Complementarity in mid-point impacts for water use in life cycle assessment applied to cropland and cattle production in Southern Amazonia

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Abstract
Southern Amazonia has been the center of a large expansion of cropland and cattle production through land use change in both Amazon and Cerrado biomes. While this expansion has had noted impacts on the regional water cycle, little information is currently apparent in life cycle impact assessments of both cropland and cattle products. This study applies existing models to quantify the mid-point impacts of water consumption and land occupation of cropland and cattle following distinct production systems that rely on agricultural extensification and intensification (cropland irrigation, and increased pasture productivity). We focus on both terrestrial and aquatic flows to highlight complementarity in current impact assessment models, expressed through the following impact categories: Water Scarcity Footprint, Terrestrial Green Water Flows, Precipitation Reduction Potential, River Blue Water Production, Groundwater Recharge Potential, and Runoff Reduction Potential (introduced here). Results show conditional changes in magnitude and sign of potential impacts when comparing rain-fed to irrigated cropland, particularly in the Cerrado biome. Cropland irrigation can increase atmospheric and terrestrial water flows as expressed through Precipitation Reduction Potential (~95 m³ ha⁻¹), River Blue Water Production (299 m³ ha⁻¹), or Groundwater Recharge Potential (215 m³ ha⁻¹). Moreover, increased pasture productivity led to an overall decrease in mid-point impacts of cattle production on the water cycle. While this study provides additional insight into the effects of cropland and cattle production systems in Southern Amazonia, our results also highlight the complementarity of existing mid-point impacts towards a better representation of freshwater use in life cycle assessment.

1. Introduction

The landscape of Brazil’s central Western region has changed significantly since the 1990s following a rapid rise in the production of agricultural commodities (Barona et al., 2010; Dias et al., 2016; Macedo et al., 2012; Simon and Garagorry, 2006). Today, the state of Mato Grosso (Fig. 1) is the largest producer of both soybean (Glycine max) and beef in Brazil, and has mostly relied on the expansion of cropland and pasture in both Amazon and Cerrado biomes to reach national and international production rankings (FAOSTAT, 2017).

The appropriation of natural resources for this expansion has grown through land use change (Lathuillière et al., 2014) with noted environmental impacts which include the loss of biodiversity (Chaplin-Kramer et al., 2015), changes in surface (Silvério et al., 2015) and stream temperatures (Macedo et al., 2013), as well as degradation of terrestrial ecosystems due to a reduction in regional precipitation that may be capable, in part, of tipping the Amazon biome into a “savannization” process (Davidson et al., 2012; Silvério et al., 2013). Water resources in Amazonia are particularly at risk of further degradation from cropland and pasture expansion, as well as through dam construction and mining activities (Castello and Macedo, 2016). In parallel, additional disruptions to the water cycle have affected regional evaporation recycling into precipitation (Lathuillière et al., 2016b), which may affect future rain-fed
agricultural production and hydropower generation (António Sumilu et al., 2017; Oliveira et al., 2013; Stickler et al., 2013).

Lathuillère et al. (2016b) defined five possible expansion options for the region which include agricultural expansion into natural ecosystems or current pastureland, and agricultural intensification using rainwater harvesting, irrigation, or by improving water vapor flows through an increase in crop transpiration over soil evaporation. Each option carries distinct uses of water resources that closely follow land management and the resulting partitioning of precipitation into blue and green water (Lathuillère et al., 2016b). The labeling of water into blue and green water resources has been central to an ecohydrological paradigm that closely links land to water resource management: blue water resources represent the liquid water typically supplied by surface and groundwater stocks, while green water resources are exclusively the moisture in the unsaturated zone of soils that is depleted through ET and regenerated by precipitation (Falkenmark and Rockström, 2006). Differences in land use for agricultural products in Southern Amazonia therefore entail different potential environmental impacts as a result of precipitation partitioning which merits further attention in life cycle assessment (LCA).

Recent methodological advances focusing on water use in LCA have addressed differences in the cause and effect impact pathways of the consumption of blue and green water, particularly as they relate to ET (Lathuillère et al., 2016a; Miña i Canals et al., 2009; Núñez et al., 2013; Quinteiro et al., 2015; Ridoutt and Pfister, 2010). Some methods have focused on the effects of water consumption on scarcity: Ridoutt and Pfister (2010) assessed changes in blue water flows as a result of changes in ET on the land, while Núñez et al. (2013) considered a ratio of water consumption to availability similar to what has been defined by the Water Footprint Network (Hoekstra et al., 2011). Other methods have proposed specific mid-point impacts reflecting changes in precipitation partitioning and the distribution of green and blue water resources at the land surface (Table 1): Quinteiro et al. (2015) introduced the Terrestrial Green Water Flows (TGWF) and River Blue Water Production (RBWP) mid-point impacts to describe changes in the respective flows to the atmosphere and to liquid stocks as a result of land use. Similar to TGWF, Lathuillère et al. (2016a) proposed the Precipitation Reduction Potential (PRP) impact as a land transformation and occupation impact following the United Nations Environment Life Cycle Initiative guidelines (Koellner et al., 2013), which could be considered complementary to Groundwater Recharge Potential (GWRP) described by (Saad et al., 2013) (Table 1).

The application of LCA for cropland and beef has gained interest in recent years with a focus on greenhouse gases for Brazilian soybean (e.g., Castanheira and Freire, 2013; Prudêncio da Silva et al., 2010) or beef (Cardoso et al., 2016; Cerri et al., 2016; Dick et al., 2015) with less attention on water resources, particularly in Southern Amazonia. The inclusion of water use in LCA for cropland and derived products has been applied in the context of irrigation (e.g., Pfister et al., 2009), and more recently land use (Cao et al., 2015; Lathuillère et al., 2017; Quinteiro et al., 2014). Water use for beef or beef products has focused on both life cycle inventory (LCI) (Kannan et al., 2017; Peters et al., 2010), as well as Life Cycle Impact Assessment (LCIA) (Harding et al., 2017; Payen et al., 2018; Ridoutt et al., 2012), but with little focus on the effects of land occupation on the water cycle using the above described impact pathways.

This study focuses on water consumption and land occupation impacts of cropland (which includes soybean) and cattle production in Southern Amazonia with the goal of comparing agricultural land use options in the region using current life cycle impact assessment (LCIA) methods. We follow the four phases of a LCA described in ISO 14044 (ISO, 2006), and also consider the water scarcity footprint following ISO 14046 (ISO, 2015) to highlight competition over blue water resources (Boulay et al., 2018): (1) Goal
and scope definition, (2) life cycle/Water Footprint (WF) Inventory, (3) life cycle/WF Impact Assessment, (4) interpretation. Results are aimed at providing input on land use options for the production of the two most common products in the region, while at the same time comparing and contrasting available LCIA methods that focus specifically on green and blue water partitioning on land (Table 1).

2. Methodology

2.1. Goal and scope definition, and functional units

The goal of the study is to compare land use for cropland (including soybean) and cattle production practices in Southern Amazonia (Fig. 1) using LCIA methods that focus specifically on green and blue water partitioning on land (Table 1). We compare extensification and intensification production systems for cropland and cattle based on possible choices of land and water resources which include the use of irrigation as well as an increase in pasture productivity in both Amazon and Cerrado (i.e., savanna) biomes (Fig. 2). We focus specifically on water quantity with mid-point impacts linked to water consumption (as the Water Scarcity Footprint (WSF) following ISO 14046 terminology (ISO, 2015)) and land occupation. According to the ISO 14046 standard, a WSF is the result of a LCA focused specifically on potential impacts due to blue water consumptive use, and has been expressed as a function of the level of water scarcity in a basin (Boulay et al., 2018). The geographic scope is limited by the boundaries of the Xingu Basin within the state of Mato Grosso (Fig. 1) while considering production practices averaged for Mato Grosso’s Amazon and Cerrado biomes. The system boundaries are the cradle-to-farm gate production of crops and cattle in 2014–2015.

For cropland, we consider rain-fed and irrigated systems as two options of interest for production in the region that respectively represent extensification and intensification options (Fig. 2). These systems are an integral part of soybean production as the primary crop of interest and the main driver of land use change in Mato Grosso (Barona et al., 2010; Dias et al., 2016; Macedo et al., 2012; Spera et al., 2016b). As such, we consider two rotations: a rain-fed soybean-maize (Zea mays) rotation, and an irrigated soybean-rice (Oriza sativa)-bean (Phaseolus vulgaris) rotation based on field measurements (Lathuillière et al., 2018b). In rain-fed systems, soybean is typically planted at the beginning of the wet season

![Fig. 2. Scenarios for land use considered in this study for estimating the mid-point impacts of cropland and cattle production systems (cradle-to-farm gate). Scenarios include cropland extensification on natural vegetation (NV) or pasture, and differences in pasture productivity for cattle: low productivity supporting 450 kg live weight (LW), and high productivity supporting 675 kg LW.](image-url)
(October—November) with maize immediately planted following the soybean harvest (February—March). In the irrigated system, soybean is planted at the end of the dry season (September) to allow for an earlier harvest to benefit a rice harvest in the wet season (April), prior to planting a triple crop of bean in the dry season (fully irrigated as described in Lathuilière et al. (2018b)). The rice in this rotation is produced as upland rice, and not as paddy rice. These options reflect the commonly used double cropping system [Spera et al., 2014], and an irrigation option which allows for a crop to be planted in the dry season (bean in this case).

For cattle, we consider the production system of the Nelore species (Bos taurus indicus, most common in Mato Grosso), focusing on the differences in pasture productivity as an indicator for an increase in cattle density, and also consider water consumed by animals in small farm reservoirs as described in Lathuilière (2018).

2.2. Life cycle inventory

We consider three life cycle inventories (LCI) elementary flows for each production system based on blue water consumption, and changes in blue and green water from land occupation (Table 1): one for the WSF (blue water consumed as described by the WF inventory), one based on land area (A), one based on ET (as effective net green water, NGWeff, described by Quinteiro et al. (2015), see equation (1)). Blue water uses for cropland and cattle production are in competition with other human and ecosystem uses in the basin. Cropland and cattle production are therefore susceptible to deprive these users of water (Boulay et al., 2018), which is expressed in LCI by the WF inventory (ISO, 2015). The WF inventory includes blue water consumed for irrigation (assuming all irrigation becomes evaportranspiration, or ET), drinking water for cattle provided by small farm impoundments, and the volume of water evaporated from these reservoirs. Blue water consumption was based on previous results for both cropland (Lathuilière et al., 2018b) and cattle (Lathuilière, 2018) (Table 2). Blue water was allocated to dry season irrigation of bean (118 mm over the season) as well as water consumption by cattle. The latter was estimated over the course of the animal’s development cycle of 48 months (Lathuilière, 2018) and divided by four to obtain mean annual consumption. Given the lack of information on small farm reservoir water balances in the region, we assumed that both cattle drinking and evaporation from the reservoirs diminished streamflow or groundwater recharge with potential impacts on future water availability. In 2014, the portion of the Xingu Basin within Mato Grosso (Fig. 1) contained 9463 ha of small farm impoundments detected using remote sensing (Lathuilière et al., 2018a). Drinking water for cattle was based on the development stage of the animal and averaged 40.5 × 10^-3 m³ y⁻¹ (kg LW)⁻¹. There were 0.141 km³ y⁻¹ of small reservoir evaporation in the basin in 2014–2015 for a total live cattle population of about 3.5 million (Lathuilière et al., 2018a), which we attributed to the total cattle live weight based on mean cattle weight in respective development phases (95 kg LW cattle⁻¹ for calves, 266 kg LW cattle⁻¹ for mid-life cattle, and 429 kg LW cattle⁻¹ at the end-of-life) (Lathuilière, 2018). This calculation provided a mean allocation of small farm reservoir evaporation attributed to the live cattle herd of 0.16 m³ y⁻¹ (kg LW)⁻¹.

Current impact assessment methods involving water partition from land occupation (Table 1) either include land area or ET (NGWeff) in the LCI phase (see below). Both methods for deriving GWRP and PRP use land occupation (A) ha as the LCI, as well as the Runoff Reduction Potential (RRP) impact introduced in this study (see Section 2.3). Land occupation is based on the annual occupation of land from the crop rotations containing soybean in both rain-fed and irrigated systems. We translate pasture consumed by cattle into hectares of land in Mato Grosso based on an averaged male and female 48-month animal development cycle (divided by four to obtain mean annual consumption), and considering low and high pasture productivity scenarios (Table 2). In the case of TGFW and RBWP, Quinteiro et al. (2015) introduced ET in the LCI as defined by effective net green water (NGWeff, m³ ha⁻¹ y⁻¹).

\[ \text{NGWeff} = \text{ET}_{LU,eff} - \text{ET}_{NV,eff} \]  
(1)

where \( \text{ET}_{LU,eff} \) (m³ ha⁻¹ y⁻¹) is the effective ET of the current land use (cropland or pasture), \( \text{ET}_{NV,eff} \) (m³ ha⁻¹ y⁻¹) is the effective ET of the natural vegetation (Amazon or Cerrado). Both \( \text{ET}_{LU,eff} \) and \( \text{ET}_{NV,eff} \) are calculated following Quinteiro et al. (2015).

\[ \text{ET}_{i,eff} = \text{ET}_{i} - \text{ET}_i \epsilon_r \]  
(2)

where \( \text{ET}_i \) is the ET of a land use i, and \( \epsilon_r \) (0.22, dimensionless) is the basin internal evaporation recycling ratio (Berger et al., 2014)

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### Table 2

**Input parameters used in this study.**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
<th>Amazon</th>
<th>Cerrado</th>
<th>Unit</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Precipitation</td>
<td>P</td>
<td>2096</td>
<td>1369</td>
<td>mm y⁻¹</td>
<td>Rodrigues et al. (2014)</td>
</tr>
<tr>
<td>ET of natural vegetation</td>
<td>ET&lt;sub&gt;NV&lt;/sub&gt;</td>
<td>1099</td>
<td>817</td>
<td>mm y⁻¹</td>
<td>Lathuilière et al. (2012); Oliveira et al. (2014)</td>
</tr>
<tr>
<td>Cropland ET, rain-fed</td>
<td>ET&lt;sub&gt;LU&lt;/sub&gt;</td>
<td>801</td>
<td></td>
<td>mm y⁻¹</td>
<td>Lathuilière et al. (2018b)</td>
</tr>
<tr>
<td>Cropland ET, irrigated</td>
<td>ET&lt;sub&gt;LU&lt;/sub&gt;</td>
<td>982</td>
<td></td>
<td>mm y⁻¹</td>
<td>Lathuilière et al. (2018b)</td>
</tr>
<tr>
<td>Pasture ET</td>
<td>ET&lt;sub&gt;LU&lt;/sub&gt;</td>
<td>794</td>
<td></td>
<td>mm y⁻¹</td>
<td>Lathuilière (2018)</td>
</tr>
<tr>
<td>Pasture productivity (high)</td>
<td></td>
<td>5.3</td>
<td></td>
<td>ton DM&lt;sup&gt;a&lt;/sup&gt; ha⁻¹</td>
<td>Thago and Silva (2006)</td>
</tr>
<tr>
<td>Pasture productivity (low)</td>
<td></td>
<td>3.0</td>
<td></td>
<td>ton DM ha⁻¹</td>
<td>Lathuilière (2018)</td>
</tr>
<tr>
<td>Dry season irrigation</td>
<td></td>
<td>118</td>
<td></td>
<td>mm</td>
<td>Lathuilière et al. (2018b)</td>
</tr>
<tr>
<td>Small farm dam evaporation</td>
<td></td>
<td>1421</td>
<td></td>
<td>mm y⁻¹</td>
<td>Lathuilière et al. (2018a)</td>
</tr>
<tr>
<td>Runoff coefficient</td>
<td>R</td>
<td>2&lt;sup&gt;b&lt;/sup&gt;</td>
<td></td>
<td>dimensionless</td>
<td>Lathuilière et al. (2017)</td>
</tr>
<tr>
<td>Regional evaporation recycling coefficient (precipitation)</td>
<td>e&lt;sub&gt;r&lt;/sub&gt;</td>
<td>0.22</td>
<td></td>
<td>dimensionless</td>
<td>Berger et al. (2014)</td>
</tr>
<tr>
<td>Regional evaporation recycling coefficient (blue water)</td>
<td>re&lt;sub&gt;r&lt;/sub&gt;</td>
<td>0.07&lt;sup&gt;c&lt;/sup&gt;</td>
<td></td>
<td>dimensionless</td>
<td>Berger et al. (2014)</td>
</tr>
<tr>
<td>Filtration distance to groundwater</td>
<td></td>
<td>0.8–1.5 m</td>
<td></td>
<td>m</td>
<td>Beck et al. (2010)</td>
</tr>
</tbody>
</table>

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<sup>a</sup> DM: dry matter.

<sup>b</sup> Assumption.

<sup>c</sup> The combined re<sub>r</sub> is equal to 0.07.
2.3. Life cycle impact assessment

We follow two frameworks from the UN Environment Life Cycle Initiative to determine mid-point impacts of water consumption and land occupation. Impacts of blue water consumption are assessed by estimating the amount of water deprived to human and ecosystems in the region using the AWARE method (Boulay et al., 2018). Briefly, this method provides a WSF (ISO, 2015) through a characterization factor that represents the degree of competition in a river basin following equation (3),

\[ WSF = LCI_w CF_w \] (3)

where \( LCI_w \) (m\(^3\) ha\(^{-1}\)) is the blue WF inventory, \( CF_w \) (dimensionless) is the characterization factor based on the available water remaining in the basin, taking into account both human and ecosystem blue water consumption. Values of \( CF_w \) were obtained following Boulay et al. (2018),

\[ CF_w = \frac{AMD_{world}}{AMD_i} \] (4)

where \( AMD_{world} \) (0.0136 m\(^3\) m\(^{-2}\) mo\(^{-1}\)) is a global normalization factor representing global blue water availability minus demand, and \( AMD_i \) represents blue water availability minus demand in river basin \( i \) as the difference between human and ecosystem consumptions divided by the area of the basin. For the Xingu River Basin, the annual value of \( CF_w \) is 1 (no irrigation) and 1.1 (with irrigation) (Boulay et al., 2018).

Secondly, impacts of land occupation on the water cycle were determined following Koellner et al. (2013),

\[ I_{occ} = ACF_I I_{occ} \] (5)

where \( I_{occ} \) is the land occupation impact, calculated using characterization factors of impact \( j \) (\( CF_I \)), area \( A \) (ha) and occupation time \( t_{occ} \) (years). Characterization factors were calculated for three midpoint impacts affecting the land and atmospheric water cycles: GWRP following Saad et al. (2013), PRP following Lathuilière et al. (2016a), and RRP which we propose in this study, all of which are shown in equations (6) to (8),

\[ CF_{GWRP} = GWR_{NV} - GWR_{LU} \] (6)

\[ CF_{PRP} = (ET_{NV} - ET_{LU}) r_e \] (7)

\[ CF_{RRP} = (ET_{NV} - ET_{LU}) r_e \] (8)

where \( CF_{GWRP} \), \( CF_{PRP} \), and \( CF_{RRP} \) (m\(^3\) ha\(^{-1}\) y\(^{-1}\)) are the respective characterization factors of land occupation for GWRP, PRP, and RRP; \( GWR \) and ET (m\(^3\) ha\(^{-1}\) y\(^{-1}\)) are respectively groundwater recharge and ET for natural vegetation (\( GWR_{NV}, ET_{NV} \)) and the land use (\( GWR_{LU}, ET_{LU} \)); \( r_e \) is the basin internal evaporation recycling ratio multiplied by a runoff coefficient which together equal to 0.07 (dimensionless), and represents a recycling ratio of water vapor returning to the basin as blue water (Berger et al., 2014). Values of GWR were obtained following the water balance equation described by Saad et al. (2013),

\[ GWR = \frac{P - ET}{R} \] (9)

where \( P \) (mm y\(^{-1}\)) is the annual precipitation, and \( R \) (dimensionless) is the runoff coefficient, both of which are defined for NV and LU to derive \( GWR_{NV} \) and \( GWR_{LU} \).

Characterization factors for TGWF and RBWP (as \( CF_{TGW} \) and \( CF_{RBWP} \) dimensionless) were obtained from equations (10) and (11) from Quinteiro et al. (2015),

\[ CF_{TGW} = \frac{ET_{LU,eff}}{ET_{NV,eff}} \] (10)

\[ CF_{RBWP} = \frac{ET_{LU,eff}}{ET_{EFR,eff}} \] (11)

where \( ET_{LU,eff} < ET_{NV,eff} \) (both of which are obtained from equation (2)), and

\[ ET_{EFR,eff} = P - x_{EFR} \left( P - ET_{NV,eff} \right) \] (12)

where \( x_{EFR} \) (0.42, dimensionless) is the fraction of environmental flow requirements to the long-term mean discharge of the Xingu River Basin (Lathuilière et al., 2016a). The conditions to apply equations (10) and (11) are the characterization factors under specific land occupation scenarios described in this study, in which both characterization factors are mutually exclusive. Therefore, if \( ET_{LU,eff} < ET_{NV,eff} \), then \( CF_{RBWP} = 0 \), and if \( ET_{NV,eff} \leq ET_{LU,eff} < ET_{EFR,eff} \), then \( CF_{TGW} = 0 \) according to Quinteiro et al. (2015).

All characterization factors were derived using previously published input data from both the Amazon and Cerrado biomes as described in Lathuilière et al. (2017) and adapted for rain-fed cropland, irrigated cropland, and pasture using input parameters shown in Table 2. Results are provided here per ha for cropland and cattle production systems, and for 1 kg of crop and 1 kg LW in the Supplemental Material.

3. Results

Potential impacts of cropland and cattle production obtained following calculations from the LCI (Table 3) showed differences with respect to the biome and production system with both positive and negative impacts based on the impact category considered (Figs. 3–5). Land occupation impacts from cropland replacing NV were greater in the Amazon biome than in the Cerrado with larger differences in all categories, particularly with rain-fed cropland’s GWRP (−1490 m\(^3\)) in the Amazon compared to −690 m\(^3\) in the
Cattle, low productivity pasture

<table>
<thead>
<tr>
<th>Product</th>
<th>Water consumption</th>
<th>A ( \text{ha} \text{y}^{-1} )</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropland, rain-fed</td>
<td>Amazon</td>
<td>0</td>
<td>1 Lathuilli et al. (2018b) IBGE (2017)</td>
</tr>
<tr>
<td></td>
<td>Cerrado</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Pasture, rain-fed</td>
<td>Amazon</td>
<td>0</td>
<td>1 Lathuilli et al. (2018b) IBGE (2017)</td>
</tr>
<tr>
<td></td>
<td>Cerrado</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Cattle, low productivity pasture</td>
<td>Amazon</td>
<td>72.5 (12.3)</td>
<td>0.74 Lathuilli (2018), IBGE (2017)</td>
</tr>
<tr>
<td></td>
<td>Cerrado</td>
<td>6.8</td>
<td></td>
</tr>
<tr>
<td>Cattle, high productivity pasture</td>
<td>Amazon</td>
<td>108.0 (18.5)</td>
<td>0.62 Lathuilli (2018), IBGE (2017)</td>
</tr>
<tr>
<td></td>
<td>Cerrado</td>
<td>6.8</td>
<td></td>
</tr>
</tbody>
</table>

\( A \) We also consider the replacement of 1 ha of pasture by 1 ha of cropland in a pasture-to-cropland land occupation.

\( b \) Values calculated when compared to Natural Vegetation.

\( c \) Value in brackets is the effective blue water evaporation following Quinteiro et al. (2015).

Both biomes carry a WSF (representing the degree of competition resulting from blue water consumption at 1298 m³ world equivalents for irrigated cropland) and reductions in impacts to water flows to the atmosphere, and to land. In the Cerrado, the main difference between rain-fed and irrigated cropland was the change in sign of the impacts of PRP (304 m³ to -95 m³), GWRP (-690 m³ to 215 m³) or RRP (97 m³ to -30 m³), while the same shift replaced the TGWF impact (-158 m³) by the RBWP impact (299 m³) (Figs. 3 and 4). The replacement of pasture with cropland for rain-fed production in both Amazon and Cerrado biomes showed much smaller impacts compared to irrigated cropland. Impacts decreased (or increased negatively) for PRP (-426 m³), and RRP (-136 m³), and increased for GWRP (968 m³), and RBWP (1346 m³). Cattle production generally showed lower impacts in the Cerrado biome when compared to the Amazon biome, and in high productivity pasture when compared to low productivity (Fig. 5). The impact categories with the largest magnitude were PRP (502 m³ ha\(^{-1}\) for a low productivity pasture system in the Amazon), GWRP (-1140 m³ ha\(^{-1}\)), TGWF (-503 m³ ha\(^{-1}\)) and RBWP (160 m³ ha\(^{-1}\)) (Fig. 5).

The above impacts were obtained from the characterization factors shown in Tables 4 and 5. Characterization factors for changes in water vapor transfers to the atmosphere with land occupation (\( CF_{PRP} \) \( CF_{TGWF} \)) ranged from -426 m³ ha\(^{-1} \) y\(^{-1}\) for a
pasture-to-irrigated cropland transition (Table 5) to 683 m³ ha⁻¹ y⁻¹ from an Amazon NV-to-pasture transition (Table 4). These characterization factors had matching values of CF_TGWFP of 0 for CF_RBWP and up to 0.28 for CF_TGWF > 0 (Tables 4 and 5). Values of CF_CFWWP were negative when considering NV-to-rain-fed cropland transitions, but positive in Cerrado NV-to-irrigated cropland (215 m³ ha⁻¹ y⁻¹) and pasture-to-irrigated cropland (968 m³ ha⁻¹ y⁻¹). Values of CF_RPP were of opposite sign to that of CF_GWRP with values ranging from −136 m³ ha⁻¹ y⁻¹ (pasture-to-irrigated cropland) to 217 m³ ha⁻¹ y⁻¹ (Amazon NV-to-pasture transition). A positive CF_CFWWP was also matched by a non-zero value of CF_RBWP in Cerrado NV-to-irrigated cropland (0.89), pasture-to-rain-fed cropland (0.73), and pasture-to-irrigated cropland (0.89) transitions (Table 5). Results per kg of cropland and kg of LW are shown in the Supplemental Material.

4. Discussion

4.1. Intensification of cropland and cattle production and their impacts on water partitioning

The combined LCIA methods were able to provide additional information on the potential effects of current intensification at the field level within current trends in the region (Macedo et al., 2012; Spera, 2017). In Southern Amazonia, pasture has historically replaced NV in both Amazon and Cerrado biomes while cropland has replaced NV but also older pasturelands, often leading to indirect land use change through additional deforestation of NV for pasture into the Amazon biome (Arima et al., 2011; Barona et al., 2010). Such deforestation activity has had noted impacts on biodiversity, above and belowground carbon as well as erosion as quantified in LCA (Lathuillière et al., 2017), which can be complemented by the effects described by precipitation partitioning. A NV-to-rain-fed cropland transition was accompanied by a reduction in water vapor transfer to the atmosphere which translated into a loss of water vapor from the basin (as quantified by TGWF), a loss of precipitation recycled within the basin (from PRP), or returning to land as blue water (RRP), leading to additional local groundwater recharge (GWRP). Overall, this transition would increase blue water resources within the basin with a trade-off between upstream groundwater (expressed in GWRP) and downstream surface water resources (TGWF, RBWP). The loss of water from the basin (TGWF, RBWP) can affect water availability downstream and, consequently, increase water scarcity as in the case of cropland irrigation. Moreover, cropland irrigation transferred blue water resources to the atmosphere through ET, especially land occupation on Cerrado NV or pasture. The amount of precipitation recycled within the basin (PRP) actually increased when considering impacts of irrigated cropland in the Cerrado, which could have potential benefits to ecosystems (Lathuillière et al., 2016a).

Impacts of cattle production were affected by low productivity and high productivity pasture mostly from the amount of dry matter that cattle can consume per hectare of pasture as well as the choice of NV. The change in land occupation impacts of cattle following a NV-to-pasture transition showed similar impacts as compared to the NV-to-cropland transitions with losses of water vapor returning to the atmosphere (PRP), as well as surface and groundwater (RRP, GWRP). Similar to irrigation, small farm impoundments constructed for cattle drinking did not carry any losses of water vapor outside of the basin as expressed through TGWF, rather they potentially reduced the production of blue water downstream. Indeed, extensive networks of small farm reservoirs can reduce stream connectivity (Callow and Smettem, 2009) and favor additional evaporation with effects on downstream water availability.

Cropping intensification into pasture overall had the lowest land occupation impacts while cattle intensification in the Cerrado had lower impacts than in the Amazon biome. The Brazilian Federal Forest Code currently places deforestation limits on properties located in the Amazon and Cerrado biomes through the Legal Reserve which requires farmers to respectively maintain 80% and
20% of natural forest cover (depending on year of deforestation and farm size) (Brasil, 2013). This difference has historically led to more deforestation in the Cerrado compared to the Amazon (Strassburg et al., 2017) with impacts on biodiversity and ecosystem services (Lathuilière et al., 2017). There were distinct blue and green water trade-offs expressed in the cropland extensification and cropland irrigation impact assessments expressed through TGWF and RBWP. Both impacts quantify the effects of land use change on downstream water availability and could therefore be linked also to the WSF (see Section 4.2) (Quinteiro et al., 2018).

We note uncertainties with the above interpretation of our results which are common in LCA studies that rely on crop water balances and can affect both LCI and the characterization factors (Payen et al., 2017; Quinteiro et al., 2017). We expect uncertainty in our LCI as a result of geographic differences in water use for crops and cattle across the basin. Our values of ET came from measurements in both rain-fed and irrigated fields (Lathuillière et al., 2018b). These values were used to represent average conditions across the state of Mato Grosso assuming no field runoff or drainage (irrespective of spatial variability in field declination or soil conditions). The amount of water consumed by cattle was based primarily on the total live weight of the animal and represented an average of male and female consumption in Mato Grosso. Drinking water for cattle can vary greatly based on climate (Palhares et al., 2017; Ridoutt et al., 2012), while we also expect geographic differences in the small farm reservoir evaporation across the basin. The characterization factors used in this study represent regional averages for the state of Mato Grosso in the case of GWRP (Lathuillière et al., 2017), while all characterization factors related to the internal processes of the basin (e.g., PRP, WSF) can also know geographic and temporal variability. While our values of ET for NV in both Amazon and Cerrado biomes have been estimated through remote sensing (Lathuillière et al., 2012; Oliveira et al., 2014), field measurements have confirmed the difference in magnitude between vegetation spanning from 965 mm y⁻¹ (Cerrado) to 1384 mm y⁻¹ (Amazon) and follow the precipitation gradient across the two biomes (Lathuillière et al., 2016b). This average difference in ET between the biomes therefore confirms the difference in impacts observed, despite the geographic uncertainty in landscape ET from cropland and pasture.

Furthermore, this study also focused on attributional LCA by allocating impacts to two products assuming that their respective systems are mutually exclusive. In fact, the options proposed for agricultural intensification overall are interconnected. For instance, between 2001 and 2010, close to 4962 km² of pasture was converted to cropland in the portion of the Xingu Basin within Mato Grosso state (Silvério et al., 2015), while cattle population increased, thereby increasing cattle density in the basin (Lathuillière et al., 2018a). Therefore, our proposed cropland extensification option could lead to cattle intensification, but also cattle extensification in- or outside the basin (indirect land use change) which was unaccounted for in this study.

4.2. Complementarity in mid-point impacts

The mid-point impacts used in this study represent changes in hydrological flows as a result of land occupation and water consumption which can be synonymous from an ISO 14046 perspective (ISO, 2015), but were considered separately in this study. On the one hand, land occupation can change hydrology based on precipitation partitioning with consequences on end-point impacts to human health, ecosystems or water resources. The mid-point impacts used in this study could be interpreted as fate factors which are used in LCA to estimate changes in the water cycle to derive end-point impacts (e.g., impacts to ecosystem quality (Núñez et al., 2016)). For instance, in Lathuilière et al. (2016a), PRP represents the precipitation volume not returning to the river basin as a result of a land occupation or transformation activity with potential impacts to terrestrial ecosystems as a result of diminished soil moisture. Similarly, other fate factors have been conceptualized in the context of groundwater extraction (van Zelm et al., 2011). The long-term effects of these fate factors also affect water availability. For instance, the land occupation impact GWRP showed increased recharge following a NV-to-cropland transition, while a reduction in precipitation could also reduce long-term runoff within the basin. These effects on water resources are also expected to change competition over remaining water resources which were expressed with the WSF in the case of blue water consumption only. Land use, land use change and dam operations are known as major contributors to changes in water availability and have already contributed to moving water scarcity further downstream (Veldkamp et al., 2017), which could also be expressed more explicitly in LCA.

The WSF as expressed using the AWARE method (Boulay et al., 2018) aims to answer the question posed by Boulay et al. (2018, 2015) with a specific focus on water quantity: “What is the potential to deprive another freshwater user (human or ecosystem) by consuming freshwater in this region?”. The method is based on the ratio of water consumption to availability where availability is defined as the amount of water remaining once human and ecosystem demands have been met (Boulay et al., 2018). In this method, the WSF is considered a “proxy” mid-point, meaning that it isn’t linked to any particular end-point impact (Boulay et al., 2018). A similar “proxy” mid-point was developed by Berger et al. (2014) as the water depletion index which was used to evaluate a water depletion risk with respect to an activity in a basin. The results from these impact indicators are therefore different than the interpretation of impacts as fate-factors used as a predictive impact assessment expressed through an end-point impact as seen with PRP, RRP, and GWRP. Impact categories TGWF and RBWP best represent the changes in water from land occupation that could be considered “consumed” and could be further represented in a WSF from land occupation (Quinteiro et al., 2018). The water depletion index proposed by Berger et al. (2014) and revised in Berger et al. (2018) is based on a ratio of water consumption to availability which considers the removal of evaporative fraction returning to the basin as precipitation from water consumption through εᵣ (without considering ecosystem water demand). If we interpret TGWF and RBWP as water effectively consumed as a result of land occupation, then we can calculate a WSF using a water depletion of 0.15 m³ depleted per m³ consumed following (Berger et al., 2018). This would provide a risk of freshwater depletion ranging from 11 to 202 m³ water depleted ha⁻¹ for cropland and 2 to 75 m³ water depleted ha⁻¹ for cattle (considering absolute values of TGWF and RBWP). Similarly, Quinteiro et al. (2018) propose to use AWARE characterization factors (or CFₑ, as described in our study, equation (4)) as a means to derive a WSF from a change in river runoff (Fig. 6).

The Water use in LCA (WULCA) working group has recommended a framework for including transfers of freshwater sources and sinks in LCA by considering hydrological “compartments” in the water cycle, keeping in mind regional and global scale effects (Núñez et al., 2018). In our study’s context, the magnitude of these transfers depended primarily on the magnitude of εᵣ, which was used either in the LCI (for TGWF, RBWP, see equation (1)), or in the LCIA phase (for PRP and RRP, see equations (7) and (8)). Our value of εᵣ was constrained to the Xingu River Basin, but could be confined to the Amazon biome or even the continent (Lathuillière et al., 2016a; van der Ent et al., 2010) as a means to represent differences between local and regional hydrological scales. For instance, Lathuillière et al. (2016a) calculated PRP of soybean production in
Amazonia in a small region (2.76 × 10^{10} \text{ m}^2), the Xingu River Basin (5.1 × 10^{11} \text{ m}^2), and the Amazon biome (7.0 × 10^{12} \text{ m}^2) and estimated respective PRP impacts of 86.5 m^3 ton^{-1} soybean, 323 m^3 ton^{-1}, and 703 m^3 ton^{-1} following the corresponding values of \(e_i\) for each area of influence. Similarly, impacts expressed in GWRP could be complemented with fate factors based on groundwater depletion as derived by van Zelm et al. (2011). Moreover, the recognition of impacts to the end-point impact to natural resources could respond to the long-term effects of land occupation on water availability, rather than expressing the impact purely as a WSF.

Within the development of these indicators in LCA, it is important to maintain focus on avoiding double counting on both water quantity and quality perspectives (Núñez et al., 2016).

5. Conclusion

This study aimed to evaluate potential agricultural production options for cropland and cattle in Southern Amazonia by observing agricultural extensification and intensification using six distinct impact assessment methods that focus on the effects of land occupation and water consumption on water quantity. Our crop-land extensification option relying on a pasture-to-cropland land use transition resulted in lower impacts of production when compared to a NV-to-cropland transition, while irrigation showed some potential benefits when focusing specifically on land occupation impacts due to additional water vapor transfers to the atmosphere. The benefits, however, need to also consider the potential for additional water scarcity in the basin expressed by the WSF. The comparison of high and low productivity pastures for cattle revealed the importance of pasture management in reducing the impacts of cattle production in the region, but also the effects of land use on downstream water availability.

While five of the impact assessment methods tested were specifically linked to land occupation, further model integration is needed to assess the full extent of land occupation on the water cycle. We have suggested a path forward to further integrate the link between land occupation impacts and the WSF, while future research should also consider longer term impacts to freshwater resources embodied in a natural resources end-point impact.

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Abbreviations

- GWRP: Groundwater recharge potential
- LCA: Life cycle assessment
- LCI: Life cycle inventory
- LCIA: Life cycle impact assessment
- LU: Current land use
- NV: Natural vegetation
- PRP: Precipitation reduction potential
- RBWP: River blue water production
- RRP: Runoff reduction potential
- TGWF: Terrestrial green water flows

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2019.02.021.

References


