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Environmental footprints show China and Europe's evolving resource appropriation for soybean production in Mato Grosso, Brazil

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Abstract

Mato Grosso has become the center of Brazil's soybean industry, with production located across an agricultural frontier expanding into savanna and rainforest biomes. We present environmental footprints of soybean production in Mato Grosso and resource flows accompanying exports to China and Europe for the 2000s using five indicators: deforestation, land footprint (LF), carbon footprint (CF), water footprint (WF), and nutrient footprints. Soybean production was associated with 65% of the state's deforestation, and 14–17% of total Brazilian land use change carbon emissions. The decade showed two distinct production systems illustrated by resources used in the first and second half of the decade. Deforestation and carbon footprint declined 70% while land, water, and nutrient footprints increased almost 30% between the two periods. These differences coincided with a shift in Mato Grosso's export destination. Between 2006 and 2010, China surpassed Europe in soybean imports when production was associated with 97 m^2 deforestation yr^{-1} ton^{-1} of soybean, a LF of 0.34 ha yr^{-1} ton^{-1}, a carbon footprint of 4.6 ton CO_2-eq yr^{-1} ton^{-1}, a WF of 1908 m^3 yr^{-1} ton^{-1}, and virtual phosphorous and potassium of 5.0 kg P yr^{-1} ton^{-1} and 0.0042 g K yr^{-1} ton^{-1}. Mato Grosso constructs soil fertility via phosphorous and potassium fertilizer sourced from third party countries and imported into the region. Through the soybean produced, Mato Grosso then exports both water derived from its abundant, seasonal precipitation and nutrients obtained from fertilizer. In 2010, virtual water flows were 10.3 km^3 yr^{-1} to China and 4.1 km^3 yr^{-1} to Europe. The total embedded nutrient flows to China were 2.12 Mtons yr^{-1} and 2.85 Mtons yr^{-1} to Europe. As soybean production

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grows with global demand, the role of Mato Grosso's resource use and production vulnerabilities highlight the challenges with meeting future international food security needs.

S Online supplementary data available from stacks.iop.org/ERL/9/074001/mmedia

Keywords: soybean, land footprint, water footprint, carbon footprint, deforestation, nutrients, Mato Grosso

1. Introduction

Environmental footprints are indicators of resource use and emissions across boundaries of consumption and production. Recently, the ecological footprint, water footprint (WF) and carbon footprint (CF) were combined into a 'footprint family' to collectively understand human appropriation of land, air, and water (Steen-Olsen *et al* 2012, Galli *et al* 2011). The ecological footprint, in global hectares (ha), expresses the human appropriated biocapacity related to a given level of consumption (Wackernagel and Rees 1996). The CF quantifies the amount of greenhouse gas emissions (g CO_2 equivalents) related to an activity or product (Hertwich and Peters 2009), while the WF estimates water consumptive use of a production or consumption activity (m^3 per activity or product; Hoekstra *et al* 2011).

Global estimates have shown that aggregate human activity and demand for resources and services approach 1.5 planets (Borucke *et al* 2013), with a global CF of 35 Gtons CO_2-eq in 2001 (Hertwich and Peters 2009), and a global 1996–2005 average WF of 9087 $Gm^3\ yr^{-1}$ (Hoekstra and Mekonnen 2012). Such estimates bring to light not only our overconsumption of the planet's renewable resources as illustrated by the ecological footprint, but the potential impacts on climate change (CF) and inefficiencies in water resource use (WF). Alongside these footprints are related resource flows that indicate the exchange of natural resource use and emissions across boundaries of production and consumption. Such flows have been employed to quantify water and nutrients involved in the production of traded goods (Hoekstra and Mekonnen 2012, Galloway *et al* 2007). Resource flows do not represent the physically traded resources, but rather provide relative metrics to evaluate resource use and pollution at the site of production and allow for the assessment of resource use efficiency of internationally traded products.

When applied to agricultural products, environmental footprints estimate the natural resources utilized by both producing and consuming countries, thus bringing to light externalities related to diets (Foley *et al* 2011). Between 1975 and 1995, the Ecological Footprint for land use increased from 10.3 to 12.6 Gha due to increases in consumption (Van Vuuren and Bouwman 2005); in 2008 it reached 5.4 Gha for cropland and grazing land alone (World Wildlife Fund 2012). In 2001, 20% of the world's CF was attributed to food production at close to 7 Gtons CO_2-eq (Hertwich and Peters 2009), while virtual water flows averaged 2320 $Gm^3\ yr^{-1}$ from the trade of agricultural and industrial products for the 1996–2005 period (Hoekstra and Mekonnen 2012). Resulting externalities have been studied using a life cycle approach (UNEP/SETAC Life Cycle Initiative 2005) to permit consideration of the natural resources used for agricultural products from 'field to fork' (Zaks *et al* 2009). Studies include the Ecological Footprint assessment of agricultural production in the Canadian Prairies (Kissinger and Rees 2009), the estimate of carbon emissions of tropical deforestation for soybean and pasture expansions (Karstensen *et al* 2013, Zaks *et al* 2009), and the WF of meat products (Hoekstra 2012, Galloway *et al* 2007). Other studies have also included nutrient flows in accounting for soil fertility and pollution from fertilizer application (Schipanski and Bennett 2012, Galloway *et al* 2007).

Brazilian soybean production is particularly of interest given the country's increasingly important role in the international trade of agricultural products in recent decades. Global soybean production rose from 143 to 227 Mtons between 2000 and 2010 among major producers (Argentina, Brazil, China and the US; FAOSTAT 2013). By 2010, Brazil had become the second largest producer of soybean in the world with 68.5 Mtons produced (FAOSTAT 2013), and is anticipated to be the world's leading soybean producer in 2014 (USDA-FAS 2014). Between 2000 and 2010, Brazil's soybean exports increased by 125%, with the share exported to China increasing from 16% in 2000 to 66% of total exports in 2010, and the share going to Europe dropping from 64% to 20% (FAOSTAT 2013, Aliceweb 2013) (figure 1).

Land use change in Brazil is strongly related to the expansion of soybean and pasture in the Cerrado and Amazon regions (Barona *et al* 2010). Deforestation and land-use intensifications make the Brazilian Amazon one of the largest sources of greenhouse gas emissions related to land cover and land use change (Galford *et al* 2013). The subsequent production and transport of export crops has made Brazil an important center for greenhouse gas emissions for exported goods (177.0 Gt CO_2-eq; Hertwich and Peters 2009), but also for flows of virtual water (112 $Gm^3\ yr^{-1}$; Hoekstra and Mekonnen 2012). In addition, Brazil is also an important exporter of nutrients for livestock feed used in live animal production systems when considering nitrogen (total of 2.1 Mtons of embedded N exported over 2000–02; Galloway *et al* 2007) and phosphorous consumed for crops, feed and livestock production (0.8 Mtons of P in 2007; Schipanski and Bennett 2012).

From 1990 to 2010, the contribution of Mato Grosso (figure 2) to Brazilian production increased from 3.1 Mtons yr^{-1} (15% of Brazilian soybean) to 18.8 Mtons yr^{-1} (27% of Brazilian production) IBGE 2013a). While the proportion of Mato Grosso soybeans shipped to Europe ranged from 11% to 29% of Brazilian soybean exports between 1997 and 2010, the portion exported to China increased

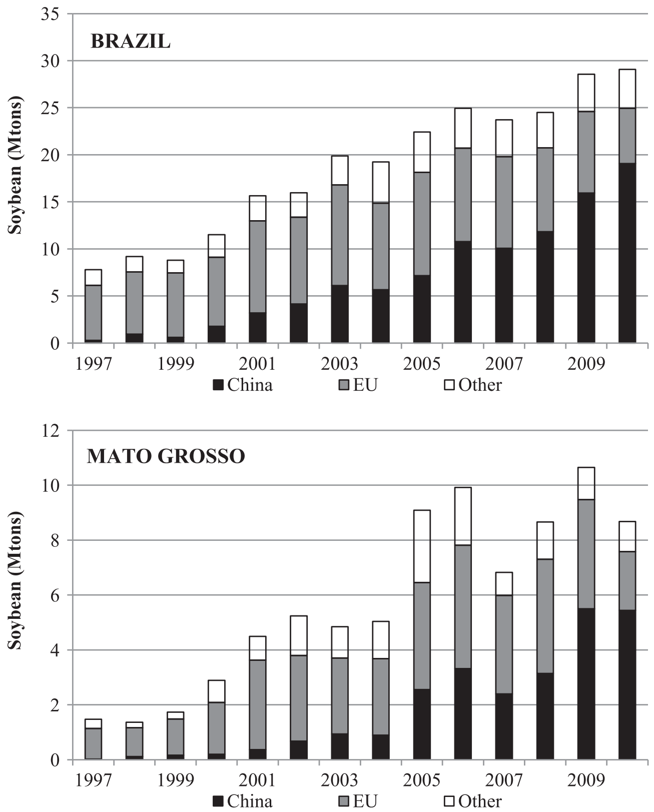

Figure 1. Brazilian exports of soybean to China, Europe and other countries (other) (top panel) with Mato Grosso's soybean exports between 1997 and 2010 (bottom panel) (Aliceweb 2013, FAOSTAT 2013, IBGE 2013a).

substantially from 0% to 31% (for 2009, 29% in 2010) (figure 1). Increased soybean production occurred through both agricultural extensification (conversion of natural environment into agricultural land), and intensification (increased production on existing agricultural lands) (Galford *et al* 2010). Agriculture has been expanding in Mato Grosso through conversion of rainforest and Cerrado/Cerradao savanna ecosystems since the 1990s (Barona *et al* 2010), with several changes in policies and increased dependencies on international markets also influencing land use change (Macedo *et al* 2012, Richards *et al* 2012). Local resource use for cropland expansion creates environmental externalities such as losses in biodiversity, ecosystem services or eutrophication, whose costs are often internalized by producers, thus causing a distortion in the real price of commodities (Galloway *et al* 2007). It is therefore important to quantify resource use and emissions across the supply chain to manage potential environmental impacts from supply and demand perspectives.

This study presents environmental footprints associated with the production of soybean in Mato Grosso, namely land, carbon, water and nutrients, and describes resource flows to China and Europe between 2000 and 2010 related to exports of the whole beans. Macedo *et al* (2012) reported a sharp reduction in deforestation in 2005–10, despite a continuous increase in soybean production, suggesting a possible change in the production system between the first and second half of the 2000s. This change also coincided with China's rise as a major soybean importer for the region, surpassing Europe. We therefore use environmental footprints and associated resource flows from soybean exports to further understand relationships between the producer and consumers in a decade of apparent change in soybean production system. Natural resources used in Mato Grosso's soybean production have previously been studied with respect to land use change (Macedo *et al* 2012) and associated carbon emissions (Galford *et al* 2010, 2011), carbon emissions allocation into soybean production and consumption (Karstensen *et al* 2013, Zaks *et al* 2009), as well as water consumptive use from rainfed agriculture (Lathuilliere *et al* 2012). To our knowledge, no research has yet integrated these footprints of production over a common time period for the purpose of understanding resource use related to soybean exports. This work provides context for Mato Grosso's role in the global food system in which importing nations are increasingly concerned with quantifying externalities related to agriculture (Ruvario *et al* 2011), particularly Europe (Steen-Olsen *et al* 2012).

2. Materials and methods

This study presents findings of resource use and emissions to land, air, water, and nutrient flows related to the production and export of soybean (*Glycine max*) in Mato Grosso for the 2000–10 decade translated into environmental footprints. Mato Grosso is located in Brazil's Central Western region and is home to three major biomes (figure 2): Amazon rainforest in the north, with a transitional Cerrado/Cerradao (savanna) extending to the Pantanal wetland in the south. We focus exclusively on soybean production and export in this study and do not consider the production of soybean oil and soy meal, which occur in Brazil as well as China and Europe, and would require additional resources (e.g. water, energy). These additional resources for products further down the supply chain would have to be considered in the country of production in order to derive a soy meal or soybean oil product environmental footprint. Also, indirect resources or emissions accounted in labor or off farm in the production of equipment and buildings are not included here. The agricultural production step ahead of such processing is considered the most resource intense stage of the entire life cycle of both products in Brazil (Milazzo *et al* 2013). Estimates show that an average of 40% of Brazilian soybean was exported as whole beans in 2000–10 (FAOSTAT 2013). Soybean that wasn't exported was processed into either oil for local consumption (18%) or soybean cake (>50%) (FAOSTAT 2013). We consider footprints and resource flows from exports to China and the European Union (hencefourth as Europe). LF, CF, WF and nutrient footprints are included, and described in more detail below. We also track deforested area as an additional indicator to be included with the land resources.

Virtual and embedded resources are differentiated as follows: virtual flows describe resources used which do not

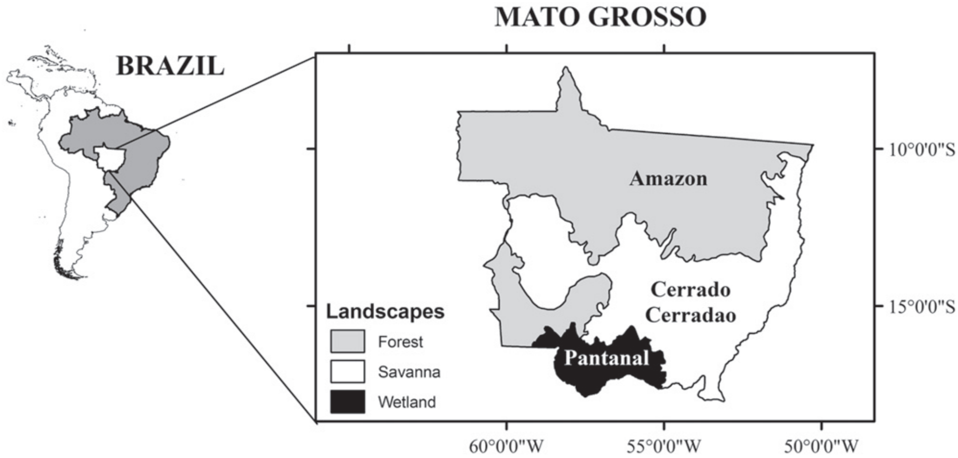

Figure 2. Map of Mato Grosso with its three main biomes: Amazon forest, Cerrado/Cerradao and the Pantanal wetland (Instituto Brasileiro de Geografia e Estatisticas (IBGE) 2013b).

comprise a physical part of the product, but were used or consumed as a component of soybean production (e.g. land use conversion, water consumptive use, CO_2 equivalents emitted during deforestation); embedded flows refer to resources which are an integral part of the product. This distinction is apparent in the case of nutrients (Galloway *et al* 2007) since elemental nitrogen (N), phosphorous (P) and potassium (K) are applied to the field as fertilizer (typically in an N-P_2O_5-K_2O formulation) and are taken up as part of the soybean plant. Moreover, 80% of the N assimilated by soybean is biologically fixed from atmospheric N via rhizobium associated with soybean roots (Smaling *et al* 2008). Thus embedded N refers to N contained in harvested soybeans, while virtual N refers to N utilized in soy production that doesn't become part of the harvested crop (e.g. N contents of crop residues, and N remaining in soil or leaching from soil). While water is also an integral part of the soybean, the plants' stored water content is small compared to the plant's consumptive use (Galloway *et al* 2007). We interpret the difference between virtual and embedded in terms of the production-consumption boundary where embedded resources physically cross this boundary. Embedded nutrients require assimilation by individual regions importing the raw material, while virtual resources represent Mato Grosso's consumption (for water) or load (fertilizer, CO_2 emissions, land) burdened by the region but which we indirectly allocated to importers.

Methodologies from previous studies were used and extended to include the 2000–10 decade (table 1) and described in more detail below. Analyses requiring information from the Brazilian Institute of Geography and Statistics (IBGE 2013a), such as data needed to calculate land, water and nutrient footprints, were carried out at the municipal scale using political units delimited in Lathuilliere *et al* (2012) because municipalities in Mato Grosso changed in size and number through the 2000s. As such, municipalities were aggregated into 104 municipal units whose number and size were constant within the study time period (Lathuilliere *et al* 2012). Resource flows were allocated as a percent of total resources used relative to the fraction of soybean exported to each country or region based on annual trade data available from Aliceweb (2013). We assigned full internalization of the footprints to importing countries as in Karstensen *et al* (2013).

2.1. Deforestation and land footprint

Two components of land use were considered in the present study. First, deforested land area (ha) associated with soybean cultivation was assessed after Karstensen *et al* (2013) for 1990–2010. Karstensen *et al* (2013) related deforestation area to soybean production based on Landsat satellite imagery from the Brazilian National Institute of Space Research (INPE 2013), in conjunction with the land use transition model from Rammankutty *et al* (2007). Initial land use following forest conversion was assumed to be 65% cropland and 35% pasture for Mato Grosso (Karstensen *et al* 2013). Second, we consider the LF without a carbon uptake from the land term as in Steen-Olsen *et al* (2012), since carbon uptake is accounted for in the CF (discussed below). While Steen-Olsen *et al* (2012) focus on consumption, we focus on the production side of the soybean supply chain and therefore consider area planted in soybean (IBGE 2013a) as the gross LF.

2.2. Carbon footprint

Deforestation and management practices were taken into account as the two main sources of greenhouse gas emissions in soybean production. Greenhouse gas emissions related to deforestation were obtained from Karstensen *et al* (2013). Biomass clearing through deforestation was assumed to be 70% burnt and 20% slash (Karstensen *et al* 2013) following agricultural practices and carbon stocks described by Galford *et al* (2010). The remaining biomass is typically separated into timber product (8%) and elemental C (2%) (Karstensen *et al* 2013). Carbon emissions are then allocated to soybean using a carbon cycle model when a biosphere-atmosphere flux is registered (Karstensen *et al* 2013). Such accounting

Table 1. Environmental footprints assessed in this study with associated methodology, assumptions, data sources, and references. Results from previous studies that did not cover the full 2000–2010 decade were extended.

Environmental footprint	Unit	Method	Assumptions	Data sources	References
Deforestation	ha	Remote sensing and land use change model based on Rammankutty *et al* (2007)	Initial share of Mato Grosso pasture to cropland is 35% and 65%	Landsat satellite imagery from INPE (2013)	Karstensen *et al* (2013)
Land	ha	Mato Grosso's soybean planted area	Expert surveys truly reflects agricultural production in non-census years	IBGE 2013a	This study
		Method following Rammankutty *et al* (2007)	70% deforestation is burnt, 20% is slash	Deforested area (see above) and updated stocks from Zaks *et al* (2009) and Galford *et al* (2010)	Karstensen *et al* (2013) Zaks *et al* (2009)
Carbon	g CO_2-eq	Emissions estimate based on Jungbluth *et al* (2007) and Nemecek and Kagi (2007)	All machinery is used the same way in the state. Machines used according to Swiss agricultural practices	Ecoinvent® database (Ecoinvent 2013)	Prudencio da Silva *et al* (2010)
Water	m^3	Crop modeling following guidelines from Allen *et al* (1998)	Soybean planted on October 1st or November 1st of each year. 10% leaching from N, 0.3% from P application for gray water calculation	Meteorological data (INMET 2013), fertilizer application information (see below)	Lathuilliere *et al* (2012) for green water and this study for gray water
Nutrients	tons	See equation (3)	Application rates are identical every year throughout Mato Grosso. Two formulations used: 2-20-18 (60%), 0-20-20 (40%)	ANDA (2010) and information from 13 farms in Mato Grosso. Embedded NPK based on Cunha *et al* (2010)	This study

does not include indirect land use change described by Barona *et al* (2010). Second, we consider emissions from machinery used in soybean production: sowing, fertilizer application, pesticide application and combine harvesting, noting that soybean is directly seeded into the stubble of the previous crop with no tillage employed in the study region. We used the average emissions of 33.4 kg CO_2-eq ton^{-1} soybean estimated by Prudencio da Silva *et al* (2010) for the Brazilian Central Western region using the Ecoinvent® database (Ecoinvent 2013). We then determined the annual emissions of machinery as a function of this emission rate and production numbers from IBGE (2013a).

2.3. Water footprint

The total WF was calculated as the sum of green WF and gray WF. The green WF represents soil moisture regenerated by precipitation and utilized for plant growth, while the gray WF is the water volume required to assimilate a pollutant load to ambient water quality standards (Hoekstra *et al* 2011). In this study, we do not consider the blue WF (e.g. the consumptive water use supplied via irrigation), as Mato Grosso's soybean production is almost exclusively rain-fed at present (Lathuilliere *et al* 2012).

Green water was calculated from actual evapotranspiration following equation (1) (Allen *et al* 1998),

$$ET_C = K_C ET_0, \quad (1)$$

where ET_C (mm day^{-1}) is the crop evapotranspiration, K_C is the crop coefficient for soybean which depends on the crop development cycle, and ET_0 (mm day^{-1}) is the reference evapotranspiration determined through meteorological parameters for the 2000 to 2010 period (INMET 2013, see online supplemental material equation S1 available at stacks.iop.org/ERL/9/074001/mmedia). Daily ET_C was calculated with variations in crop coefficient for the initial growing stage ($K_C = 0.56$), mid-season ($K_C = 1.50$) and harvest ($K_C = 0.50$) for a total development cycle of 126 days (Lathuilliere *et al* 2012). Final ET_C values were obtained from ten day averages of ET_0 (Allen *et al* 1998) before being compared to effective precipitation to derive soybean green water: if effective precipitation was less than ET_C, then green water was equal to precipitation; if effective precipitation was greater than ET_C, then green water was equal to ET_C as per Lathuilliere *et al* (2012). Finally, the sum of modeled green water for soybean harvests between 2000 and 2010 were related to harvested area and total soybean produced to derive an average green WF (m^3 ton^{-1} yr^{-1}) for the decade (see supplemental material). A sensitivity analysis was carried out according to planting assumptions defined below (section 2.5).

Gray WF was obtained following equation (2) (Hoekstra *et al* 2011),

$$\text{gray WF} = \frac{L}{c_{\max} - c_{\text{nat}}}, \tag{2}$$

where gray WF is calculated in $m^3\ yr^{-1}$ as a volume, and also per ton of soybean for each harvest year ($m^3\ ton^{-1}\ yr^{-1}$), L is the pollutant load ($g\ ton^{-1}\ yr^{-1}$), c_{max} is the ambient water quality standard ($g\ m^{-3}$) and c_{nat} is the natural concentration of the pollutant in the water body of interest ($g\ m^{-3}$). L was determined as a fraction of N and P application rates according to Hoekstra *et al* (2011). Pollution sources affecting the gray WF were determined from N and P fertilizer application; other agricultural inputs such as pesticides were not considered. We used 10% as the fraction of applied N fertilizer and 0.3% of applied P fertilizer allocated to ground and surface water as non-point source contamination (Hoekstra *et al* 2011, Riskin *et al* 2013a). Fertilizer application rates are described in the following section.

Gray water was then estimated as the volume of water needed to dilute the larger pollution load (N or P) considering the range of fertilizer application rates (Section 2.4). Therefore, the pollutant with the smallest volume is considered co-assimilated along with the contaminant with the largest pollutant load. The values of c_{max} (10 mg NO_3-N L^{-1} and 0.1 mg total P L^{-1}) were selected from water quality standards established under the Brazilian National Environmental Council (CONAMA) which specifies limits for class 2 water for human consumption, environmental conservation, recreation, irrigation and aquaculture (SEMA 2010a, 2010b, 2010c). The value for c_{nat} was assumed to be zero, which is known to underestimate the gray WF (Hoekstra *et al* 2011) and is preferred in order to allow gray WF estimates to be conservative.

Virtual water flows from total green and gray water consumed by soybean production then described the amount of local water resources appropriated for cultivation and assimilation of pollutant loads from fertilizer application. As described above, embedded water is small compared to virtual water (Galloway *et al* 2007) and therefore only virtual flows, representing indirect water use for the commodity, are discussed in this study.

2.4. Nutrient footprints

Virtual NPK and embedded NPK were determined as per Schipanski and Bennett (2012) and Galloway *et al* (2007) in equation (3):

$$F_v = F_a - F_e \quad with\ F_v \geqslant 0, \tag{3}$$

where F_v is the virtual fertilizer (N, P or K) remaining on the field (kg), F_a is the applied N, P or K fertilizer (kg) based on planted area and application rate (see below), and F_e is the embedded N, P or K (kg) based on production and elemental concentrations in Mato Grosso soybean (see below). Embedded NPK represents the nutrients physically traveling with the soybean which requires full assimilation by importers. Virtual NPK are the resources needed to grow the crop but that do not become embedded in the soybean. Just as in the virtual water case, the virtual NPK is related to local resources or the amount of NPK remaining in Mato Grosso's soil after soybean harvest but allocated to importers indirectly. This differentiation prevents double counting of NPK remaining in the soil and the amount physically crossing the production–consumption boundary through export. This separation also allows for an interpretation of potential environmental impacts experienced by the producing region (via virtual NPK) and consuming regions (via embedded NPK). We constrain virtual nutrients to be greater than or equal to zero, assuming that the plant has assimilated all applied fertilizer not lost to pollution. Eighty percent of N assimilated by the soybean occurs through biological N fixation (Smaling *et al* 2008) in which case a negative virtual N would imply a spurious impoverishment of the soil N (a detailed N soil balance is available in Smaling *et al* 2008). Similarly, lower K application rates (see below) could result in negative virtual K values. All fertilizer applied to soybean fields was assumed to be industrially derived as per Smaling *et al* (2008). Values obtained for P_2O_5 and K_2O application rates were divided by 2.3 and 1.2, respectively to determine elemental P and K application rates.

Total applied NPK was determined by top-down and bottom-up approaches following similar steps by Riskin *et al* (2013b). The top-down estimate focused on information from ANDA (2010) that reports combined fertilizer sales aggregated for all crops. Mato Grosso's share of total fertilizer consumption in Brazil's Central Western Region (as the states of Mato Grosso, Mato Grosso do Sul, Goias, and Federal District) was constant over 2000–10 at 56% (±1.5%) (ANDA 2010). We assumed that fertilizer sold to Mato Grosso had the same elemental NPK composition as that sold in the Central Western region for all years, except 2008 and 2010 for which Mato Grosso fertilizer sales were specifically reported by ANDA (2010). We then assumed a constant ratio of fertilizer used for soybean to total NPK consumed for 2010 (ANDA 2010), considering that 60% of fertilizer used for soybean was of the 2-20-18 formulation (N-P_2O_5-K_2O), and 40% was the 0-20-20 formulation (ANDA 2010). In the bottom-up estimate, average NPK application rates were obtained for the 2000–10 decade following Brazilian fertilizer consumption data (Cunha *et al* 2010, Heffer 2009, Fertistat 2007, FAO 2004) and information obtained from 13 farms combined with agricultural production information IBGE 2013a). These two approaches, when combined to planted soybean area from IBGE (2013a) gave a range of applications rates of 0–5 $kg\ N\ ha^{-1}$, 28–34 $kg\ P\ ha^{-1}$ and 39–62 $kg\ K\ ha^{-1}$. These ranges of fertilizer application rates were used in our analyses for all municipalities of Mato Grosso and were assumed to be constant within the study period.

Embedded NPK data (i.e. elemental concentrations in soybean) were obtained from Cunha *et al* (2010), where concentrations were reported to be 59.2 $g\ N\ kg^{-1}$, 5.5 g P $kg^{-1}$and 18.8 g K kg^{-1} of soybean leaving the field. The elemental concentrations were assumed to be constant within the decade for all soybean cultivated in Mato Grosso.

2.5. Sensitivity analysis

We performed a sensitivity analysis to provide uncertainty estimates for results when reporting environmental footprints and resource flows with the exception of LF obtained from IBGE (2013a). For the deforestation CF, we consider legacy emissions as described in Karstensen *et al* (2013). Most greenhouse gases emitted from deforestation would have occurred from biomass burning following forest clearing, with releases occurring in the year of deforestation. Legacy emissions include the biomass decay from slash, which is allocated to subsequent years following land clearing (Karstensen *et al* 2013). We used two planting dates for our annual green WF estimate (October 1st and November 1st) to account for differences in agricultural practices as per Lathuilliere *et al* (2012). We also included both extremes of the range of application rates derived from the top-down and bottom-up estimates to assess the differences in gray WF and virtual nutrient flows to China and Europe. Discussion of assumptions and comparisons to literature values can be found in the supplemental material.

3. Results and discussion

3.1. Deforestation and carbon emissions allocated to soybean production

Deforestation and carbon emissions related to soybean production decreased more than 70% between the 2001–05 and 2006–10 periods (figure 3, table 2), representing soybean production systems linked to agricultural extensification and intensification, respectively. Areas of new deforestation allocated to soybean occurring within the 2001–05 period totaled 2.92 Mha, but only 0.79 Mha in 2006–10. Carbon emissions from land use conversion declined from 1277 Mtons CO_2-eq to 373 Mtons CO_2-eq between time periods, and fell 80–90% between the years 2000 and 2010. Machinery emissions increased with the planted area between 2000 and 2010 (2.9 Mha yr^{-1} to 6.2 Mha yr^{-1}) but remained <1% of total carbon emissions in the decade. The 2000–10 decade was marked by deforestation policy initiatives and enforcement, as well as a 'soybean moratorium' in 2006 that excluded producers who clear rainforest for soybean production from the supply chain (Brando *et al* 2013, Macedo *et al* 2012). Currency exchange in early 2000s also favoured soybean expansion with a subsequent devaluation of the Brazilian Real slowing down deforestation (Richards *et al* 2012). Despite these factors, deforestation as quantified by INPE (2013) that is attributable to soybean production (0.41 Mha in 2000, 0.057 Mha in 2010) continued to represent 65% of total Mato Grosso deforestation at the end of the decade.

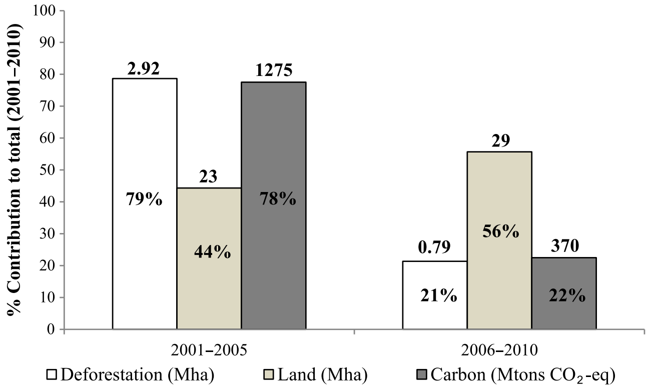

Figure 3. Contributions of deforestation, land and carbon emissions allocated to Mato Grosso's soybean production for the 2001–2005 and 2006–2010 periods expressed as a percent of total 2001–2010 results. Values above the histograms represent the 2001–2005 and 2006–2010 sums of deforestation (Mha), land (Mha) and carbon emissions (Mtons CO_2-eq) allocated to soybean in the respective periods.

Deforestation associated with soybean cultivation showed impacts on climate change as illustrated by the CF, but also suggest potential impacts on biodiversity and ecosystem services. Brazil's total carbon emissions from land use/land cover change were roughly equal in 2000 and 2005 at 1250 Mtons yr^{-1} (IBGE 2012) meaning that Mato Grosso's deforestation for soybean production contributed about 14% and 17% of total Brazilian land use change emissions in 2000 and 2005, respectively. Carbon emissions from land use change remain the largest contributor to Brazil's total emissions (IBGE 2012). In 2005, the number of extinct or endangered fauna and flora species reached 150 in the Amazon rainforest, >225 in the Cerrado, and >50 in the Pantanal wetland (IBGE 2012).

Ecosystem services related to climate and water cycling may also be impacted as a result of a loss of forest cover in the region (Lathuilliere *et al* 2012, Nepstad *et al* 2008), which may in turn impact the productivity of soybean agriculture. Simulations show that climate feedbacks from expanding agriculture further into the Amazon may reduce soybean productivity by 16–26% in 2041–50 (Oliveira *et al* 2013) such that the benefits of agricultural production are far outweighed by the costs to biodiversity and ecosystem services. While there are noted differences in deforestation and carbon footprint between 2001–05 and 2006–10, the above mentioned impacts on biodiversity and ecosystem services which would have been greater from soybean production based on extensification (2001–05) would likely carry over into the second half of the decade and into the future.

3.2. Resource use for Mato Grosso's soybean production

3.2.1. Water and nutrient resource use and potential sources of contamination. Water and nutrient resource use related to soybean production increased almost 30% between the 2001–05 and 2006–10 periods (table 2) along with

Table 2. Total resources used for the production of soybean in Mato Grosso for the 2001–2005 and 2006–2010 periods with amounts exported to China and Europe: deforestation, land, carbon (nonleg—without legacy emissions; leg—including legacy emissions), water (average of planting assumptions and fertilizer application); average virtual phosphorous (P_v), potassium (K_v); embedded nitrogen (N_e), phosphorous (P_e) and potassium (K_e).

	Deforestation (Mha)	Land (Mha)	C emissions (Mton CO_2-eq) nonleg—leg	Water (km^3)	P_v (Mtons)	K_v (Mtons)	N_e (Mtons)	P_e (Mtons)	K_e (Mtons)
Total production									
2001–05	2.92	22.8	1269–1285	127	0.34	0.081	3.940	0.366	1.25
2006–10	0.79	28.6	370–376	162	0.42	0.094	5.020	0.467	1.60
Exports[a] to China									
2001–05	0.212	1.8[b]	92.6–93.6	10.4	0.028	0.0067	0.321	0.0298	0.102
2006–10	0.163	6.6	77.4–78.7	37.7	0.096	0.019	1.17	0.109	0.372
Exports to Europe									
2001–05	0.702	5.4	305–309	30.2	0.080	0.018	0.937	0.0871	0.298
2006–10	0.195	6.2	90.9–92.4	35.0	0.091	0.019	1.086	0.101	0.345

[a] Exports to China and Europe are for soybean as whole bean only, not soybean meal or oil.
[b] Land values reported for China and European exports are based on harvested area.

intensification of the soybean production system. Water use, as total of green and gray water, increased 28% from 127 km^3 for 2001–05 compared with 162 km^3 for 2006–10. Total water resources consumed were dominated by green water (106 km^3 for 2001–05 versus 135 km^3 for 2006–10), while gray water represented approximately 17% of total water consumptive use (averaging 21.2 km^3 to 26.7 km^3 for 2001–05 and 2006–10, respectively). Virtual P increased 24%, from 0.34 Mtons to 0.42 Mtons and virtual K by 16%, from 0.081 Mtons to 0.094 Mtons between the early and later five-year periods. In 2001–05 and 2006–10, embedded N increased from 3.9 Mtons to 5.0 Mtons with no associated virtual N. Virtual P (0.34 Mtons to 0.42 Mtons per 5 year period) was similar to embedded P (0.36 Mtons to 0.47 tons per 5 year period) while embedded K (1.25 Mtons to 1.60 Mtons per 5 year period) was over ten times larger than virtual K (0.081 Mtons to 0.094 Mtons per 5 year period).

The increasing virtual P illustrates the increasing amount of nutrients remaining in the state of Mato Grosso with possible impacts to water bodies. Loss of applied P is known to cause eutrophication of water bodies over time in areas where P is not strongly bound to the soil. Local soil conditions in the study area, however, have a very high P fixing capacity thus largely preventing it from reaching aquatic environments (Riskin *et al* 2013a, 2013b, Schipanski and Bennett 2012). Monthly water quality campaigns were performed in 2007–09 in the Amazon River, Paraguay River and Tocantins-Araguaia River basins (74 sites in total) (SEMA 2010b). Total P measurements exceeded the CONAMA limit of 0.1 mg total P L^{-1} on 40 out of 705 occasions, averaging 0.30 mg P L^{-1} ($n = 40$; sd = 0.25). Finally, there is evidence of ongoing accumulation of P in soils (Riskin *et al* 2013b, Schipanski and Bennett 2012), which eventually could lead to saturation of P in the long term with consequential eutrophication, although results to date have not shown increases in P loading from on farm experiments (Brando *et al* 2013).

3.2.2. Non-renewable fertilizer inputs. In addition to concerns over contamination of water bodies, the argument that agricultural trade results in exports of soil fertility has been made (Galloway *et al* 2007). However, this claim may not apply to Mato Grosso's Cerrado soils since its fertility is mostly derived from fertilizer inputs. Placing soybean production within the context of the fertilizer supply chain therefore provides additional insight into resource use for soybean exports, especially since Brazil is the fourth largest buyer of fertilizer in the world, with 10.1 Mtons of NPK fertilizer purchased in 2010 (ANDA 2010). Devaluation of the Brazilian Real through the early 2000s led to an increase in the costs of production, especially fertilizer, which increased in price from US\$ 55.3 ha^{-1} in 2000–01 to US\$ 180 ha^{-1} in 2009–10 (adjusted for inflation) for typical macronutrients (Agroserra 2001, IMEA 2010). Brazil's demand for fertilizer raw materials between 2000 and 2010 increased 6.6% per year, with 60–80% of materials imported for domestic fertilizer production (Amaral and Peduto 2010). In 2008, Brazil imported 4% of the world's N fertilizer, 7% of P and 14% of K (Amaral and Peduto 2010). For 2005–10, P fertilizer was mainly sourced from Israel (35%), Tunisia (19%), the E.U. (17%) Morocco (10%), and other international partners (19%) (UN Comtrade 2012). K fertilizer during this period was mainly sourced from Canada (27%), Belarus (22%), the E.U. (20%), Russian Federation (15%), Israel (12%), and other countries (4%) (UN Comtrade 2012). Brazil's dependence on K imports is particularly critical, with only 8% of K fertilizer use supplied by national production (IBGE 2012).

Mato Grosso fertilizer use ranks among the highest in Brazil for P and K fertilizer sales. In 2010, fertilizer sales in Mato Grosso were equivalent to application rates of 58.9–82.3 kg P ha^{-1} of cultivated area, and 72.9–94.3 kg K ha^{-1} (IBGE 2012). High P application rates and high K use efficiency (defined as the ratio of embedded to applied K fertilizer, see Section 3.3.2. below) suggest that Mato Grosso's soybean production works as a node for nutrient transfer from soil to soybean. Most applied K is

Table 3. Average environmental footprint per ton of soybean produced in Mato Grosso for the 2001–2005 and 2006–10 periods: deforestation, land, carbon (as average of legacy and non-legacy emissions), water (average of planting assumptions and fertilizer application range), virtual phosphorous (P_v) and potassium (K_v) (considering fertilizer application range). Standard deviations are shown in brackets.

Periods	Deforestation (m^2 yr^{-1} ton^{-1})	Land (ha yr^{-1} ton^{-1})	C emissions (ton CO_2-eq yr^{-1} ton^{-1})	Water (m^3 yr^{-1} ton^{-1})	P_v (kg yr^{-1} ton^{-1})	K_v (g yr^{-1} ton^{-1})
2001–2005	455 (115)	0.34 (0.02)	19.9 (4.8)	1908 (15)	4.0 (0.45)–6.0 (0.55)	0.0042 (0.12)–1.1 (0.49)
2006–2010	97 (63)	0.34 (0.02)	4.6 (2.8)			

exported out of Mato Grosso through the commodity trade since embedded K was larger than virtual K. Within this context of international trade, soybean production in Mato Grosso and exports remain strongly tied to non-renewable resource extraction in other countries, noting that global reserves of phosphate rock for P and potash for K are believed to be declining due to long regeneration time scale (Obersteiner *et al* 2013, Schipanski and Bennett 2012). This strong dependency also creates an important node for spikes in regional soybean price and costs of production due to fertilizer price fluctuations that strongly depend on oil supply and local fertilizer production capabilities (Amaral and Peduto 2010).

3.3. Mato Grosso soybean exports to China and Europe

3.3.1. China and Europe's land use appropriation through soybean imports. Of the major producers of soybean, Argentina and Brazil are the only countries that have expanded production in recent years, with Brazil's agriculture representing 30.9% of Brazil's land base in 2000 and 32.3% in 2010 (FAOSTAT 2013). Since much of production is bound for the export market, the LF related to exported soybean is attributable to China and Europe as the importers seeking to meet their soybean raw material demands. The dynamic for increasing LF of soybean production (0.34 ha ton^{-1} in the decade, table 3) has changed between 2000 and 2010 (table 2) as the soybean production system evolved from a system based on extensification (2001–05) versus intensification (2006–10). Deforestation allocated to soybean declined between the 2001–05 and 2006–10 periods as Chinese soybean imports slowly surpassed European imports (figure 1). The area of forest impacted by soybean exports decreased from 0.212 Mha to 0.163 Mha per period for China, and from 0.720 Mha to 0.195 Mha per period for Europe due to declining deforestation post-2004. Reduced deforestation resulted in a 10% decrease in carbon emissions for China (93.1 Mtons to 78.1 Mtons CO_2-eq per 5 year period) and 70% decrease for Europe (307 Mtons to 91.7 Mtons CO_2-eq per 5 year period). Virtual resource flows related to soybean exports increased for China and remained roughly steady for Europe, as did the quantities of soybean imported by Europe (figure 1, bottom panel). By 2010, China had surpassed Europe in resource appropriation with twice the amount of resources flowing from Mato Grosso to China as compared to Europe. Embedded nutrients increased over time with increases in imports by China and steady values for Europe.

China's increase in soybean imports from Mato Grosso post-2005 coincided with a sharp drop in deforestation and CF (table 3) as soybean production intensified. Deforestation per ton of soybean harvested was 455 m^2 ton^{-1} for 2001–05 and 97 m^2 ton^{-1} for 2006–10. At the same time, the CF decreased from 19.9 tons CO_2-eq ton^{-1} (on average resulting from biomass burning and subsequent emissions from decaying slash) to 4.6 tons CO_2-eq ton^{-1} between the two 5 year periods. In other words, Chinese soybean importers indirectly had smaller externalities on global climate change, Mato Grosso biodiversity and ecosystem services compared to European importers earlier in the decade when the soybean production system was more focused on agricultural extensification. The rise in China's imports of soybeans coincided with more effective deforestation policies and enforcement, yet China's virtual resources imports, on the basis of planted area in Mato Grosso and exported volumes, remained larger than Europe's in 2010.

3.3.2. Efficiency in resource use through soybean trade. The average green WF during the study period was 1590 m^3 ton^{-1}, which is lower than the world soybean average (2037 m^3 ton^{-1}) and soybean produced in China (2428 m^3 ton^{-1}), but within the range of some European countries (Italy: 1177 m^3 ton^{-1}; Germany: 1948 m^3 ton^{-1}) (Mekonnen and Hoekstra 2011). Nutrient exports embedded in soybean exports were found to increase over the 2000–10 period with the largest NPK values in 2010 (table 2). We assessed the phosphorous use efficiency for phosphate in fertilizer (PUE) and potassium use efficiency (KUE), defined as the ratio of embedded nutrients to applied P and K fertilizer. Averages over the study period were 0.48–0.58 for PUE and 0.90–1.43 for KUE based on fertilizer application rates. Average soybean PUE values in this study were similar to values previously reported for Brazil in 2002 (0.57) (Schipanski and Bennett 2012), but larger than the PUE for a farm in Northern Mato Grosso (0.39) (Riskin *et al* 2013a). The average KUE >1 suggests that essentially all of the applied K is taken up by the plant and exported. Additional K sources not quantified here may come from decomposing crop biomass left on the field during no-till agricultural practices (Bertol *et al* 2007).

There is little or no averted water contamination through soybean trade due to the high P fixing capacity of soils, as discussed by Schipanski and Bennett (2012). However,

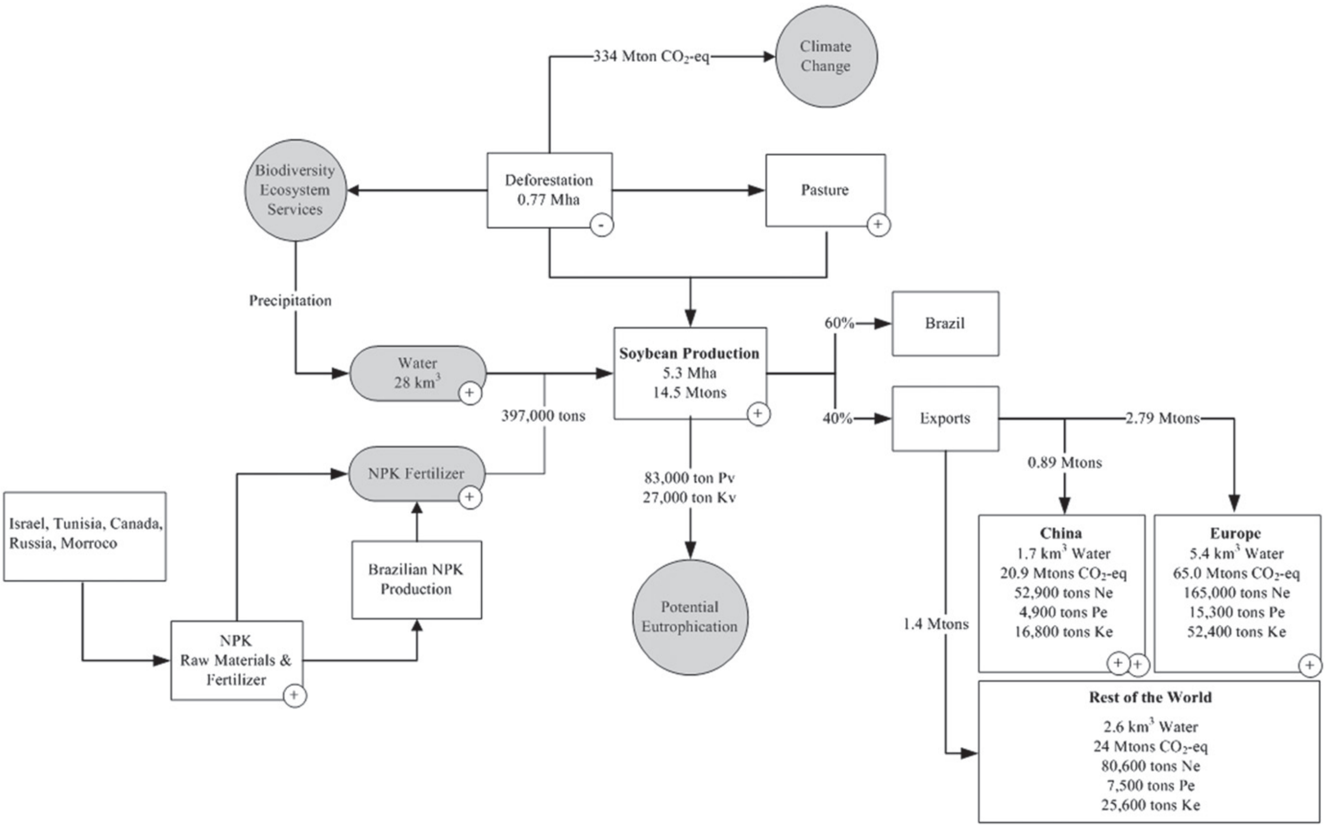

Figure 4. Resource use summary for Mato Grosso soybean production using 2004 as a reference year, with differences in pressures post-2004 (+ increases; − declines) (FAOSTAT 2013, ANDA 2010). Ne, Pe, Ke are embedded nitrogen, phosphorous and potassium respectively; P_v, K_v are virtual phosphorous and potassium. Exports to China, Europe and the Rest of the World only concern soybean as whole beans.

embedded nutrient exports may represent a management issue for countries importing soybean or derived products. Soybean produced in Mato Grosso is exported as whole bean or meal (not considered in detail in our study) with implications for waste management and contamination issues from human consumption or animal production in China and Europe. The argument of resource use efficiency through international trade has been made in the case of water resources and nutrients. For instance, China's virtual water imports via soybean from Mato Grosso illustrate water resource use efficiency through international trade, also depicted in the Global Virtual Water Trade Network where 101 $km^3\ yr^{-1}$ of virtual water was traded between South America and Asia, and 77 $km^3\ yr^{-1}$ to Europe in 2007 (Dalin *et al* 2012). Similarly, Schipanski and Bennett 2012 note that low crop PUE countries can import crops from countries with greater PUE. The policy of virtual water imports however remains contested (Liu and Savenije 2008). Domestic soybean production in China was relatively constant within the study period at 15.4 Mtons of soybean in 2000 and 15.1 Mtons in 2010 (FAOSTAT 2013). China and Europe remain the largest importers of soybean (FAOSTAT 2013), much of which is used for the production of animal feed within importing countries (Nepstad *et al* 2008). The use of water and nutrients abroad for Chinese soybean production remains an important question in the country's agricultural policy as depicted in the supply side of soybean production in the context of Mato Grosso's land use dynamics described here.

Rather than exporting native soil fertility, Mato Grosso is virtually exporting its water resources, photosynthetic capacity and the N fixing capabilities of soybean to China and Europe through the product. As soybean production in Mato Grosso is currently rain-fed (e.g. production relies almost exclusively on soil moisture derived directly from local precipitation without supplemental irrigation), the interannual variation in rainfall can affect yields. We calculated the average green WF for soybean as 1450 $m^3\ ton^{-1}$ for a planting date of November 1st, versus 1750 $m^3\ ton^{-1}$ for soybean with a planting date October 1st. This difference in green WF reflects the possible water stress that the plant may undergo which could greatly impact yields, especially during the development and flowering stages. Macedo *et al* (2012) identified that 22% of increased soybean production in the state was due to yield increases related to land and water management practices. Moreover, simulations on climate feedbacks of deforestation show a potential connection with precipitation such that soybean yield may become affected (Oliveira *et al* 2013).

4. Conclusions

This study presented a synthesis of environmental footprints for the production of soybean in Mato Grosso, Brazil, an important global food security node upstream of the animal protein supply chain (figure 4). The estimate of resource use and related flows is a key step in assessing externalities which are often not included in commodity prices, but whose related impacts must be considered within the 'planetary safe operating space' (*sensu* Rockstrom *et al* 2009). Soybean production represented 65% of deforestation in Mato Grosso during the 2000–10 study period, and 14–16% of total Brazilian land use related carbon emissions despite a close to 70% decline between the 2001–05 and 2006–10 periods. The decline in deforestation suggested a shift in soybean production system from one based on extensification (agricultural expansion into natural ecosystems) to one based on intensification (increase in yield on available land) with differences in resource use and allocation to trade partners with China replacing Europe as the main soybean importer.

Over the study decade, soybean production required ever larger amounts of water and fertilizer inputs related to increases in planted areas and average yields, both of which are key to ensuring adequate total crop production. On the one hand, current practices of soybean production in Mato Grosso depend on rainfall, a free resource whose magnitude and timing can affect yields. On the other hand, soybean production requires inputs that strongly depend on international suppliers to construct soil fertility in Mato Grosso before exporting nutrients abroad. Our estimated environmental footprints suggest potential regional impacts on climate, biodiversity, ecosystem services, and a possible incremental soil P saturation that could increase the risk of eutrophication in the long term.

Supply side footprint assessment and resource flows can help reconcile policy actions that seem to have opposing goals, such as the inclusion of the environment in agricultural policy but its exclusion in trade policy (Wurtenberger *et al* 2006), or actions led by countries supporting deforestation reduction initiatives and their dampening effects from increased demand of soybean and meat products (Karstensen *et al* 2013). Such pressures will only increase with affluence, which has shown to contribute to further land displacement (Weinzettel *et al* 2013) as illustrated here by China's continuing growth in demand for Mato Grosso's soybean. Options for future soybean production will have to consider the trade-offs between agricultural extensification and intensification illustrated by the combined footprint analysis and an associated impact assessment to address costs and benefits of future food production strategies.

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