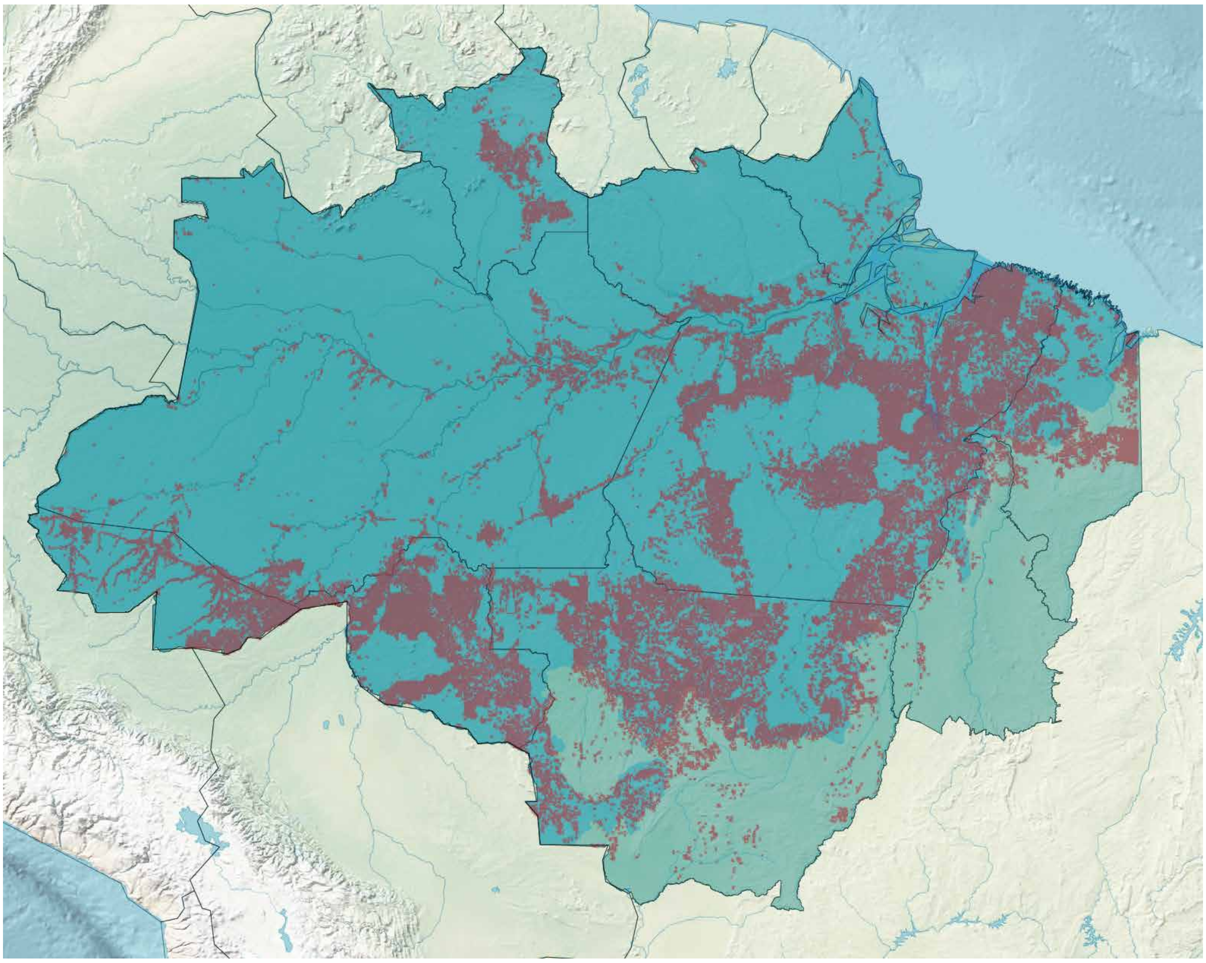

Figure 20. Deforestation in the Amazon 2000-2015 as identified by the PRODES system

Source: INPE data accessed through Global Forest Watch. www.globalforestwatch.org

We are not aware of better sources to use to assess the rate of soy expansion onto high carbon stock areas in the Amazon, and therefore here we also base our assessment on Gibbs et al. (2015). We note that even in the period before 2008, Richards et al. (2017) reports that the tree cover loss estimates from GFC tended to be slightly higher than the PRODES deforestation values (on average by 6% in the period 2002-2007). For this report, we therefore adjusted the assumed fraction of soy expansion on high carbon stock land in the Amazon to reflect assumed under-reporting of deforestation in PRODES data after 2008. Richards et al. (2017) suggested that real deforestation rates could be as much as 100% higher than those reported with PRODES, but analysing two sample plots (Milodowski et al., 2017) estimated underreporting at 15% to 50%. Here, we take the middle of the three values and apply a 50% upwards adjustment on the Gibbs et al. (2015) analysis. This gives a final assumption that 3.5% of Amazon soy expansion is on high carbon stock land.

A few points should be noted about this value. Firstly, Gibbs et al. (2015) allows only three years for soy establishment after deforestation if soy is to be counted as a proximate deforestation driver. This may understate the deforestation link given that other studies suggest typical periods of five years or more between deforestation and soy establishment. As an example, Gibbs et al. (2015) reports that taking a six year instead of three year period for proximate deforestation would double the assessed amount of soy planted in 2012 on deforested land, Secondly, as discussed above it is possible that the recent uptick in Amazon deforestation rates includes an increased role for soy. No adjustment is attempted to reflect this possibility, as there is simply not enough data to draw any firm conclusion. Thirdly, the identification of soy area in Gibbs et al. (2015) focused on three Amazon states (Mato Grosso, Pará and Rondônia). These represent most Amazon biome soy cropping, but this does mean that the analysis is not comprehensive of the whole Amazon region. Finally, it should be understood that the adjustment to the Gibbs et al. (2015) results based on Richards et al. (2017) and Milodowski et al. (2017) is subject to considerable uncertainty, as it is not known whether deforestation increments that are structured to avoid detection by PRODES are more or less likely to be associated with soy production.

3.2.ii) Cerrado

As was seen in Figure 15, the Cerrado has been the main area of soy expansion in Brazil over the previous ten years. The Cerrado is often referred to as a savanna landscape, but much of it is wooded and falls under the high carbon stock land definition of the RED II. De Miranda et al. (2014) provides a meta-analysis of studies of biomass distribution in the Brazilian Cerrado. Three land categories are considered – forest (*cerradão*), shrubland *(sensu stricto* with subtypes sparse, typical and dense) and grassland (*campo limpo* and *campo sujo*). The *cerradão* forest areas have high levels of canopy cover, but much of what is referred to as shrubland would also meet the Commission definition for high carbon stock land – for example, Felfili & Da Silva (1993) characterises Cerrado *sensu stricto,* identified with shrubland by De Miranda et al. (2014), as having tree cover from 10-60%, while Gibbs et al. (2015) notes that 65% of Cerrado vegetation meets the Brazilian forest definition (10% canopy cover or more). Average biomass stocks[28] identified by De Miranda et al. (2014) ranged from 97 tonnes per hectare for forest to 58 tonnes per hectare to shrubland to 18 tonnes per hectare for grassland. This suggests that in general the Commission definition of high carbon stock land is appropriate for identifying high carbon stock areas in the Cerrado.

28 Note that carbon constitutes only part of the biomass.

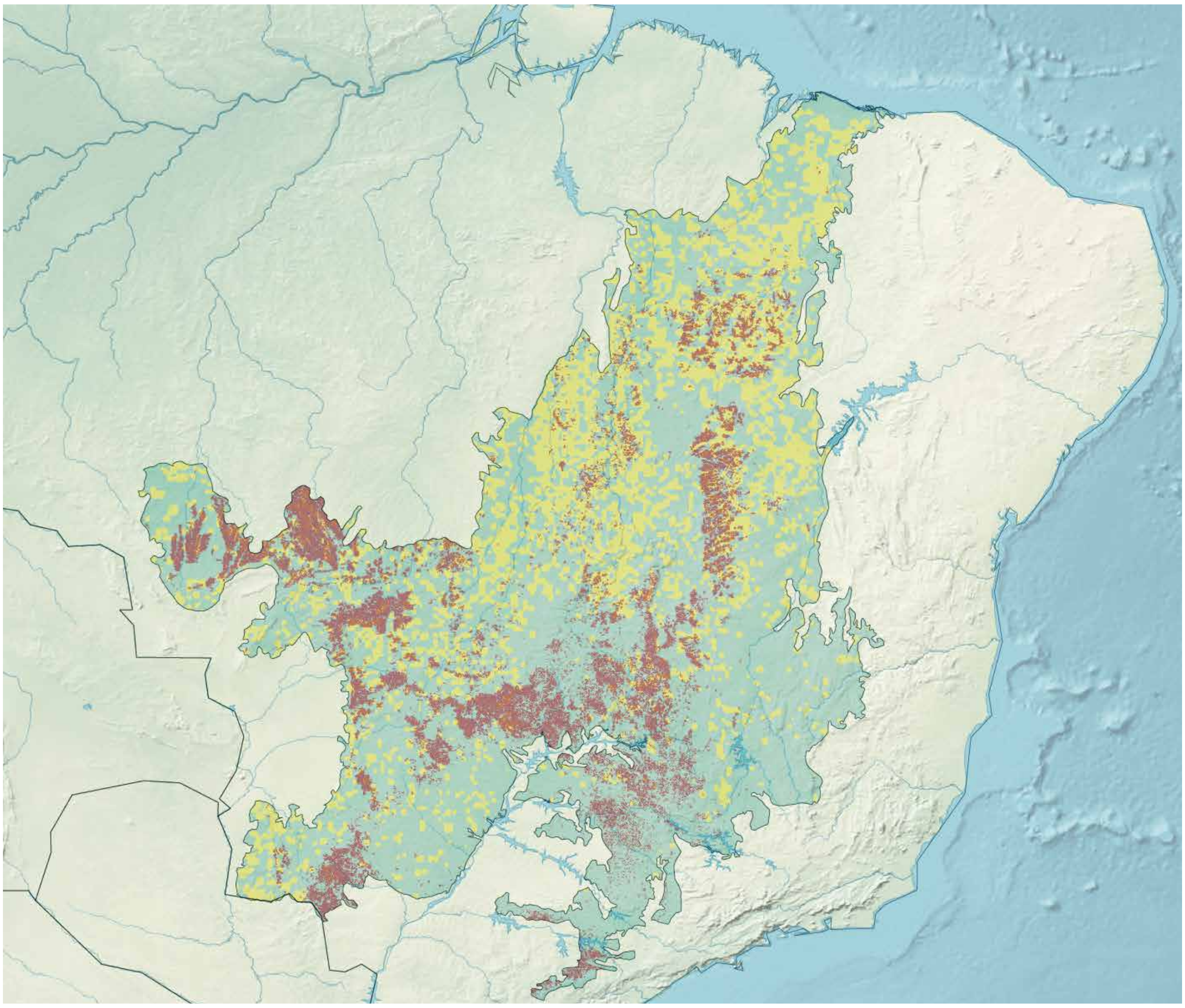

Figure 21. Commodity-driven forest loss 2013-18 (yellow) compared to 2014 soy area (red) in the Brazilian Cerrado

Source Global Forest Change dataset 1.7 via https://www.globalforestwatch.org/ *(Hansen et al., 2013); Rudorff et al. (2015)*

Figure 21 compares the main soy production areas in the Cerrado (as of 2014) with commodity-driven deforestation identified in the GFC dataset. It is clearly apparent that while in parts of the Cerrado deforestation has occurred in main soy producing areas, extensive deforestation has also occurred away from the soy frontier, likely for pasture. As in other areas, soy expansion is secondary to livestock ranching as a proximate deforestation driver in the Cerrado.

European Commission (2019a) cites Gibbs et al. (2015) for the rate of expansion of soy on high carbon stock land in the Cerrado. Gibbs et al. (2015) found that 15% of soy expansion in the period 2009-2013 was on land cleared of forest within three years. Noojipady et al. (2017) presents an updated satellite analysis of cropland expansion in the Cerrado in the period 2003 to 2013. Several of the authors also worked on Gibbs et al. (2015). While soy is identified as perhaps the most important crop in terms of deforestation impact, we note that the analysis did not distinguish between crop types. The Noojipady et al. (2017) analysis did not place a temporal limit on identification of forest to cropland transitions, i.e. where an area was forest in 2003 and converted to cropland at any point in the period to 2013 this was identified as a crop expansion linked land use change. This could tend to overcount cases where cropping was a proximate deforestation driver (for instance if soy was planted nine years after an episode of forest loss it is less likely to be properly consider a proximate driver than if planted three years after forest loss). Tending to balance out any such overestimation out is the fact that for the most recent deforestation episodes in the period considered planned crop planting may still not have occurred (for instance if forest was cleared for soy in 2012 it is unlikely the soy would already have been planted in 2013). The definition of forest used is 10% canopy cover, which is consistent with the Commission definition of high carbon stock land. It is reported that there were 9 million hectares of cropland expansion in the Cerrado in this period, 1.7 million hectares of it in wooded areas. Based on SIDRA data we estimate that soy area accounted for about 70% of this reported expansion. In the period 2009-2013, 26% of crop expansion was identified as replacing wooded land. The role of crop expansion as a proximate driver of deforestation was particularly strong in the 'Matopiba' region[29].

Here, we take the Noojipady et al. (2017) estimate (26%) for the average for 2009-2013 as the best available assessment of the fraction of soy expansion on high carbon stock land in the Cerrado. While the assessment is not able to distinguish soy from other crop expansion, we are not aware of any evidence suggesting that soy is less deforestation-associated than other crops in the Cerrado region. Note that Noojipady et al. (2017) identifies a discrepancy between its MODIS based assessment of deforested areas, and deforested areas identified in the GFC (a significant fraction of the forest to cropland transitions identified with the MODIS analysis are not shown as tree cover loss in the GFC).

3.2.iii) Other Brazil

Malins (2019) assumed that 7.5% of soy expansion in other Brazilian biomes was on to high carbon stock land. European Commission (2019a) cite an unpublished paper submitted by Agroicone for other parts of Brazil, but it is not clear what the assumed fraction of expansion on high carbon stock land is. Soy expansion outside the Amazon and Cerrado is concentrated in the Mata Atlantica (Atlantic forest) biome and in the Pampas (Rio Grande do Sul). Graesser et al. (2015) identify minimal cropland expansion onto forest in the Pampas, but in the Atlantic forests identify about 30% of new cropland coming from high carbon stock land. In the absence of strong new evidence, we follow Malins (2019) and assume that an average of 7.5% of soy expansion is deforestation associated in the rest of Brazil.

3.2.iv) The Chaco

Outside of Brazil, the Gran Chaco region), spanning parts of Paraguay, Bolivia and Argentina (Figure 22), is a key region of soy expansion and deforestation concern (Malins, 2019; Yousefi et al., 2018). The Gran Chaco covers an area of about 100 million hectares. The Chaco can be split into the semi-arid dry Chaco (the larger drier western and southern part of the area) and the humid or wet Chaco, concentrated in Paraguay on the eastern side. NASA Earth

29 An agricultural frontier region spanning parts of Maranhão, Tocantins, Piauí, and Bahia states.

Observatory (2019) reports that about one fifth of the Gran Chaco forest was cleared in the period 1985 to 2013, with 2.9 million hectares cleared for pasture and cropland in the period 2010 to 2018. This may mark a reduction in deforestation rate compared to the previous decade - Fehlenberg et al. (2017) reports that 7.8 million hectares were lost in the period 2001 to 2012. In the same period, soybean area was identified as more than doubling, increasing by 2.9 million hectares. Cattle head increased from 21 to 27 million in the period.

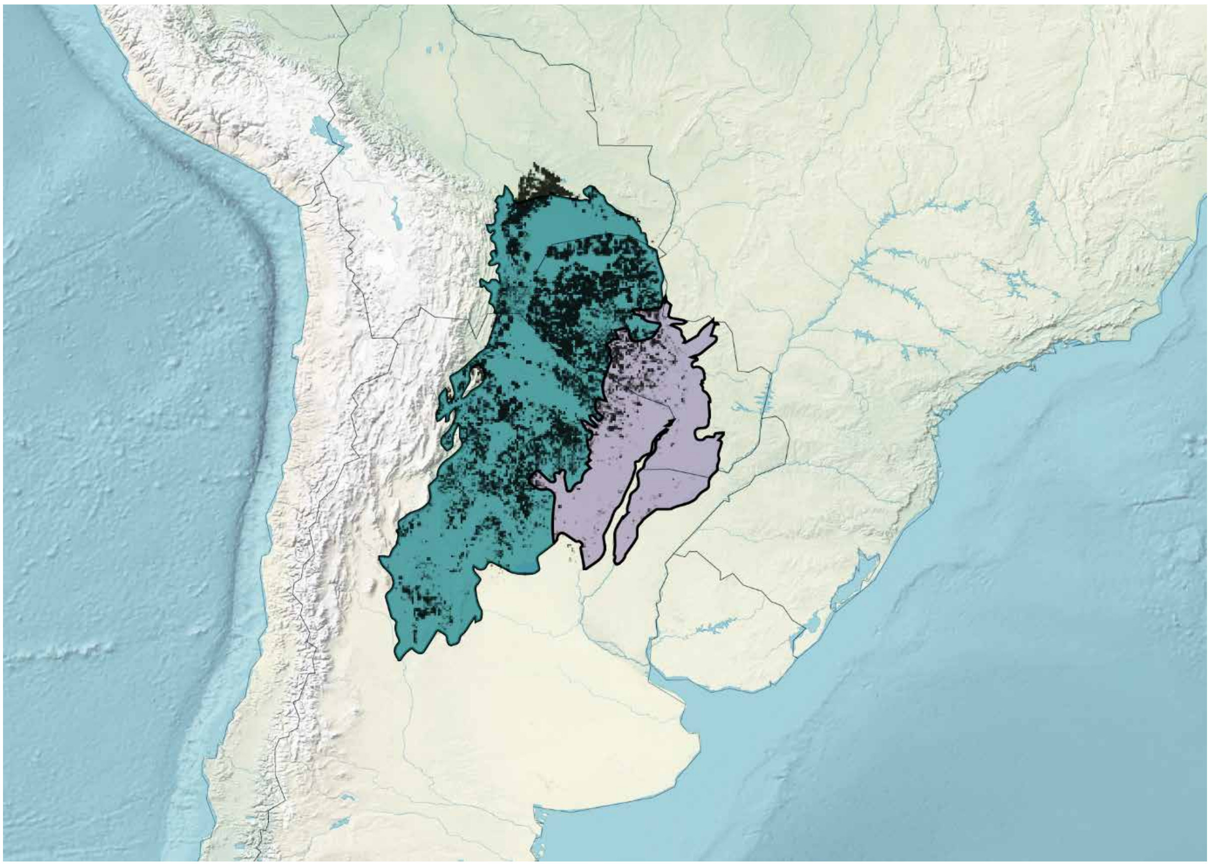

Figure 22. Deforestation (black) in the Gran Chaco 2011-2018

Source: Arévalos et al. (2018) Accessed through Global Forest Watch www.globalforestwatch.org

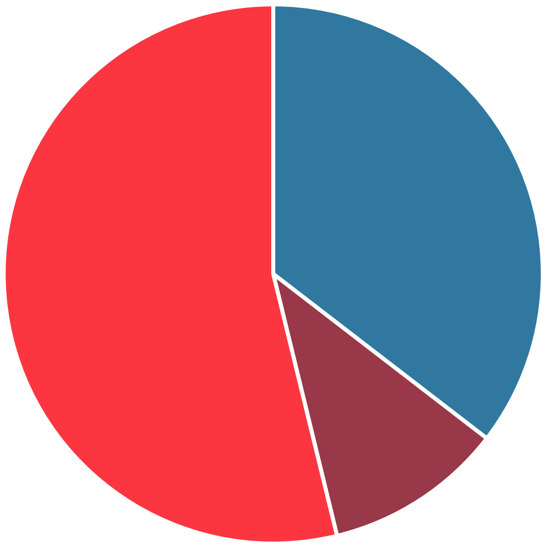

Figure 23. Fraction of reported forest lost in the Chaco January 2012 – June 2018

Source: Arévalos et al. (2018)

Deforestation in the Chaco is monitored by the Guyra NGO in Paraguay using LandSat data (Arévalos et al., 2018). In the period from January 2012 to June 2018, 3 million hectares of forest lost were identified, with over 50% in Paraguay (Figure 23). Through the period, deforestation rates have reduced slightly in Argentina, been relatively steady in Paraguay and increased slightly in Bolivia (Figure 24b).

Fehlenberg et al. (2017) uses econometric analysis to explore correlations between soy expansion, cattle expansion and deforestation in the Chaco in the period 2000-2012. In general, the correlation between cattle head and deforestation rate was found to be stronger than the correlation between soy area and deforestation rate. The correlations were stronger in the Argentinean Chaco. This to some extent echoes analysis in Baumann et al. (2016) identifying soy expansion as the main underlying driver of deforestation in the (mainly Argentine) dry Chaco, but pasture expansion as the main deforestation drier in the wet Chaco (largely in Paraguay). Econometric analysis in Fehlenberg et al. (2017) found that soy expansion was linked to additional deforestation, with 0.03-0.08 hectares of additional deforestation in the period 2000 to 2012 for every additional hectare of soybean area. The results are mixed, however, with a negative relation shown between total national soy area and deforestation rates in Bolivia and Paraguay.

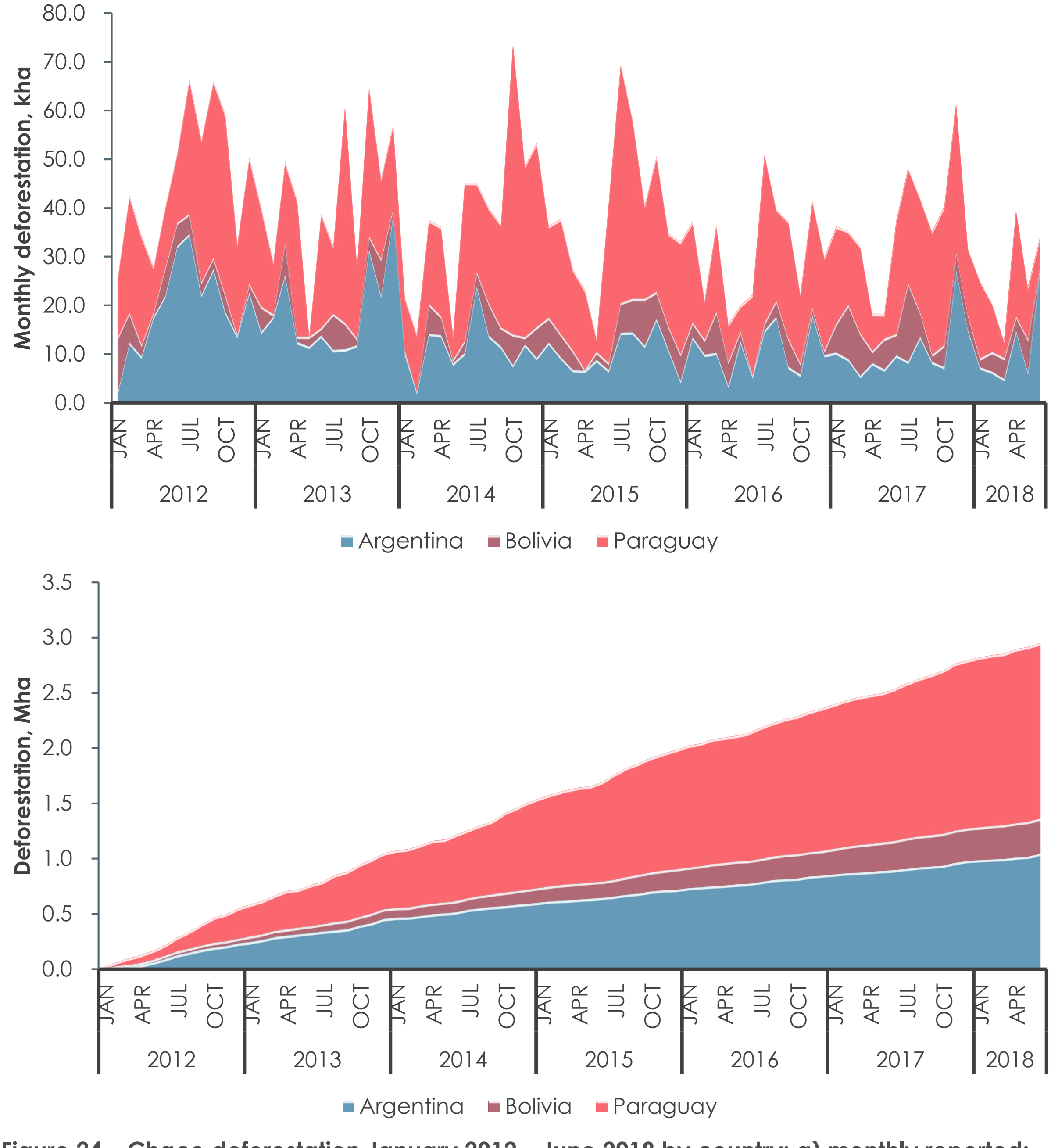

Figure 24. Chaco deforestation January 2012 – June 2018 by country: a) monthly reported; b) cumulative

Piquer-Rodríguez et al. (2018) also use econometric analysis with a view to identifying drivers of deforestation in the Argentinean Chaco, this time in the period 2000-2010. This paper found that deforestation for cropland was significantly more likely in the vicinity of existing cropland areas. It also found that conversion of woodland to cropland was relatively insensitive to the profitability of cropland, and hypothesised that, "It is nearly always profitable to convert woodlands to croplands or grazing land independently of profit change." The paper 'cautiously concludes' that intensification of agricultural use (i.e. conversion of pasture to cropping) may be more sensitive to international commodity demand than absolute rates of deforestation.

Leake et al. (2016) study deforestation in one region (Salta) of the Argentinian Chaco in the period 2004 to 2015 that is highlighted by (NASA Earth Observatory, 2019) (Figure 25). Based on assessment of hearings for land use change projects, it was found that 56% of land use change proposals by area related to livestock projects, 25% to a combination of livestock and cropping, and 17% to projects for cropping only.

Figure 25. Satellite evidence of deforestation in the Salta region of the Chaco, 2000-2019

Source: NASA Earth Observatory (2019)

Baumann et al. (2017) assessed proximate drivers of deforestation in the Paraguayan Chaco, concluding that expansion of grazing land was the dominant proximate driver of deforestation, but that deforestation still made a significant contribution to cropland expansion where cropland expansion occurred – 47% of new cropland in 2012 was converted from woodland since 2000, broadly consistent with the Graesser et al. (2015) result that 57% of crop expansion in Paraguay is associated with deforestation.

Deforestation in the Chaco is taken into account in our assessment of the fraction of soy expansion on high carbon land through the national fractional values detailed in the next section.

3.2.v) Other South America

In European Commission (2019a) Paraguay and Bolivia are both identified as having a very strong proximate link between cropland expansion and deforestation, based on results from Graesser et al. (2015) for the period 2001 to 2013, with 57% and 60% respectively of new cropland on deforested land in the period 2001-13. Only part of this deforestation falls within the Chaco. According to FAOstat, harvested soy crop area increased by 870 thousand hectares in Paraguay and 240 thousand hectares in Bolivia in the period 2010-15, suggesting that soy expansion could have contributed to this deforestation.

For Paraguay, Graesser et al. (2015) shows cropland expansion primarily in the east of the country. Paraguay spans three forest biomes: the dry Chaco, the humid Chaco, and the Atlantic forest. In 2004, Paraguay introduced laws limiting further deforestation of the Atlantic forest areas[30], after more than two thirds of forest in the region was lost in the period 1973 to 2000 (Huang et al., 2007), but these have been only partially successful - Da Ponte et al. (2017) reported that only 10% of the Upper Parana Atlantic forest remains. Nevertheless, Baumann & Fehlenberg (2016) report that these measures have led to an increased rate of pasture conversion to cropland in the east of the country, with a knock-on increase in pasture expansion in the Chaco, which is also suggested by Fehlenberg et al. (2017). In the absence of a more up to date assessment, we follow European Commission (2019a) and assume that 57% of soy area expansion in Paraguay is likely to be at the expense of forest.

Using the Graesser et al. (2015) result for Bolivia is more problematic. The satellite analysis for the study identified only 30 thousand hectares of net cropland expansion in the period in question, apparently all[31] in the Chiquitano dry forest, to the north of the Chaco (examples of deforestation in this area are provided by Yousefi et al., 2018). In contrast, FAOstat reports an increase of cropland area by 1.3 million hectares in the same period, including a 610-thousand-hectare expansion of harvested soy area. Unless the reporting to FAO is completely misleading, It seems likely that the Graesser et al. (2015) satellite land use identification approach was unable to accurately distinguish land uses for Bolivia.[32] Given that the results cover only a small fraction of reported total cropland expansion in Bolivia, it is difficult to justify assuming only on that basis that fully 60% of cropland expansion is likely to result in deforestation. Supporting evidence is provided, however, by Müller et al. (2014) which reviewed drivers of deforestation in Bolivia and identified 1.8 million hectares of deforestation in the period 2000 to 2010, with the proximate driver split about equally between cattle ranching and cropland expansion (split roughly 3:2 between mechanised and small scale agriculture). In that period FAOstat identified Bolivian total cropland expanding by 1.2 million hectares. That suggests that in the period 2000 to 2010 as much as 70% of new cropland in Bolivia came from forest. We therefore continue to assume that 60% of soybean expansion in Bolivia will come at the expense of forest.

In Argentina, Graesser et al. (2015) shows cropland expansion in the humid pampas as well as Chaco, but with very little associated deforestation. Similarly, cropland expansion is shown on the Uruguayan savanna but with very little deforestation impact.

We again follow European Commission (2019a) by using the Graesser et al. (2015) values (9% and 1% respectively) for Argentina and Uruguay.

3.3. Soy expansion in the United States

Spawn et al. (2019) provides an assessment of the carbon consequences of U.S. cropland expansion in the period 2008-12. In this period, soy area expanded by about 1,500 km2 per

30 https://wwf.panda.org/?16890/paraguay-bans-conversion-of-the-atlantic-forest

31 The reported results for cropland expansion Bolivia appear identical to the results for the Chiquitano, which suggests that the identified cropland expansion lies entirely in the intersection between Bolivia and the Chiquitano ecoregion.

32 It should be noted that there are also discrepancies between satellite identified and reported cropland area change for other countries, but Bolivia stands out for having a factor 40 discrepancy.

year with an estimated average carbon stock loss of 62 tonnes per hectare, mostly from soil carbon (Figure 26).

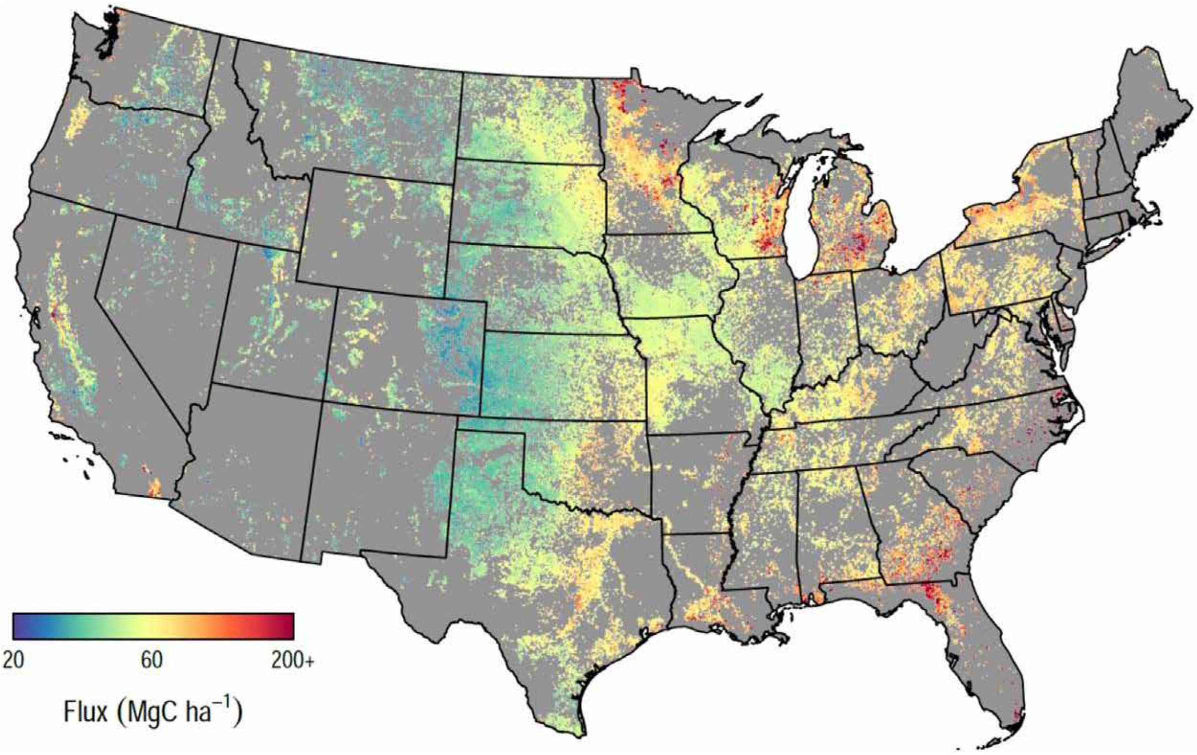

Figure 26. Mean carbon flux resulting from U.S. cropland expansion 2008-2012

Source: Spawn et al. (2019), Figure 1

Overall cropland expansion in the U.S. was dominated by grassland conversion, with about 8% shrubland conversion, 3% forest conversion and 2% wetland conversion. There is no evidence given that soy expansion was disproportionately associated with forest loss compared to other crops, and the identified carbon loss per hectare associated with soy expansion is comparable to other crops. European Commission (2019a) assumes that 2% of soy expansion outside of South America is at the expense of forest. For the analysis below, we increase this for expansion in the USA to 3% on the basis of the Spawn et al. (2019) results.

Shrubland will generally have more biomass carbon stocks than grassland, however the mass of carbon is still generally relatively modest - Spawn et al. (2019) report (in the supplementary data) a range from 0.5 to 8 tonnes carbon per hectare in aboveground biomass for shrubland. Grassland is assessed with aboveground biomass carbon stocks up to 2 tonnes per hectare. The average for forests is reported as about 30 tonnes per hectare. Typical soil carbon loss on forest conversion is also expected to be higher (about 90 tonnes per hectare) than for grassland or shrubland (50 and 40 tonnes per hectare respectively). In general therefore we would expect that carbon losses from grassland and shrubland conversion in the United States will be significantly lower than from conversion of RED II defined high carbon stock areas, although there will undoubtedly be some specific cases where conversion of grassland/shrubland with less than 10% canopy cover but with high soil organic carbon content would result in larger CO_2 emissions than conversion of wooded land with low soil organic carbon content.

3.4. Soy expansion elsewhere

There is much less focus on or documentation of links between soy expansion and deforestation in the rest of the world, in part because the areas involved are much smaller than in the Americas, and in part because the association is expected to be much weaker than in Latin America. We follow European Commission (2019a) and assume that 2% of soy expansion in other countries is deforestation associated.

3.5. Estimated fraction of soy expansion onto high carbon stock land

By combining assumptions about the fraction of soy expansion likely to occur on high carbon stock land with data on the location of recent soy expansion, it is possible to come up with an updated estimate of the likely fraction of global soy expansion that affects high carbon stock areas. The most significant change in the analysis presented here as against the analysis presented by European Commission (2019a) is that a larger fraction than previously of Brazilian soy expansion is now occurring in the Cerrado, and that the assumed fraction of soy expansion in the Cerrado that is deforestation associated has been increased based on the evidence from Noojipady et al. (2017).

This leads to a higher assumed fraction of soy expansion in Brazil that is deforestation associate, increasing to 15.6% compared to the 10.4% assessed by European Commission (2019a) or 9% assessed by Malins (2019). Globally, the revised assessment suggests a fraction of 10.5% of soy expansion on high carbon stock land, higher than was anticipated by the European Commission (2019a) or Malins (2019). If that value was confirmed by additional analysis for the European Commission, under the current rules soy oil would cross the threshold to be identified as a high ILUC-risk biofuel feedstock.

Table 2. Expansion of global soy area and expected associated impact on high carbon stock areas

		Fraction of global soy expansion[33]	Fraction of expansion associated with deforestation	Corresponding value from European Commission (2019a)[34]
Brazil	*Caatinga*	3.1%	7.5%	3.0%
	Cerrado	21.6%	26.0%	14.0%
	Mata Atlântica	7.1%	7.5%	3.0%
	Pantanal and Pampa	6.6%	7.5%	3.0%
	Amazônia	7.4%	3.5%	2.2%
	Total	***45.8%***	***15.6%***	***10.4%***
Argentina[35]		0.0%	9.0%	9.0%
Paraguay		3.0%	57.0%	57.0%
Uruguay		1.0%	1.0%	1.0%
Bolivia		1.0%	60.0%	60.0%
Total Latin America		**49.7%**	**18.6%**	**14.0%**
USA		24.6%	3.0%	2.0%
Rest of world		24.9%	2.0%	2.0%
Global total		**100.0%**	**10.5%**	**8.0%**

33 Analysis based on most recent eight years of data available – within Brazil 2013-20 soy expansion data from SIDRA to give the best estimate of the fraction of soy expansion on forest in Brazil, then for averaging across countries 2011-18 data from FAOstat.

34 Note that the Commission assessment also considered a different period (2008-17) for the fractions of global crop expansion.

35 The total reported area of soybeans in Argentina has shrunk over the period considered, primarily due to incraese in maize and wheat area, even though some expansion into forested areas continues to be reported in the Chaco. The fraction of net global soybean expansion attributed to Argentina is therefore 0%. The accompanying value for assumed fraction of forest expansion associated with deforestation is non-zero because a) the period considered by Graesser et al. (2015) is longer; b) it is calculated for all cropland, not only for soy cropping; and c) it reflects gross expansion.

4. Soy market development and prospects, 2005-2030

The global soy market is primarily driven by the livestock feed market – soy meal represents between two thirds and half the value of the soy crop. Biofuel demand is a secondary driver, but increasing the use of soy oil for biofuels may displace soy oil from human consumption. Due to links between vegetable oil markets this may affect palm oil area as much as or more than soybean area. The global soy market is expected to continue to grow in the coming decade driven by growing meat consumption, resulting in further increases in soy area. In the EU, the classification of palm oil as high ILUC-risk may create a market opportunity for soy oil, which is perhaps the most likely alternative for biofuel producers currently reliant on palm. We explore scenarios in which soy consumption for EU biofuels would grow by between two and five million tonnes by 2030 to meet demand under RED II. Considering ILUC modelling for the European Commission we show that this would be expected to deliver at best a very small net GHG emissions reduction, and at worst could cause millions of tonnes of additional net CO_2e emissions compared to continuing to use fossil diesel.

4.1. The global soy market

The soy crop is primarily an animal feed crop. While about 2% of global production is consumed directly by humans in products such as tofu Goldsmith (2008), about 90% of the global crop is crushed to produce soy meal for animal feed and soy oil, with the remainder fed to animals directly as soybeans OECD-FAO (2020).

The price of soybean oil is higher (per unit mass) than that of soybean meal, but as shown in Figure 27 the meal still provides most of the value from the soybean crush because more meal is produced. Meal accounted for about two thirds of the value on average across the 20-year period shown.

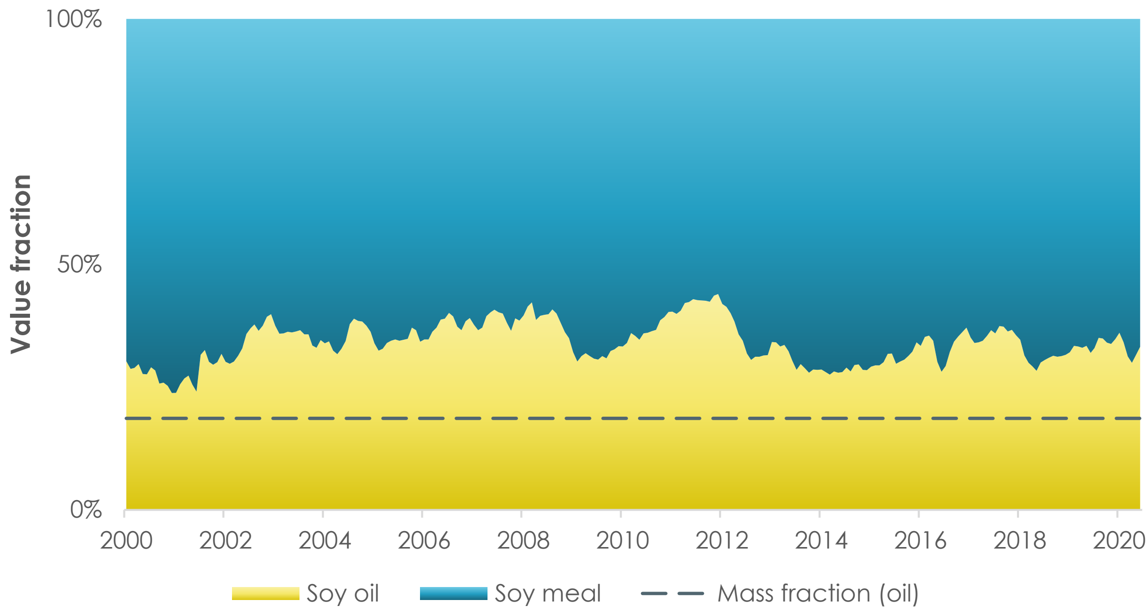

Figure 27. Relative value of soy meal and oil, 2000-2020

Source: World Bank (2020); Irwin (2017)

Soy production expanded rapidly in South America from the late 1990s, led by increasing global demand for meat (and therefore animal feed), especially in China (le Polain de Waroux et al., 2017). By 2002 total soybean production in South America had overtaken the U.S., and production in Brazil this year is expected to exceed that in the U.S. for the first time ever.[36] This reflects a combination of reduced area planted in the U.S.[37] in 2019 (OECD-FAO, 2020) and continued expansion of the area in Brazil (Figure 8).

Berkum & Bindraban (2008) identified the combination of population growth and economic growth as the main underlying factors driving increases in soy demand, in particular because of the role of soy as a livestock feed product (with soy demand therefore being driven by increasing meat demand). China in particular is heavily dependent on soy imports for cattle feed and therefore a major driver of the international soy trade. In general China imports whole soybeans, crushing them domestically to produce soy meal for livestock and soy oil for food use (providing an alternative to palm oil imports). Import data from UN ComTrade[38] shows that growing Chinese demand for soybeans has indeed been the most important market development over the past fifteen years. As shown in Figure 28, over the period from 2005 to 2018 China's imports from the major soy exporters[39] more than trebled from 26 million tonnes to 84 million tonnes. Total soybean imports by other countries changed relatively little through the period.

36 https://www.world-grain.com/articles/13108-brazil-to-overtake-us-as-leading-soybean-producer

37 This was influenced by adverse planting conditions in parts of the U.S. in the first half of the year.

38 https://comtrade.un.org/

39 Here we consider the USA, Brazil, Argentina and Paraguay.

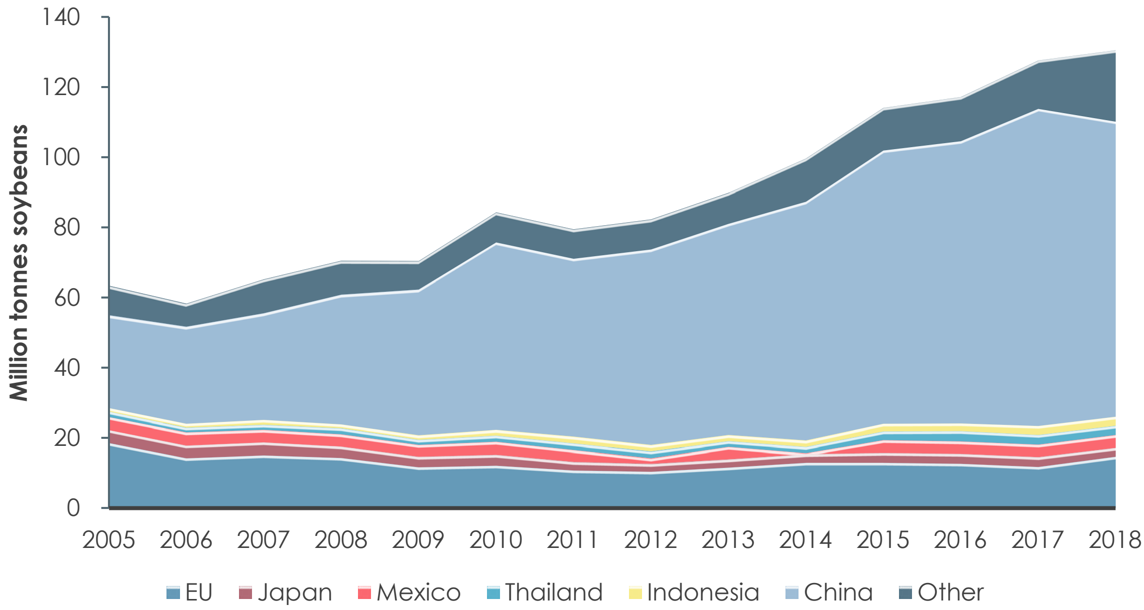

Figure 28. Global imports of soybeans (uncrushed) from USA, Brazil, Argentina and Paraguay, 2005-2018

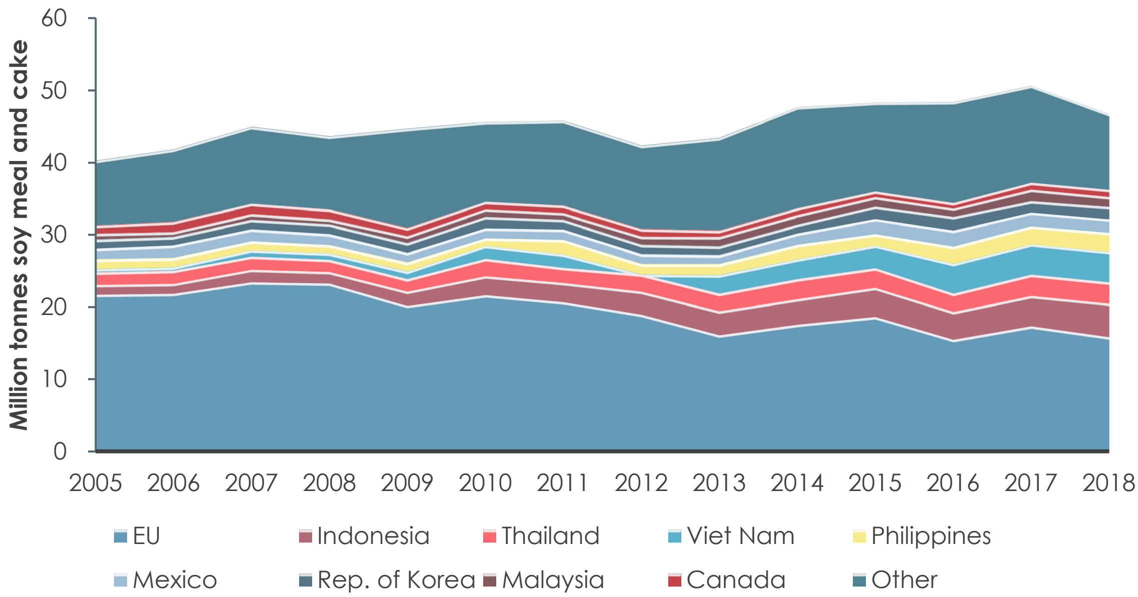

Figure 29. Global imports of soy meal from USA, Brazil, Argentina and Paraguay, 2005-2018

The trade in soybeans is larger by both mass and value than trade in soy meal and soy oil from crushing. Figure 29 and Figure 30 show that the amount of meal and oil traded has been relatively stable overall through the period, with Indian oil imports increasing as China's have reduced. The increase in imports from India does not appear to be primarily driven by biodiesel use (biodiesel consumption is reported to have risen from around 100 million litres per year from 2010-2015 towards 200 million litres in 2018 (Aradhey & Wallace, 2018), but this would account for only a fraction of the reported increase in soy oil imports).

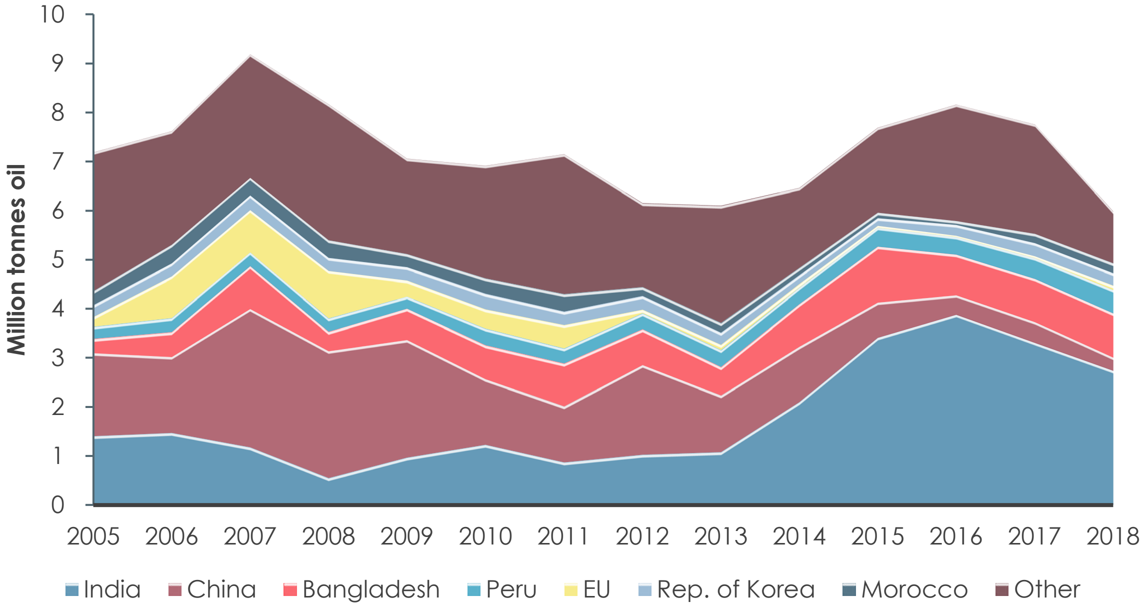

Figure 30. Global imports of soy oil from USA, Brazil, Argentina and Paraguay, 2005-2018

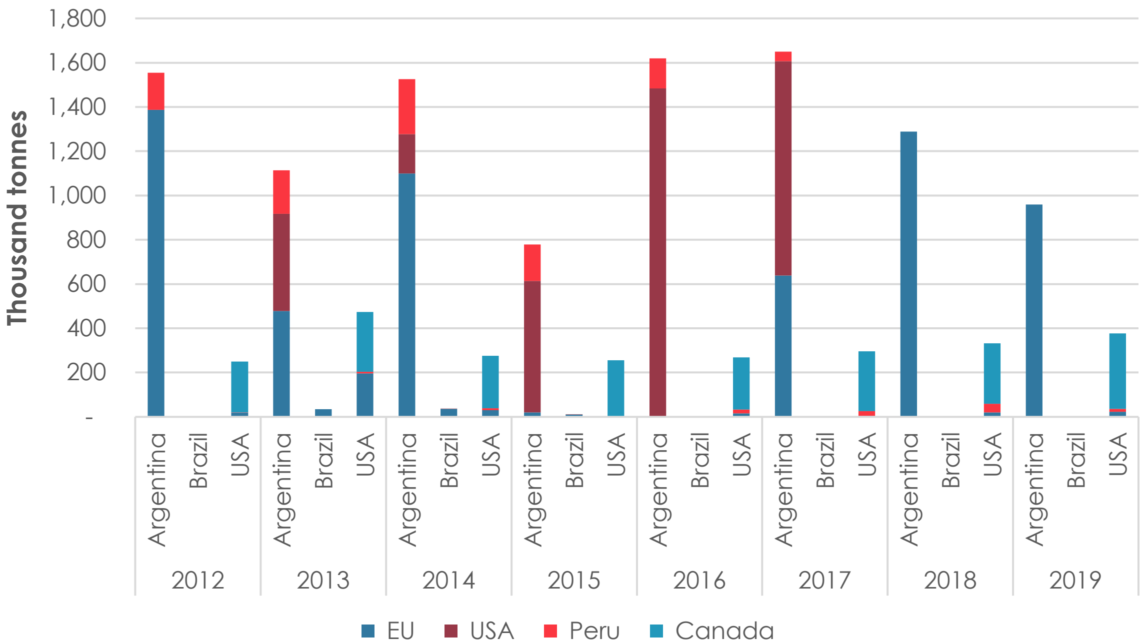

Figure 31. Exports of biodiesel from the USA, Brazil and Argentina by destination country, 2012 to 2019

Source: UN ComTrade

Soy oil imports to Europe increased during the 'biofuel boom' in the period 2005 to 2011, but since 2012 have been much reduced. Since anti-dumping measures against Argentinian biodiesel were relaxed in 2018 significant export of biodiesel to the EU has resumed, as shown in Figure 31. This has been accompanied by a reduction in soy biodiesel exports from Argentina to the U.S. Argentinian biodiesel exports are likely entirely or almost entirely soy biodiesel. While there are a few significant trade flows, overall the global soy biodiesel market is dominated by domestic production for domestic use. About half of Argentinian biodiesel production (about 1.3 billion litres) is consumed domestically (most of the rest is exported to the EU) along with almost all Brazilian biodiesel (about 5.8 billion litres) and the substantial majority of U.S. biodiesel (of which about 4 billion litres is soy based).

4.1.i) Soy market to 2030

OECD-FAO (2020) anticipates that soybean production will grow by about 1.3% per annum in the period to 2029, led by yield increase (two thirds of output growth) over harvested area increase (one third of growth). Growth in protein meal demand is expected to be more modest in the coming decade than the previous one, due to slower growth in global pork and poultry production and by anticipated efforts in China to reduce the protein share in livestock rations. Prices for vegetable oils and meals are forecast to be relatively stable in real terms to 2029.

Fuchs et al. (2019) discusses the possibility that souring trade relations between the United States and China could drive a spike in deforestation if soy area expands in South America to compensate for reduced exports of soy from the U.S. to China. This paper hypothesises that China is likely to increase imports from Brazil, and that this could require an area increase of up to 6 million hectares. In the context of weakening enforcement of forest protection rules in Brazil it is suggested that this could drive a spike in deforestation. Evidence from the year to date suggests that this scenario may indeed be realised[40], with a 30% year-on-year increase in soy exports from Brazil to China in the first half of 2020.

In terms of soy oil demand as a biofuel, there is most potential for consumption growth between now and 2030 in the EU itself and in Brazil, the United States and Argentina, and by the aviation industry. As discussed in Malins (2020), Brazil has a target to increase biodiesel blending from 11% to 15% by 2023. It has also been reported recently that the Brazilian national oil and gas regulator is consulting on increased production of soy oil renewable diesel by the national oil company Petrobras.[41] In the medium case for 2030 soy biodiesel demand from these markets discussed in Malins (2020) total use of soy oil for biodiesel would increase by about 12 million tonnes through the decade. If realised, that increase in soy oil demand would use up about a third of the total global increase in vegetable oil production expected by OECD-FAO for the decade (OECD-FAO, 2020)[42]. If all regions followed the high demand scenarios outlined by Malins (2020).

40 https://chainreactionresearch.com/soy-brazil-deforestation-cerrado/

41 https://www.bnamericas.com/en/news/brazil-bets-on-green-diesel-to-help-energy-transition; https://www.greencarcongress.com/2020/08/20200803-petrobras.html

42 At the time of writing the OECD-FAO outlook only goes up to 2029, so we have treated the production increase forecast for 2019-2029 as indicative of the increase that could be expected 2020-30.

4.2. Prospects for soy biofuel demand in the EU and UK, 2020 to 2030

Soybean oil is currently the third most used virgin vegetable oil for biodiesel and renewable diesel production in the EU, with nearly a million tonnes a year being consumed by EU producers. According to figures from OilWorld, the use of soy oil as feedstock[43] by European biodiesel and renewable diesel processors has trebled since 2013. Added to imports of soy biodiesel primarily from Argentina (also nearly a million tonnes in 2019, down from 1.7 million tonnes in 2018[44]), about 1.7 million tonnes a year of soy oil demand came from the EU biofuel market in 2019 (Figure 32).

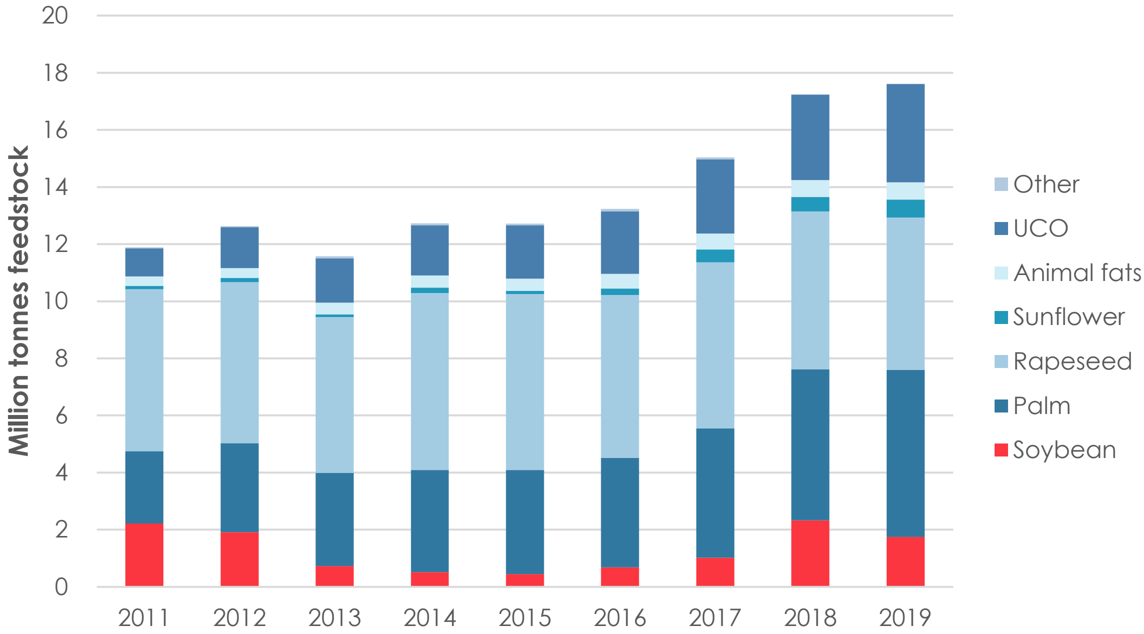

Figure 32. Feedstocks for EU biodiesel and renewable diesel consumption, 2011-2019

Repeated figure. See notes on Figure 6.

The EU's imports of finished biodiesel have been strongly impacted over the past twenty years by the interplay between various production subsidies in exporter nations and EU countervailing import tariffs. For example, at the end of the last decade biodiesel was being exported to the EU from the U.S. taking advantage of the "splash and dash"[45] subsidy, under which biodiesel shipments (potentially originating outside the United States) would gain subsidy under both U.S. and EU schemes. This led the EU to impose countervailing tariffs on

43 In this report we assume that one tonne of soybean oil can be converted into 1,113 litres of biodiesel or 1,025 litres of renewable diesel.

44 In this report, we have treated all Argentinian biodiesel imports as soy based. Some fraction may be imported UCO biodiesel, as the EU market draws UCO based fuels from across the globe, but evidence from UK biofuel statistics (the UK being one of the most attractive UCO biodiesel markets in recent years) suggests that this is small compared to soy use in the Argentinian case.

45 http://www.biodieselmagazine.com/articles/1863/the-end-of-splash-and-dash

biodiesel imports from the U.S. (European Commission, 2020; Stearns & Graham, 2009), which are due to expire this year. Similar action has been extended recently against Indonesian palm oil biodiesel exports (European Commission, 2019b), but has been relaxed for Argentinean biodiesel imports[46,47]. With anti-dumping measures ongoing against Indonesia but removed (for now) from imports from the U.S. and Argentina, combined with prospective limits on palm oil biodiesel under the high ILUC-risk provision, there is scope for EU imports of soy oil biodiesel to increase further. There is also potential for soy oil to replace palm oil as the 'import oil of choice' for EU-produced biodiesel.

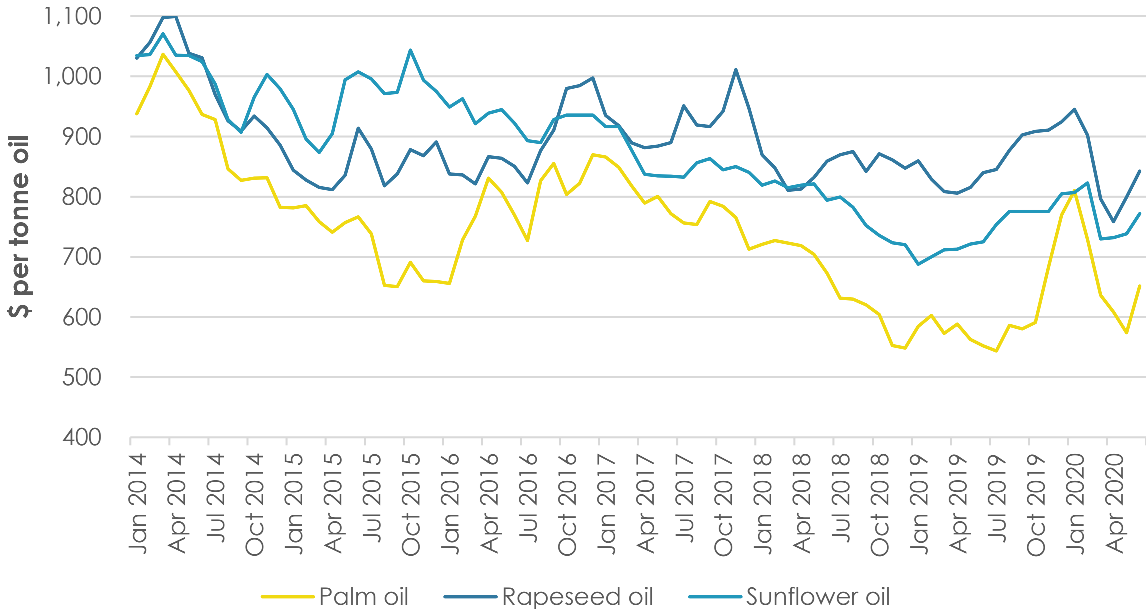

Figure 33. Vegetable oil prices on the world market, 2014-2020

Source: (World Bank, 2020) inflation adjusted. Note that y-axis starts at 400 $/tonne.

As is seen in Figure 32, imported oils have never superseded rapeseed oil as the main crop oil processed into biodiesel in the EU, but palm oil in particular has had a significant market share nevertheless. The use of tropical oils (soy and palm) for biodiesel is somewhat limited because they have worse cold flow properties than rapeseed oil[48] (i.e. they are more likely to start to solidify in the vehicle in cold periods in the winter), but there is a clear business case to consider using lower cost imported oils for biodiesel production, especially in the summer. Commodity price data published by the World Bank (2020) illustrates this (Figure 33). It is clear that palm oil is systematically the lowest cost virgin oil available on the world market – on average, the difference between rapeseed oil price and palm oil price was 150 $/tonne in

46 Cf. https://bioenergyinternational.com/markets-finance/european-biodiesel-imports-from-argentina-and-indonesia-increase-sharply

47 As we understand it, countervailing duties on Argentinean biodiesel imports are still formally in effect, but only for biodiesel below a certain price. (Flach et al., 2020) states that implementing this floor price has allowed imports from Argentina to continue.

48 This does not apply to renewable diesel, for which the feedstock's cold flow properties are not important).

the period shown. The soy oil price tracks rapeseed oil much more closely, but in general is lower – by an average of 40 $/tonne in this period.

The tariffs on vegetable oil imports are also relevant in this context. Commission Regulation (EC) No 1549/2006 sets a tariff of 3.2% on crude soy oil imports for industrial uses, but no tariff on equivalent palm oil imports. During the period considered, the cost of this additional tariff was on average about 70% of the cost spread between rapeseed and soy oils, which could undermine the business case for using soy as a palm oil alternative. With a much narrower spread between rapeseed and soy oils than rapeseed and palm , it is not immediately clear to what extent a reduction in palm oil supply due to the high ILUC-risk regulation would lead to increased use of soy oil in the EU supply as opposed to increased use of rapeseed (or perhaps other options).

Malins (2020) presents three scenarios for the development of EU demand for soy oil as biofuel feedstock in the period to 2030 – a low scenario in which direct demand is eliminated through a high ILUC-risk designation, a medium scenario in which soy oil use grows to partly substitute reduced palm oil use, and a high scenario in which soy oil completely replaces palm oil use. Given the relative prices, something like the medium scenario seems most likely in the absence of a high ILUC-risk designation.

Assessing potential demand for soy oil for EU biofuels requires some consideration of potential total biodiesel demand in 2030. To estimate total food-based biodiesel consumption in the EU plus UK[49] in 2030, we assume total transport energy demand and liquid transport fuel demand reported by European Commission (2018) with the gasoline:diesel:kerosene split modelled by European Commission (2016), and then assume blending of B7 biodiesel across the EU and that the maximum allowable amount (1.7% of transport energy) is supplied in the form of waste-based biodiesel.

That leaves 8.4 billion litres of food-based biodiesel consumption in 2030. We estimated that in 2019 about 6.6 billion litres of rapeseed and sunflower oil biodiesel were consumed in the EU. If the supply of rapeseed and sunflower biodiesel remains steady to 2030 and soy is the only other major crop feedstock for biodiesel, it would leave about 1.6 billion litres of soy oil biodiesel consumption (1.5 million tonnes soy oil demand). If instead the supply of rapeseed and sunflower oil biodiesel declined in line with the overall size of the food-based biodiesel market, that would leave 4.3 billion litres of soy biodiesel consumption (3.9 million tonnes of soy oil demand).

The calculation above would be consistent with about 4% of EU+UK[50] transport energy demand coming from food-based first-generation biofuels, below the 7% maximum limit set under RED II. While some Member States are likely to set lower caps on food-based biofuels, the total food-based biofuel supply in 2030 is generally expected to be more than 4%[51]. We therefore consider that additional food-based biofuel would be supplied into the market in

49 The UK is included in this generation of EU energy modelling.

50 As regards the UK in particular, the Renewable Transport Fuel Obligation sets a relatively low limit on the use of crop-based biofuels going forward, setting a cap at 2% for 2032 (Department for Transport, 2017). The UK Government anticipates that this cap will primarily allow ongoing supply of crop-based ethanol, and therefore we would anticipate a minimal supply of soy oil fuels to the UK in the 2030 timeframe.

51 And we note that a suggestion from the European Commission to reduce food-based biofuel use to about this level by 2030 in the original RED II proposal was rejected as too constrictive.

the form of renewable diesel (or jet) to meet Member State targets. An additional 7 billion litres of renewable diesel supply would bring food-based biofuel use to 6% of EU+UK transport energy, given the other assumptions above. This would be broadly consistent with the rapidly expanding renewable diesel capacity in Europe.[52] If half of this was produced from soy oil it would increase implied soy oil demand for EU biofuels in 2030 by an additional 3.4 million tonnes, more than total current demand from the EU biofuel market. If a third was soy based that would increase soy oil demand by 2.3 million tonnes.

Note that because this analysis based on the split between fuel types predicted in the EUref2016 scenario from European Commission (2016), the more recent reduction in diesel vehicle sales[53] has not been factored into the assessment of overall potential diesel-type fuel sales, which may lead to an overestimation of capacity to supply blended biodiesel. Reduced supply of food-based biodiesel would, however, create space within the cap on food-based biofuels for additional HVO supply, and thus reduced sales of diesel cars may not have a strong impact on total crop-based vegetable oil demand for biofuels.

In practice, demand for soy oil for EU biofuels will be sensitive to factors including the rate of electrification of road transport, to the price spread between soy oil and rapeseed/ sunflower oils, and to any measures taken by soy producing countries to support exports (and to any EU counter-measures to reduce imports). Here, we consider two levels of soy oil demand for EU biofuels in 2030. The lower level assumes **1.6 billion litres of soy biodiesel consumption and 2.3 billion litres of soy HVO consumption**, approximately double current estimated soy oil consumption for EU biofuels. The higher level assumes **4.3 billion litres of soy biodiesel consumption and 3.5 billion litres of soy HVO consumption**, more than four times current estimated soy oil consumption for EU biofuels. This gives a demand range from **3.7 to 7.3 million tonnes of soy oil demand as feedstock** in 2030.

Given the association between soy expansion and deforestation, and other reductions of land carbon stocks, continued supply of soy-oil-based biofuels has GHG emissions implications. Here we assess potential net emissions changes (compared to fossil diesel use) by assuming that supplies of both soy biodiesel and soy HVO have an average reportable emissions saving of 60% under the RED II methodology (considering only direct emissions), and then adding two scenarios for ILUC emissions based on studies for the European Commission with IFPRI-MIRAGE (Laborde, 2011) and GLOBIOM (Valin et al., 2015) (56 and 150 gCO_2e/MJ respectively). The ILUC factor was adjusted down by 7% for HVO to reflect the higher energetic yield per tonne soy oil compared to biodiesel.

Table 3 indicates the estimated net emissions change from using either 3.7 or 7.3 million tonnes of soy oil for biofuel in 2030, assuming a 60% reportable 'direct' GHG emission saving (equivalent to assumed average lifecycle emissions of 37.6 gCO_2e/MJ) and using the ILUC values for soy biofuels given by analysis As has been extensively discussed in the past (see e.g. Malins et al., 2014) there is considerable uncertainty associated with estimating ILUC emissions, and indeed more generally with assessing the net emissions impact of increased biofuel production (Whitaker et al., 2010). These values should therefore be treated as indicative of the range of potential impact.

52 https://www.icis.com/explore/resources/news/2019/11/28/10449207/insight-rise-of-hvo-to-be-the-downfall-of-traditional-biodiesel-in-europe

53 See e.g. https://www.acea.be/press-releases/article/fuel-types-of-new-cars-diesel-15.5-petrol-19.8-electric-43.8-in-second-quar

For the lower level of soy biofuel consumption (3.7 million tonnes feedstock demand) the calculation gives a range from a slight net carbon saving (400 thousand tonnes CO_2e) to a significant net increase in GHG emissions (about 19 million tonnes CO_2e). For the higher level of soy biofuel consumption (7.3 million tonnes of feedstock demand) the range is from a small net saving of 600 thousand tonnes CO_2e to a net increase in GHG emissions by 38 million tonnes CO_2e. A large difference is apparent in the results between the two ILUC studies because based on the IFPRI analysis the climate impact of soy biofuels is likely to be similar to that of fossil diesel, giving only a small net change, whereas the GLOBIOM ILUC results suggest that soy biofuels could be significantly worse for the climate than fossil diesel – i.e. in the better case using soy biofuels is about the same for the climate as using fossil fuels, whereas in the worse case using soy biofuels would significantly increase net CO_2 emissions.

Table 3. Estimated net GHG emission reduction (positive value) or increase (negative value) associated with scenarios for 2030 consumption of soy-based biofuels in the EU

		Lower case	Higher case
Soy oil demand (million tonnes)		4	7
Estimated increase from today (million tonnes)		2	6
Net emissions change (million tonnes CO_2e)	ILUC from IFPRI	0.4	0.6
	ILUC from GLOBIOM	-19	-38

Another way to consider the implications of this level of additional soy oil demand is to consider the land that would be required. Based on the default yield assumptions in BioGrace (2017) on average about 500 kg of soy oil is produced per hectare. At this yield, 3.7 million tonnes of soy oil production would require 7.2 million hectares, roughly equivalent to the size of Ireland. The high scenario of 7.3 million tonnes of soy oil production would require 14 million hectares, roughly equivalent to the size of Greece. These gross land demand values are slightly misleading, however, as they ignore the simultaneous production of soy meal from the soybean crop. If allocating land demand between meal and oil on an energy basis **the net land requirement for additional soy oil production would become 2.4 to 4.2 million hectares** for the lower and higher demand scenarios respectively (between the size of Slovenia and that of the Netherlands).

There is no precise way to estimate the deforestation impact of a given level of feedstock demand – the most sophisticated tools for making that assessment are ILUC models, which are seen to have considerable uncertainty in their outcomes. We can however do a simple calculation to give some indication of the potential deforestation risk. If we assumed that half of 2030 EU soy oil demand for biofuels was met by area expansion and the other half by yield improvement and/or food consumption reduction, ignored demand transmission into other vegetable oil markets[54], and used the estimate given above that 10.5% of new soy area globally replaces forested areas, we get **an indicative estimate of 130 to 230 thousand hectares of additional deforestation.**

54 In reality it is likely that there is significant demand transmission from the soy oil to the palm oil market, as discussed by Malins (2018).

5. Discussion and conclusions

Despite the introduction between 2004 and 2010 of a range of measures to manage tropical forest loss (including in the major soy producers Brazil, Argentina and Paraguay), agricultural expansion remains a major driver of tropical and sub-tropical habitat loss. In South America, this agricultural expansion has been dominated by livestock farming and soy farming. The identification of proximate deforestation drivers, especially to the level of distinguishing the impact of individual crops, is challenging, but in the areas of most interest (the Amazon, the Cerrado, the Chaco) we can conclude that while pasture expansion remains the dominant direct driver of deforestation, there is still significant expansion of soy and other crops onto high carbon stock areas.

Since the European Commission's preliminary assessment of high ILUC-risk feedstocks in 2019 (European Commission, 2019c), an increased fraction of soy expansion has occurred in the Brazilian Cerrado, and there is evidence (Noojipady et al., 2017) that a larger fraction of cropland expansion in the Cerrado has occurred on high carbon stock land than was assumed in the previous assessment. Integrating the more recent evidence we estimate the fraction of soy expansion onto high carbon stock land as 10.5%, which is above the threshold for categorisation as a high ILUC-risk biofuel. It should be understood that this assessment is based on literature review, and in parts on evidence that is five years old by now. Ideally, the final designation of high ILUC-risk biofuel feedstocks before 2023 will consider additional sources of data not available to us at this time. Nevertheless, if ongoing research for the Commission confirms this result, then soy oil biofuels will need to be phased out from support in the EU under the RED II in the period 2023-2030.

Identifying soy biofuels as high ILUC-risk would significantly reduce demand from the EU biofuel market by 2030, and avoiding the ILUC emissions associated with that volume could reduce net emissions by tens of millions of tonnes of CO_2e, although the size of that benefit is sensitive to estimates of how large those ILUC emissions really are. While the terms of the RED II encourage a focus on high ILUC-risk fuels, is It should not be forgotten that assessments of ILUC from other food oils (rapeseed and sunflower) have also shown large expected ILUC emissions. The benefits of switching from soy biodiesel to rapeseed biodiesel may be no greater than the benefit of switching back to fossil diesel. While the magnitude of the potential benefit from reduced soy biofuel utilisation is subject to the same uncertainty as all efforts to estimate ILUC emissions, what seems certain is that better climate outcomes would be delivered if the transition to advanced biofuels could be accelerated as an alternative to the use of crop-oils for biodiesel production.

The rules for identifying high ILUC-risk biofuels focus on which crops are proximate drivers of deforestation, in large part because proximate drivers are more readily identified than indirect drivers. In reality, though, agricultural systems are interlinked. Exploring the literature on the relationship between soy, cattle and deforestation in South America reminds us that commodity demand can play a complex role as an indirect driver of land use change. It also highlights that in reality deforestation can arise from a complex interaction of circumstances (legal enforcement, exchange rates, population displacement, capital availability and so forth) that are only covered in rough approximation by ILUC modelling and that cannot be addressed at all simply by identifying which agricultural systems replace specific areas of tree cover. It must also be remembered that even when land is not forested converting it to agricultural use may well result in more carbon emissions than can be avoided with the biofuel produced on that land for decades afterwards. The high ILUC-risk designation in the RED II is a useful tool to reduce the negative impacts of EU biofuel policy but is not on its own a solution to broader issues of indirect land use change.

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