How thirsty is our beef?
Understanding water use for feed production

Ylva Ran

Propositions

1. Green water use should be judged in connection to land use.
   (this thesis)

2. To sustainably manage water resources, we need to integrate different schools of thought.
   (this thesis)

3. Consumption of livestock products should be driven by our common responsibility for a healthy planet rather than our individual responsibility for a healthy diet.

4. Interventions will become more robust and applicable if they integrate experience-based mapping and behavioural insights.

5. Physical activity improves our cognitive learning and contributes to a long and healthy scientific career.

6. If "men" start acting like "women" we can close the gender pay gap.

Propositions belonging to the thesis, entitled

How thirsty is our livestock? – Understanding water use for feed production.

Ylva Ran

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Ylva Ran

Thesis committee

Promotor

Prof. Dr I.J.M. de Boer

Professor of Animal Production Systems

Wageningen University & Research

Co-promotors

Dr C.E. van Middelaar

Researcher, Animal Production Systems Group

Wageningen University & Research

Prof. M. Herrero,

Chief Research Scientist, Agriculture and Food.

Commonwealth Scientific and Industrial Research Organisation, Australia

Other members

Prof. Dr P.J.G.J. Hellegers, Wageningen University & Research

Dr K.K.E. Descheemaeker, Wageningen University & Research

Dr P.W. Leenes, University of Groningen

Dr P. Quinteiro, University of Aveiro, Portugal

This research was conducted under the auspices of the Graduate School Wageningen Institute of Animal Sciences (WIAS)

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Thesis

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Ylva Ran
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Abstract

The global livestock sector currently consumes about 33% of global water withdrawals, primarily for irrigation of feed crops. To understand how the livestock sector can potentially mitigate the impacts of its water use, two factors are critical. First, there are different types of water resources available in the landscape and each have markedly different impacts on the social-ecological landscapes in which they are consumed. Second, increased use of crops for animal feed causes greater competition for water use between the production of feed for animals and food for humans.

This thesis aims to improve our understanding of the effects of consumptive blue water (i.e. ground or surface water) and green water (i.e. soil moisture) in a landscape, and to develop and apply a method to better assess such effects of consumptive water use (CWU) associated with livestock production. We first identified differences in existing methods and developed a conceptual framework for assessing CWU of livestock that aims to address the aforementioned critical factors. This framework was subsequently applied to beef production systems in Brazil and Uruguay. We focused on CWU for animal feed production as this constitutes the vast majority of water demand in livestock systems. Furthermore, we chose beef production since beef cattle can be fed entirely on pastures or on a mixture of pastures and crops.

Results from this thesis confirm the importance of considering both blue and green water resources separately. Moreover, it argues that green water should be considered in regard to the land on which the water resources are used, e.g. cropland or grassland. We showed that the traditional measures of water use efficiency (i.e. litres of CWU per kg of beef produced) is lowest in extensive systems where cattle are fed on natural pastures, and increases if cattle are fed on improved pastures and with feed crops. Our newly developed water use ratio (WUR), however, showed that beef production systems that use high opportunity cost feeds, such as feed crops, can potentially contribute more human digestible proteins by growing food crops than by producing beef. Similarly, it was shown that by using low opportunity cost feeds, such as grass and by-products, livestock systems can have an important contribution to food and nutrition security while avoiding feed-food competition over land and water resources. This thesis illustrates that there are multiple pathways to increase beef production without significantly increasing feed-food competition, and that low-opportunity cost feeds can effectively contribute to a sustainable development of the food sector in areas where resources are scarce.

It was concluded that estimates of water use in livestock value chains should distinguish between the different types of water, i.e. green and blue water and that the water use should be considered in a local context in order to identify potential impacts of CWU in the landscape. To address the impacts resulting from green CWU, green water use should always be categorised according to the land area and land use where it is consumed, for example on cropland or grasslands. This allows for an identification of alternative uses of that land and corresponding water resources and can contribute to more sustainable use of green water resources and the development of a sustainable food sector.

Table of contents

Chapter 1

General Introduction

1.1 Background

There is increasing evidence that the physical limits of the Earth set ultimate boundaries for all human activity (Fischer et al., 2007; Steffen et al., 2015). Our key challenge is, therefore, to produce enough nutritious food for a growing and increasingly affluent global population while avoiding unsustainable use of natural capital that results in the loss of key functions of our global socio-ecological system.

The global food system is the largest user of natural resources: 43% of the global ice- and desert-free land and two thirds of all global water withdrawals are used to produce food (Poore and Nemecek, 2018). A major part of these freshwater withdrawals takes place in regions with high levels of freshwater scarcity, defined as a region with a lack of sufficient available water to meet the demands of water usage for all humans living in that region. Mekonnen and Hoekstra (2016) estimate that four billion people suffer from severe water scarcity for at least one month each year. Thus, as the global population continues to grow and to demand more food, reducing water use in food production has become highly important.

The global livestock sector is responsible for a significant share of natural resource use by the global food system. The sector currently uses more than 80% of agricultural land globally and consumes about 33% of global water withdrawals, primarily used for irrigation of feed crops (Poore and Nemecek, 2018). Two factors are crucial to an understanding of the potential for the livestock sector to mitigate the impacts of its water use.

First, there are different types of water resources available in the landscape and each have markedly different impacts on the social-ecological landscapes in which resources are consumed. The majority of the water used in livestock and feed crop production is rainwater, taken up by plants to sustain the growth of crops and grasses (Mekonnen and Hoekstra, 2012). This water resource is referred to as *green* water, meaning rainfall available as soil moisture for plant growth in the unsaturated zone. The second type of water resource used in agricultural production is the freshwater available in lakes, rivers and aquifers that can be withdrawn and used for drinking water and irrigation among other things. This is defined as *blue water* (Falkenmark, 1995).

Second, livestock systems have historically converted leftovers and by-products from arable land and grass resources, such as low opportunity cost feed (van Zanten et al., 2018), into valuable human edible food, manure and other ecosystem services beneficial to humans. However, the production of and demand for livestock produce has increased rapidly in recent decades (FAOSTAT, 2018) as a result of population growth and increasing average incomes, and the global demand for meat is expected to increase still further (Godfray et al., 2018). This puts an increasing pressure on the livestock sector to increase productivity. As a result, livestock are being fed with crops, such as cereals and oil crops, instead of low opportunity cost feed – a trend that is projected to continue (e.g. Alexandratos and Bruinsma, 2012). Increased use of crops for animal feed causes greater competition for water use between the production of animal feed and of food for human consumption. It also results in a change in the water requirement for the production of animal feed. For example, crops generally require more water per kg than grass (Mekonnen and Hoekstra, 2011a), and croplands are irrigated to a higher degree than grasslands.

Before I define the objectives of the thesis in Section 2, however, the following chapter will first introduce and discuss the relevant terminology in the domain of water use in agriculture, and more particularly for livestock.

1.1.1 Water use in livestock production systems

The relevance of blue and green water

The pathway by which water enters into a livestock system is illustrated in Figure 1. Water can reach the soil through precipitation and then either forms run-off or infiltrates the soil. Part of the infiltrated water remains in the soil as water available for plant uptake via their root systems in the unsaturated soil zone, also referred to as green water. The remaining water can infiltrate further, eventually forming a blue water flow that recharges blue water storage in the saturated zone. Both ground and surface water (i.e. blue water) can be pumped and applied to the soil as irrigation or used to supply livestock drinking water, water for feed mixing and servicing water for animals (Figure 1).

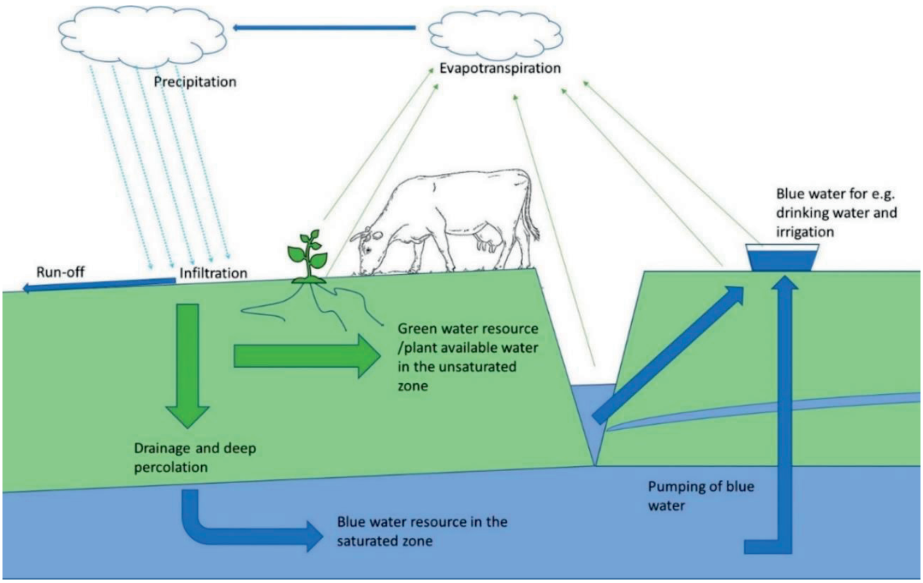

Figure 1: Hydrological flows in cattle production systems. Authors own.

Green and blue water resources are not static pools. They generate green and blue water flows which means that they are not entirely exclusive. Green water use, and altered green water flows, can result in changes in blue water availability. For example, if consumptive green water use increases upstream because cropland is afforested, blue water generation may subsequently be smaller downstream (Karlberg et al., 2009). It is important to distinguish between these two types of water use since their impact on the ecohydrological landscape, as well as strategies for overcoming issues associated with their overuse, are markedly different.

Although it is possible to argue that freshwater is a renewable resource (Pradinaud et al., 2019), availability is regulated in space and time: in space by the amount of precipitation that falls over a defined area; and in time by, for example, drought and rainfall periods that regulate the amount of precipitation that falls in a specific time period. Freshwater availability is also dependent on the ecohydrological context in the area where the precipitation is received, that is, how water is infiltrated and circulated in a landscape (Figure 1). Thus, freshwater availability is directly connected to landscape parameters, such as soil type, soil composition and agricultural management practices (Pradinaud et al., 2019). Water availability is also regulated by the multi-purpose nature of water resources, and the multitude of users within the landscape (Schyns et al., 2015).

Historically, water use estimates have focused on assessments of blue water use for irrigation, industry and/or domestic use (e.g. Quinteiro et al. 2018; Schyns et al. 2019), thereby excluding the majority of water use in agriculture, which constitutes use of green water. Blue water is not spatially bound to where it is extracted from but can be withdrawn and used in a different location. The impact of blue freshwater withdrawal, such as groundwater that is pumped and used to irrigate crops, is immediately apparent as less water is available for extraction by other users that share the same source. It is possible to argue that withdrawing water from a basin that is experiencing water stress, that is, an inability to meet human and ecosystem demands for freshwater (Quinteiro et al., 2018), would be of greater consequence to other users, such as households, agriculture and industry, than if doing so from a basin where water is abundant (Ridoutt and Pfister, 2010).

Green water availability, however, remains largely invisible within the landscape (Schyns et al., 2019). Green water is restricted to land (Falkenmark and Rockström, 2006; 2010) and therefore indirectly managed and affected by land use decisions (Schyns et al., 2019). If competition for land and associated green water resources is high, the use of those resources can have a big impact on conflicting users that depend on green water availability in the same area, for example, agricultural production of food, feed, fuel and fibre.

Consumptive water use in livestock production systems

This thesis primarily focuses on water use that is consumptive. C*onsumptive water use* (CWU) is water that is withdrawn from a watershed but not discharged to the same watershed because it evaporates, is embodied in plants or animals, or is discharged to a different watershed (Falkenmark and Lannerstad, 2005). Water can be withdrawn from a watershed without being consumed, for example, to generate hydropower electricity, meaning that after use it is circulated back to the same watershed.

More than 90% of consumptive blue and green water use in livestock production is associated with feed production (De Boer et al., 2013; Mekonnen and Hoekstra, 2012; Steinfeld et al., 2006) and the majority constitutes use of green water. Blue water use for the production of animal feed is directly related to the use of irrigation water during feed crop production. In addition to differences in agricultural management practices, CWU for livestock production will differ between regions as a result of crop and grass-specific water requirements (Figure 2a and b). One kilogram of maize for use as livestock feed requires about 1200 litres of CWU if grown in Australia but it requires 1600 litres of CWU to grow 1 kg of maize in Brazil and 2300 litres in India. (Figure 2a). These differences in water requirement mean that two animals reared in two different regions, despite having identical feed compositions, will require significantly different green and blue water volumes to produce their feed.

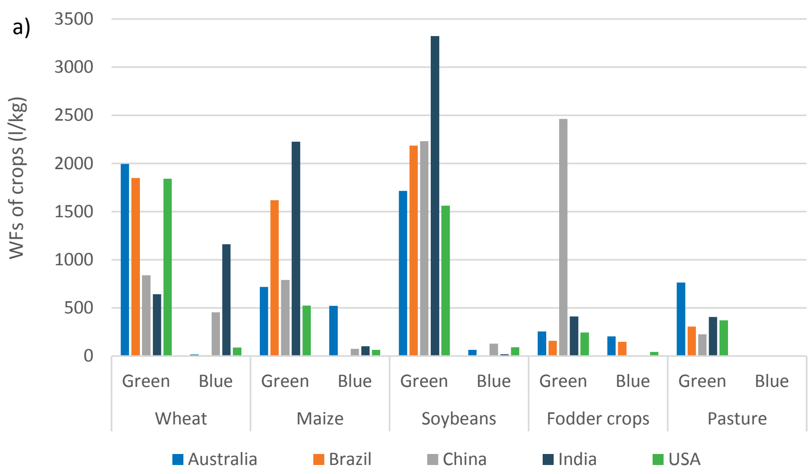

Figure 2a: Water footprints (WF) for major feed crops, fodder crops and pasture in five countries, divided into green and blue water.
Source: Mekonnen and Hoekstra, 2010a

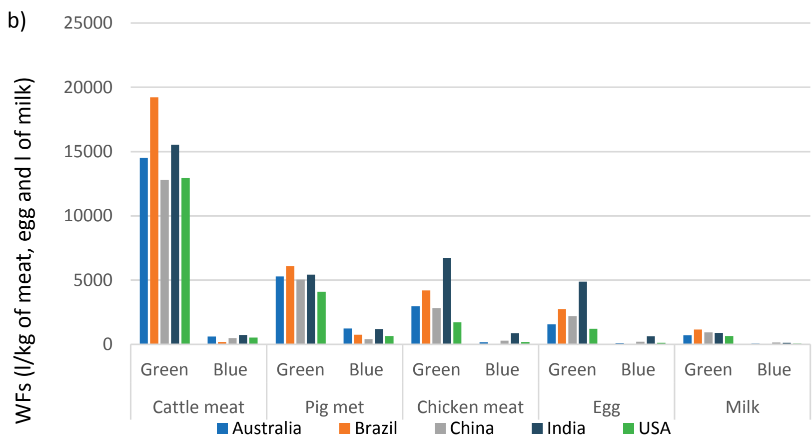

Figure 2b: Water footprints (WFs) for different livestock products in five countries, divided into green and blue water.
Source: Mekonnen and Hoekstra (2010a)

Since pig and poultry production systems rely almost entirely on crops for animal feed, and crops are irrigated to a greater extent than pasture, the proportion of blue water use will be higher for those systems compared to pasture-based ruminant systems (Figure 2b). In pasture-based ruminant systems, blue water is primarily consumed as water required for drinking, feed-mixing and cleaning, approximately between 2–8% of the total CWU (De Boer et al., 2013; Mekonnen and Hoekstra, 2012; Steinfeld et al., 2006), as global grasslands are hardly irrigated.

Animal products embed between 5 and 20 times more virtual water, that is, consumptive water embedded in a product (Hoekstra et al., 2011), per kg than crop products (see Figures 2a and 2b). Cattle meat is the largest consumer of water by far, followed by pig and poultry meat. Other livestock products such as eggs and milk, however, have a lower CWU than meat, ranging from 1000 to 5000 litres per kg depending on the region of production.

It is not just that cattle meat requires significantly more water per kg than other animal products. Cattle production systems are particularly relevant subjects for study with respect to the sustainable use of natural resources in a landscape with a multitude of users because, unlike pig and poultry systems, they make use of both crops and grasslands. Grasslands can be native pastures unsuitable for agricultural production, or improved pastures or planted grasslands that could well be used to produce an array of crops. Thus, native grasslands have no opportunity cost for food production and such water use is arguably of lesser consequence than water use with high opportunity costs, for example for use on croplands.

To ensure that global food production does not exceed the planetary boundaries for water resource use, it is fundamental to understand and estimate the requirement for both types of water (i.e. green and blue) in livestock systems. Furthermore, we need to explore how water use differs between production systems and regions, and to identify the associated impacts in the landscape, for example on an area where a multitude of users and functions share the same resources. This would enable identification and potential avoidance of unnecessary green and blue water stress. This thesis therefore addresses these aspects while focusing on water use for cattle production systems, with a specific emphasis on the production of livestock feed as this is responsible for the vast majority of water use in livestock rearing.

In addition, it is particularly important to acknowledge that the livestock sector is part of a global food system. Globalisation have resulted in livestock value chains in which feed and animal productions stages are increasingly decoupled (Erb et al., 2009; Galloway et al., 2007). Livestock in Europe, for example, are increasingly fed with feed ingredients from across the entire globe, such as soy from Brazil, while the animal-source food products are consumed in Europe or reexported and consumed elsewhere. As a consequence, the use of natural resources embedded in consumed products, such as water, is becoming spatially dissociated from the location of consumption and the negative environmental impacts associated with the production supply chain occurs at a location that is not visible to consumers.

1.2 Knowledge gaps in understanding water use for livestock, traded livestock products and livestock feed

There are currently a multitude of methods that seek to estimate water use in livestock production systems and inform decision-makers on how best to use water resources with regard to livestock. The most commonly applied approach is the water footprint assessment (WFA), which was introduced in 2002 (Hoekstra and Huynen, 2002). A water footprint (WF) is defined as the total volume of consumptive water used to produce goods and services consumed by individuals or communities or produced by a business. Water use in a WF is generally measured in terms of the volume of green and blue water consumed (evapotranspired or incorporated into a product) (Hoekstra et al., 2011). In an attempt to also account for polluted water resources, Hoekstra et al. (2011) introduced the term grey water use, which refers to the volume of freshwater required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality standards. Grey water is an indirect proxy for water quality and does not address consumptive water use and is therefore not included in this thesis.

At first, there was a strong emphasis on estimating total consumptive water use in livestock systems and WFAs were commonly presented as a single WF, without highlighting the different types of water resource used, (green, blue and grey). These aggregated WFAs obscure the markedly different impacts that green, blue and grey water use have in the landscape (e.g. Perry 2014; Ridoutt and Huang 2012; Ridoutt et al. 2012a). However, more recent WFA studies do distinguish between the different types of water resources and present results in terms of individual blue, green and grey WFs (e.g. Mekonnen and Hoekstra 2012).

WFAs have successfully highlighted the large amounts of water required for the production of livestock products (e.g. Mekonnen and Hoekstra 2012). Moreover, they highlight that water use should be considered from a value-chain or life-cycle perspective in order to manage water resources in increasingly globalised livestock supply chains (Hoekstra, 2017; Lathuillière et al., 2018a). The vast majority of the WFA studies, however, remain focused on volumetric estimates and/or comparisons, and therefore fail to contextualize the potential impact of the water use, illustrated as WFs, in the landscape.

In response to the criticism that WFAs generally lack any connection to the relevant and scale-dependent impacts of water use in the area in which the water is abstracted, the Life-Cycle Assessment (LCA) network developed an impact assessment of blue CWU. The LCA-based WF was developed as a complementary method to the traditional WFA (ISO, 2014). LCA-based blue WFs measure water scarcity as a ratio between water use and water availability (Pfister et al., 2009), thereby excluding natural run-off and environmental flow requirements[1]. This method has been used to measure the impact of CWU for livestock products in several studies (e.g. Ridoutt et al., 2012a; Zonderland-Thomassen and Ledgard, 2012; Zonderland-Thomassen et al., 2014), by quantifying the volumetric impact on blue water availability in litres of water/water equivalents per kg of beef or fat and protein corrected milk.

[1] Environmental flow requirement is defined as the volume of water of sufficient quality required to sustain freshwater and estuarian ecosystems as well as human-well-being and livelihoods that depend on these ecosystems (Hoekstra et al., 2011)

The blue water focus of LCA-based WF assessments has generated criticism that they disregard the large volumes of green water used in the production of agricultural products, (Hoekstra, 2016). In response, the LCA community has recently tried to include green water use in LCA-based WF assessments by relating water uses to the impact on freshwater availability. Green water is included as 'net green water', that is, the net change in green water available for natural vegetation (Pfister et al., 2017; Ridoutt and Pfister, 2010). This method is distinctly different from the method used in the WFA assessments that include all the green water required for production. As many challenges remain with regard to how to estimate the relative impacts of changes in green CWU (Quinteiro et al., 2018), LCA-based WF assessments still primarily focus on blue water.

Similar to the LCA community, the WF network also developed a methodological approach on how to contextualise WFs and relate them to impacts. To this end, the water footprint sustainability assessment (WFSA), launched in the WF manual by Hoekstra et al. (2011), determines the environmental sustainability of green, blue and grey water use by dividing aggregated WFs with water resources available for human purposes (Hoekstra et al., 2011). The WFSA for blue water is based on the same approach that was developed for the LCA-based WF studies (Ridoutt and Pfister, 2010, 2013; Ridoutt et al., 2012a). Like the sustainability assessment of blue WFs, the WFSA for green water is calculated as the total green WF in a basin divided by the available green water for human purposes. The green water availability in a defined area and time is calculated as the total evapotranspiration of rainwater from land, excluding evapotranspiration from land that is reserved for natural vegetation and nature conservation according to predefined standards of such land requirements, and from land that cannot be made productive (Hoekstra et al., 2011).

While both the LCA-based WF approach and the WFSA contextualise blue WFs, they only partly contextualise green WFs as they do not explicitly relate green water use to its land use (e.g. Schyns et al. 2019). Thus, by disregarding the competition over, and alternative uses of, land and green water resources other than that of natural vegetation, current methods miss out on the opportunity to identify improvement options that contribute to a more sustainable use of green water resources.

Thus, there is a need to further explore and study the impacts of CWU, especially green water CWU, for livestock production systems and livestock products in the context of the landscape where the water is consumed, and with regard to the multitude of potential uses within that landscape. As livestock and agricultural value chains are becoming increasingly global, we also need to better understand the interlinkages and dependencies between scales in the global water system (Rockström et al., 2014; Vörösmarty et al., 2013). Thus, methodological approaches are needed that increase transparency in supply-chains to better understand consumer and producer linkages to water use at the location of consumption.

To address these knowledge gaps, the first objective of this thesis is to improve our understanding of the effects of CWU (i.e. blue and green) on a landscape. In this sense the landscape is defined as an area with a multitude of functions and users that share the same land and water resources, such as production of food, feed, fuel, fibre and maintenance of biodiversity and ecosystem services. The second objective is to develop and apply a methodological approach to better assess such effects of CWU.

Section 1.3 describes the contribution of each chapter to these objectives and provides an outline of the thesis.

1.3 Contribution and outline of thesis

To address the objectives, various South American beef production systems, ranging from extensive to intensive, were compared in Uruguay and Brazil. The countries and systems were chosen because they are able to inform and help to answer the above-mentioned research objectives. Livestock systems, like agricultural production more generally, in Uruguay and Brazil, are primarily rainfed and therefore largely reliant on green water resources (e.g. Lathuillière et al., 2016a; Mekonnen and Hoekstra, 2010a). They are situated in an agricultural region where natural pastures are part of the native vegetation (e.g. Lahsen et al., 2016; Modernel et al., 2013). Thus, cattle rearing is an important agricultural sub-sector and both countries are large producers, consumers and exporters of cattle meat (FAOSTAT, 2018). In addition, their cattle production systems are still primarily pasture based – much more so than, for example the United States and Europe (e.g. Cardoso et al., 2016; Millen et al., 2009).

Despite the current dominance of pasture-based beef systems, agricultural production is currently undergoing a transition and both countries have become important producers of cereals and oil crops, primarily soybean, that are to a large extent exported for use as animal feed in an increasingly global livestock sector (e.g. Arima et al., 2011; Picasso et al., 2014). Levels of both cattle and crop production are expected to continue to increase in the region (e.g. Alexandratos and Bruinsma, 2012; Flachsbarth et al., 2015). Thus, cattle compete for resources both directly and indirectly with a rapidly expanding crop production sector in both countries. This makes Uruguay and Brazil suitable areas of study for analysing existing and potential CWU for cattle production systems and identifying pathways to increase food production while minimising the negative impact of CWU for other users and ecosystem functions in the landscape.

The structure of the thesis is illustrated in Figure 3. Chapters two and three aim to contribute to a better understanding of CWU in livestock production systems and across scales, and to understand the different methods of assessing CWU. The knowledge gaps identified in these chapters are subsequently used to develop the method presented in Chapter four. The methodology from chapter four is applied and further developed in Chapter six. Chapter five discusses global livestock supply chains. The study contributes a methodological approach to better including and assessing CWU of traded livestock feed in CWU assessments of livestock products. Chapter seven discusses all the chapters with regard to the thesis objectives and to the existing literature in relevant fields.

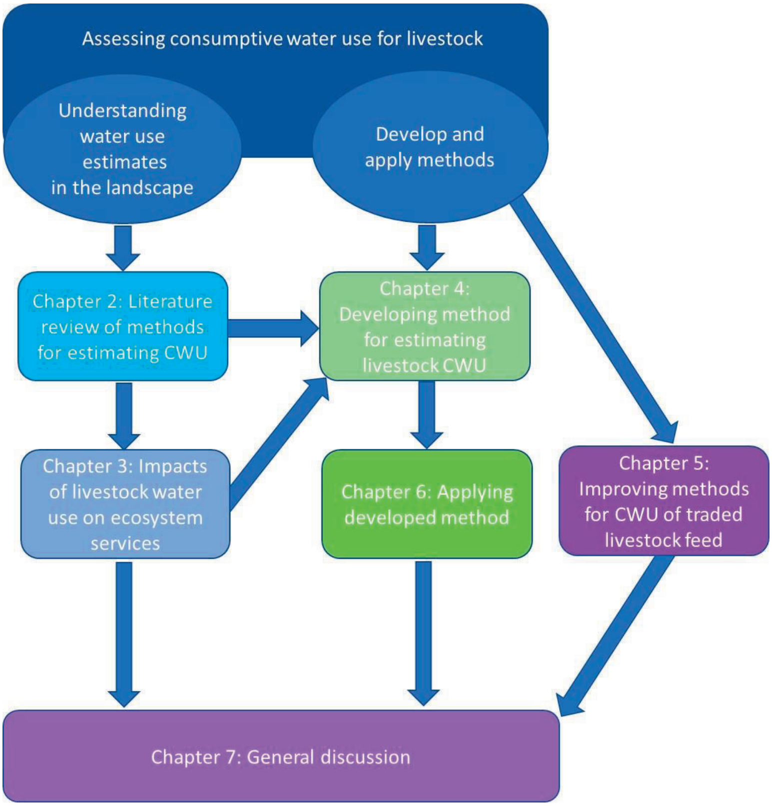

Figure 3: Outline of thesis.

1.3.1 Thesis chapters

Chapter two links green and blue water use to spatially explicit impacts by investigating the demand for water in livestock production and the potential connection with water-related ecosystem services in a landscape. The chapter analyses and compares water use for beef production systems along an intensification gradient, from largely pasture based to intensive feedlot production systems.

Chapter three presents a literature review that explores the different methods for estimating livestock water use, identifying similarities and differences, and highlighting areas for methodological development.

Chapter four addresses the shortcomings of current water use assessment methods and proposes a new method, referred to as the water use ratio, for evaluating water use in livestock production systems from a food systems perspective. The water use ratio calculates the maximum amount of human digestible protein (HDP) that can be produced from 1 kg of animal sourced foods (ASF) and corresponding CWU and compares this with potential HDP derived from food crops using the same CWU. This method is then used to estimate the CWU of beef production systems along an intensification gradient, exploring trends and potential pathways for beef production with a sustainable use of water resources.

Chapter five addresses the CWU of traded crops that can be used as livestock feed. The chapter responds to the limitation in traditional methodological approaches that aim to assess water use associated with traded commodities. The chapter presents a novel methodological approach to improving CWU assessments of traded agricultural and livestock feed crops. This chapter focuses on blue water use and scarcity in the production of two Brazilian feed and food crops, soybeans and sugarcane.

Chapter six investigates past, present and future trends in water use in four beef production systems in the Brazilian Cerrado. CWU is compared over time to see how water use has changed, and feed-food competition over resources is assessed. In addition, we calculated maximum potential beef production, and estimated the associated CWU in the Cerrado states to explore how Brazil can meet expectations of an increase in beef production while at the same time using water resources sustainably to minimise competition over already scarce resources.

Chapter seven is a general discussion of findings of all the chapters in regard to the existing literature.

Chapter 2

Rapidly intensified beef production in Uruguay: Impacts on water-related ecosystem services

This chapter is adapted from: Ran, Y., Deutsch, L., Lannerstad, M., Heinke, J. (2013). Rapidly intensified beef production in Uruguay: Impacts on water-related ecosystem services. *Aquatic Procedia 77-87*. DOI: 10.1016/j.aqpro.2013.07.007

Abstract

Livestock production is one of the fastest growing agricultural subsectors globally and requires large amounts of natural, often scarce, resources, such as land and water. To identify sustainable management of resources in livestock systems it is important to quantify resource use, but also to connect such use to ecosystem impacts. The aim of this paper, therefore, was to investigate consumptive water use (CWU) for three beef production systems in Uruguay; extensive, mixed and intensive, by quantifying the water use and categorising it according to the type of water used, e.g. rainwater or ground or surface water, and land use. In addition, the paper explores impacts on water-related ecosystem services associated with each beef production system. The mixed beef production system was identified as the dominant system and thus, had the largest total CWU. However, the system required the least amount of water per kg of beef. The extensive system had the largest CWU per kg of beef but had the least potential effect on water-related ecosystem services, such as erosion control, habitat and soil formation. The feedlot system required slightly more water per kg of beef than the mixed production system, but intensification of beef production was linked to negative impacts on ecosystem services to a higher degree than the other two systems.

2.1 Introduction

Livestock production is one of the fastest growing agricultural subsectors worldwide. It contributes about 40% of global agricultural gross domestic product and involves about 50% of the world's farmers, with 11 billion people in developing countries depending on livestock for their income and livelihood (WB, 2009). Today, livestock is estimated to use one third of global cereal production (Mottet et al., 2017) and by 2050, global food production is expected to increase by 70-110% and livestock production by 70-80% . This result in that the livestock sector is estimated to require 50% of the additional 1 billion tons of grains that will be produced, for animal feed (IAASTD, 2008; Tilman et al., 2011). In addition, if we would adopt a western-based diet, exemplified as an average USA diet, an additional 138% of land would be required for food production (Alexander et al., 2016).

To ensure that the expected agricultural increase does not imply unsustainable exploration of natural resources, it will be imperative to limit agricultural expansion into vulnerable ecosystems and avoid irreversible undermining of agroecosystem resilience (Naylor, 2009; Rockström et al., 2009b). Thus, a large part of the increase in production must be met through sustainable intensification of agriculture (Tilman et al., 2011), i.e. "sustainable intensification methods that improve efficiency gains to produce more food without using more land, water, and other inputs" (Herrero et al., 2010). Livestock production systems must adapt and develop to be able to meet the increasing demand for meat, at the same time sustaining ecosystem services functions and biodiversity (Modernel et al., 2016).

Pasture providing grazing for livestock already covers about 30% of the ice-free land surface globally (FAOSTAT, 2013) and a third of global cropland is already dedicated to cultivation of animal feed. Consequently, livestock production already uses huge amounts of land and water resources, i.e. one third of the total global water use for agriculture (Mekonnen and Hoekstra, 2012). The number of people enduring water scarcity, i.e. when their access to annual renewable freshwater is less than 500 m^3 per capita (Rijsberman, 2006), is steadily increasing, from 1.2 billion around 2007 to 1.8 billion in 2025 (FAO, 2007; Molden et al., 2007a).

In addition to playing a vital role in food production, water provides, supports and regulates a multitude of ecosystem services in agricultural and livestock production systems (Deutsch et al., 2010). However, the complexity of water related processes coupled to various livestock management practices in relation to direct and indirect consequences for ecosystem services over time, is not well understood.

This chapter outlines how livestock production and associated water use may impact ecosystems functions by looking at ecosystem services, coupled land use change associated with different agricultural management practices in animal feed production.

By choosing appropriate agricultural practices unintended negative effects in social-ecological systems can be avoided. For example, certain agricultural practices, such as conventional tillage and long-term continuous mono-cropping, may negatively affect the ability of an ecosystem to provide services beneficial to human purposes, such as erosion control and favourable soil formation (e.g García-Préchac and Durán 2001, Fernandez et al. 2002, Dogliotti 2003, Garcia-Préchac et al. 2004, Bot and Benites 2005, Raudsepp-Hearne et al. 2010), both important ecosystem functions for agricultural systems. Thus, without a thorough understanding of established linkages between management and affected ecological systems and functions, agricultural management may cause negative effects on ecological features that are imperative for long-term sustainable agricultural production.

In this chapter, we first explain the links between ecosystem services, ecohydrology and livestock production. Second, we apply a case study of Uruguayan beef production systems to illustrate potential impacts of observed changes in livestock management practices due to intensification, on the generation of water-related ecosystem services over time, taking an ecohydrological approach. Water use for beef production is quantified for three production systems of varying degrees of intensification to compare tentative differences in hydrological impacts from changes in agricultural practices.

2.2 Background

2.2.1 Ecosystem services and water in livestock production systems

Agricultural systems, of which livestock are often an integral part, are multifunctional and can generate a wide range of ecosystem services simultaneously (Figure 1). Provisioning ecosystem services, such as provisioning of food, both plant and animal-source food, and water availability are central to agricultural systems by definition, and dependent on water for functioning (Deutsch et al., 2010). However, as an integrated part of an ecosystem, maintaining supportive and regulating ecosystem services, such as soil formation, erosion control, climate regulation, habitat provisioning, water cycling, water quality and primary production (i.e. grassland productivity) is essential to generate provisioning services (MEA, 2005). In addition, livestock production systems contribute cultural ecosystem services, such as spiritual, recreational, aesthetical, social and educational services, respectively. However, these are not the focus of this study, and therefore not discussed further in this chapter.

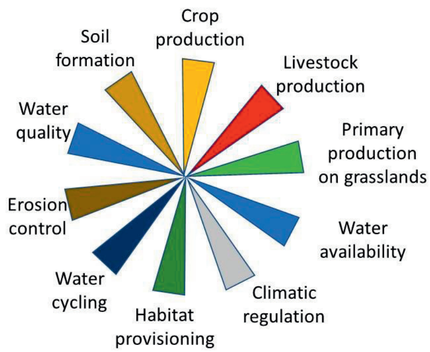

Figure 1. Bundle of ecosystem services associated with livestock production systems.
Source: Adapted from Ran (2012).

For livestock production systems to be sustainable, they must deal with the challenge of managing these multiple ecosystem services over time and across multiple scales. Maintaining or increasing one specific ecosystem service may have negative impacts or result in negative trade-offs, on the supply of other ecosystem services (MEA, 2005). Such trade-offs and interlinkages must be properly identified and understood in order to sustain the multitude of ecosystem services that livestock production systems provide.

Impacts of livestock production systems on water-related ecosystem services (WRES) can be separated into three categories (Deutsch et al., 2010): i) withdrawal of water for irrigation of crops, which affects water availability in downstream aquatic ecosystems and/or groundwater reserves; ii) change in land cover, e.g. deforestation for pastures and croplands, which alters water cycling, precipitation patterns, climate regulation, habitat formation and the functioning of ecosystems; and iii) land use change, e.g. cropping patterns, tillage and grazing practices, which may affect runoff, infiltration, erosion control as well as evapotranspiration (Pradinaud et al., 2019). In addition to supporting provisioning ecosystem services, formation of soil, i.e. the composition of soil, plays a critical role in water cycling processes to regulate freshwater supply in terrestrial ecosystems (O'Geen et al., 2010).

2.2.2 Ecohydrology of livestock production systems

Globally, more than 90% of the consumptive water use (CWU), embedded in livestock products originates from the production of animal feed. Water use is consumptive when water is withdrawn from a watershed, and not discharged to the same because it evaporates, is embodied in plants or animal products or is discharged to a different watershed (Falkenmark and Lannerstad, 2005). The vast majority of CWU for livestock is green water, i.e. water available as soil moisture in the upper part of the soil, from infiltrated precipitation in rainfed agriculture (Falkenmark, 1995; Mekonnen and Hoekstra, 2012). The remainder is blue water, or liquid freshwater in surface water and groundwater bodies, used for drinking purposes, servicing and feed-mixing (De Boer et al., 2013; Falkenmark, 1995; Mekonnen and Hoekstra, 2012; Steinfeld et al., 2006). An increase or decrease of CWU for livestock is therefore directly linked to the demand for, and composition of, feed at a specific site and for a specific production system.

Water enters the livestock production system when rainfall reaches the soil surface, and either infiltrates into the soil profile, adding to the green water resource as soil moisture, or forms surface runoff. Soil moisture held in the unsaturated zone is available for plant water uptake and soil evaporation (Figure 2). Infiltration rate is dependent on, for example, the soil texture and structure, vegetation types and cover, water content of the soil, soil temperature and rainfall intensity (O'Geen et al., 2010; Saxton and Rawls, 2006)

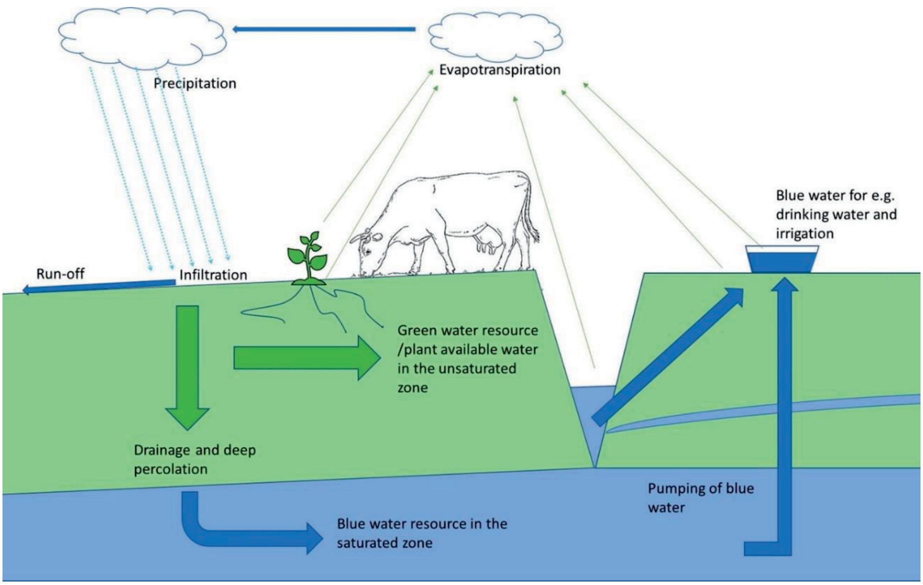

Figure 2: Hydrological flows in livestock production systems. Authors own.

Evaporative flows of water from the soil (evaporation) or vegetation (transpiration and interception) are collectively termed green water flows or evapotranspiration (Falkenmark, 1995). In the livestock production system, consumptive green water supports feed and fodder production, and blue water from surface water and ground water is sometimes used to supplement the green water resource as irrigation. In addition, livestock systems consume blue water for drinking water and servicing. Water that is not retained in the soil profile or consumed as evaporation may continue through the soil profile to form groundwater (i.e. blue water). Amplified blue water extraction can locally increase the risk for water stress for people dependent on the same river basin further downstream (Ridoutt and Pfister, 2010).

Soils differ in their capacity to retain water as a function of soil properties and soil depth. Soil water holding capacity (SWHC) varies with porosity, particle size, soil organic matter (SOM) and structure, thus strongly affects water flow and partitioning in the soil (Figure 2). A high SOM content supports the formation and stability of soil aggregates (Bot and Benites, 2005), reducing the risk of soil crusting and contribution to the general stability of the pore structure (Saxton and Rawls, 2006), which can reduce both runoff and erosion. High SOM levels also positively influence soil porosity via macrofaunal activity, e.g. bioturbation by earthworms. In effect, influencing infiltration into, and percolation through, the soil, which increases hydraulic conductivity. SWHC is also positively linked to high levels of SOM, due to an increase of micro- and macropores. The greater pore space results in increased ability to maintain moisture in the soil, i.e. it increases green water availability (Bot and Benites, 2005; Saxton and Rawls, 2006). While SOM content remains stable over time in a crop rotation system incorporating pastures and forage crops, it decreases significantly over time under continuous cropping (Figure 3). Fertilizer application may reduce the decline in SOM content by increasing biomass production both above and below ground (Bot and Benites, 2005; Dogliotti et al., 2003).

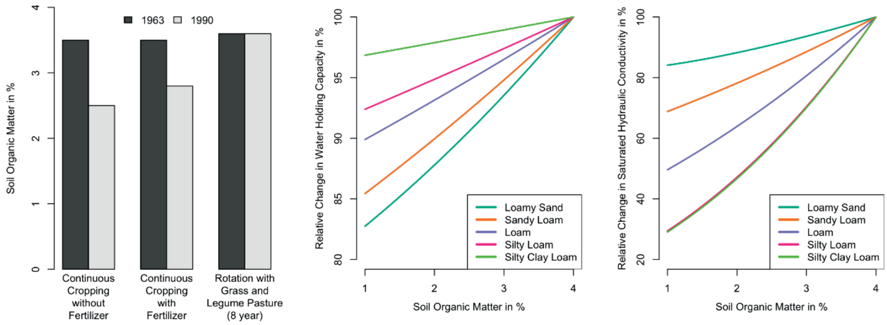

Figure 3. Relationship between soil hydraulic properties and soil organic matter content (Dogliotti et al., 2003; Saxton and Rawls, 2006).

Moreover, soil compaction from farm machinery and livestock has been shown to impact infiltration capacity. For instance, soil compaction caused by livestock exerting a pressure of 200-250 kPa has been shown to reduce infiltration properties of more than 80%, compared to non-compacted soils (Chyba et al., 2014). Green water availability may be affected by agricultural practices, e.g. continuous cropping and tillage (Bot and Benites, 2005; O'Geen et al., 2010). In addition, green water use can affect blue water availability by altering water cycling and reduce water flows towards groundwater formation (Deutsch et al., 2010). The topsoil layer can also be lost as a result of erosion, which will reduce infiltration and increase runoff (Pimentel and Burgess, 2013). Excessive grazing is well-known for causing erosion with reduced soil depth and siltation downstream as unwanted consequences. Lastly, livestock rearing may impact the water quality downstream from leaching of nutrients, pesticides and antibiotics.

2.3 Methodology and case study description

Uruguay has a long history of extensive cattle production, with 70-80% of the country's land area under permanent meadows and pastures (natural, improved and cultivated) (FAOSTAT, 2012). Since 1960, beef production has almost doubled, from about 300,000 to nearly 600,0000 tons (Figure 4a), even though the domestic consumption of locally produced beef decreased by more than half during the same period. In 2010, more than 80 percent of Uruguayan beef was exported (Figure 4a). The country is also exporting a large amount of crops, mainly soybeans used as animal feed. Between 2000 and 2009 soybean production soared from 70,000 tons to 1.2 million tons, with more than 90% of the production going for export (Figure 4b).

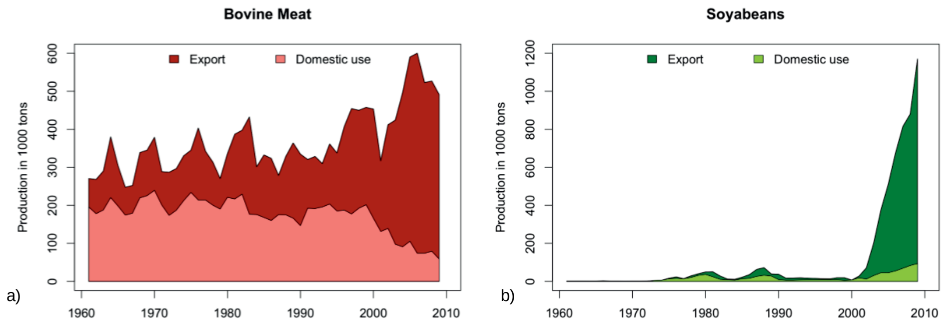

Figure 4. Export and domestic use of a) beef and b) soybean in Uruguay, 1961-2009.
Source: FAOSTAT (2012).

Trends in Uruguayan agriculture are driven by the growing global demand for meat and animal feed. An increased demand, and export prices, on feed crops, is a major driver behind an expansion of the crop area in Uruguay, increasing from about 1 to almost 2 million ha, which has occurred at the expense of permanent pastures (Figure 5). Since 2002, most of the increase in crop area has been due to increased soybean production for export. The cropping area for soybeans has increased from less than 80,000 ha to almost 900,000 ha, now covering almost half of the area used for cultivation of crops in Uruguay (Figure 5).

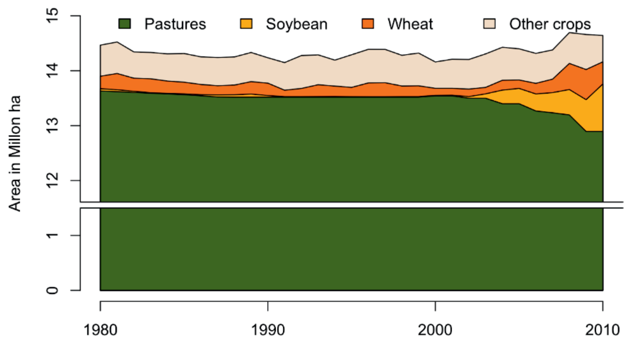

Figure 5. Area under wheat, soybean, other crops and pasture, Uruguay, 1980-2010.
Source: MGAP (2010).

Underpinning this observed increase in production of beef are changes in livestock production systems, and specifically an intensification of agricultural management practices resulting in higher livestock densities in pastures, an increase in supplementary feeding with high-protein feed crops and finishing animals in feedlots (Chiara and Ferreira, 2012; Modernel et al., 2013).

2.3.1 Identification of drivers and change in water-related ecosystem services in Uruguayan livestock production systems

To identify trends in Uruguayan agricultural management practices, related to the studied production systems that may influence ecosystem services in the landscape, a set of semi-structured interviews were conducted with key actors in the beef and soy value chains. In total, over 40 actors in the Uruguayan soybean and beef production supply chain were interviewed including researchers, grain producers, service provisioners, government officials, logistics functions, equipment providers (e.g. agronomists, insurance agents and cooperatives), commodities merchandisers and non-governmental organizations. Interviews were transcribed and used to identify patterns and drivers of agricultural development and agricultural management practices used in livestock and animal feed production. In addition, we complemented the findings from the interviews with information and data found in the literature.

Thereafter the identified trends in management practices were linked with changes in ecohydrological processes, and subsequently with potential impacts on the generation of ecosystem services over time.

2.3.2 Estimating beef production and crop water use for three hypothetical livestock production systems

To quantify the consequences of an intensification of Uruguayan beef production on water use, we compare three conceptual beef production systems along an intensification gradient, operating on the Uruguayan Pampas. The three systems differ in terms of feed composition and cattle production cycle as illustrated in Figure 6 and Table 1.

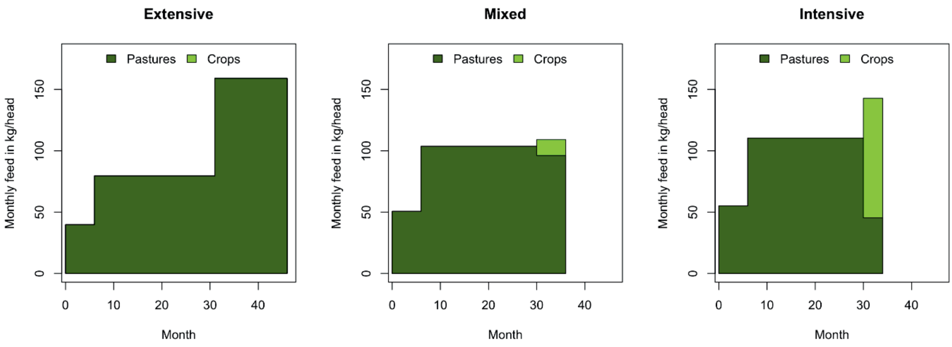

Figure 6: Feed composition of pastures and/or animal feed crops, and the lifespan of cattle in three production systems in Uruguay; 1) extensive, 2) mixed and 3) intensive beef production (Becoña, 2012; Becoña et al., 2014; Beretta, 2003; Modernel, 2012; Pigurina, 1998).

The extensive system relies entirely on natural pastures, and cattle are kept in the system for 43 months. In the mixed production system, cattle are primarily fed on improved pastures, i.e. seeded pastures of fescue (*Festuca arundinacea Schreb.*), white clover (*Trifolium repens L.*) and birds foot trefoil (*Lotus corniculatus L.)* (Modernel et al., 2013). However, during the last six months, the finishing stage, 14% of their feed comes from supplements comprised of sorghum, wheat, maize, soybeans and sunflower seeds. The lifespan of cattle is 33 months in the mixed systems. For the intensive system, cattle are also primarily fed on improved pastures. However, in the short finishing system of four months, cattle are fed 85% supplements and the life cycle is 31 months (Table 1).

As illustrated in Table 1, dry matter intake for cattle is calculated based on coefficients in Mieres et al. (2004), NRC (1996) and AFRC (1993). The systems differ in terms of dry matter intake, daily weight gain and land area requirement. Thus, production of meat per input of resources and over time, such as water quantity and land area, is a function of the availability and quality of feed.

Table 1: Parameters for the three Uruguayan beef production systems and cow-calf (CC), backgrounding (B) and finishing (F) stages. Adapted from Picasso et al. (2014), Modernel et al. (2013) and Modernel (2012).

System	Extensive			Mixed			Intensive			Source
Phase	CC	B	F	CC	B	F	CC	B	F	
Dry matter intake (kg/animal/day)	9.5	9.5	9.5	9.5	9.5	10.0	9.5	9.5	12.3	Mieres et al. (2004), NRC (1996) and AFRC (1993)
Average dry matter digestibility (%)	55			67			81			Mieres et al. (2004)
Crude protein in diet (%)	9.5			15.0			11.1			Mieres et al. (2004)
Metabolizable energy in diet (Mcal/animal/day)	17.5			22.2			28.5			Mieres et al. (2004)
Average daily gain (kg/animal/day)	0.3			0.7			1.3			Ferrés (2004); Pigurina (1998); Risso (1997)
Time (months)	3	24	16	3	24	6	3	24	4	
Area of system (ha per animal)	0.9			0.7			0.7	0.4		(Modernel et al., 2013)

The production for each system is based on expert opinion and slaughter ages (INAC, 2012), to identify the share of the total cattle herd that are finished in each of the three different systems.

CWU for the various feed crops used in beef production in Uruguay was estimated by combining modelled crop and grass water requirements from the global dynamic hydrological model entitled Lund Potsdam Jena managed land, LPJmL (Bondeau et al., 2007; Fader et al., 2010; Gerten et al., 2005; Haberl et al., 2007; Rost et al., 2008a) with feed composition and feed requirements for cattle in the three production systems as listed in Table 1.

The LPJmL model estimates area specific crop water requirement for each feed type, both irrigated and rainfed, vegetation growth and yield per pixel at a resolution of 0.5°. The LPJmL model yields results on CWU per type of vegetation, in m^3 per ton of fresh matter for crops, or per ton of dry matter for grasses. Irrigated areas are determined by land use data input (Monfreda et al., 2008) and no water stress is assumed during biomass growth. When the upper soil layer experience insufficient water content, water is assumed to be added to maintain the ratio of 0.7 between plant canopy water supply and atmospheric demand for transpiration. This water balance provides the daily irrigation requirement for maintaining conditions (Bondeau et al., 2007).

The LPJmL model depends on input data of monthly averages of e.g. temperature, precipitation, days with precipitation, hours of sunshine (Bondeau et al., 2007) and soil texture and concentration of CO_2 (Sitch et al., 2003). Monthly precipitation data is diverted over each day determined by a generator (Gerten et al., 2004). The LPJmL model furthermore includes a dataset of land use connected to each crop functional type (CFT). This implies that the cover of each CFT in percentage is represented for each pixel, both irrigated and rainfed, on a yearly basis.

2.4 Results and discussion

2.4.1 Consumptive water use and beef production

Beef production systems in Uruguay are almost exclusively dependent on green water resources to support pasture and feed crop production, i.e. rainfed (Figure 7). Blue water use is zero in the extensive system and constitute less than one percent in both mixed and intensive production, originating from irrigation of crops in mixed and intensively managed systems. About half of Uruguayan beef production is produced in mixed systems, and thus have a larger total CWU than the other two systems; about 4 km^3 and 40 % of total CWU, compared with 3 km^3, 35 % of total CWU, and 2 km^3 and 25 % for extensive and intensive systems, respectively.

To enable a comparison in regard to water use efficiency, we also compare CWU per kg of produced beef in each of the three systems. Extensive and intensive production systems have similar water use; 19 300 litres/kg and 18 900 litres/kg respectively, whilst mixed systems require less water per production unit; 15 800 litres/kg.

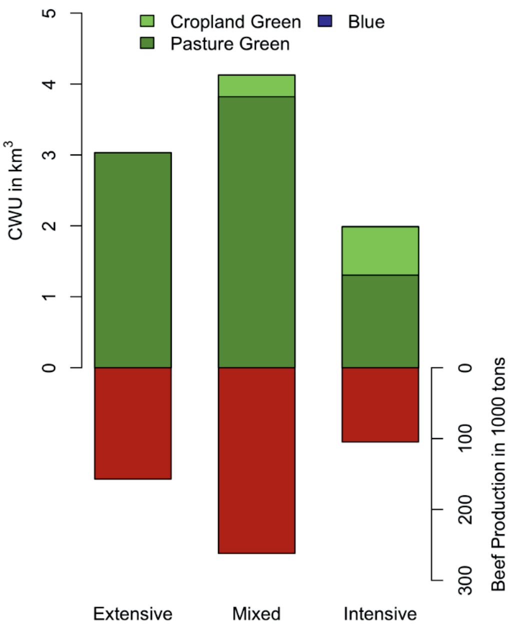

Figure 7: Consumptive water use for three beef production systems in Uruguay estimated per product and as total consumptive water use.

These differences are a result of a number of factors; the length of the cattle cycle and daily weight gain, type and quality of feed and agricultural management practices are all parameters that affect the feed composition, thus, the CWU for the cattle production systems. For example, feed supplements constituting of soybean, wheat, maize and sorghum have a higher water requirement than grass on natural and improved pasture. However, the increased weight gain for cattle result in that they reach their final weight much faster than in the extended system which results in that less water is required for production of animal feed over time. The larger use of water in the intensive system per kg of produced beef, in comparison to the mixed system is the result of that the latter uses much more feed crops as animal feed, and that the cattle cycle is only two months shorter than in the mixed system, which is not enough to further increase the water use productivity of the system.

Actual amounts of CWU for livestock production are only relevant in the local context, for example related to water availability, water scarcity and competition over water resources in a shared river basin. In recent decades in Uruguay, water management has become more important as drought frequency and rainfall variability have increased because of changes in the El Niño–Southern Oscillation induced by climate change. Although the total amount of rainfall has increased, the interannual variation has also increased (Barreiro, 2010; Cazes-Boezio et al., 2003; Pisciottano et al., 1994), and frequency of dry spells and the number of consecutive dry days per year are expected to become more frequent (Eleftheratos et al., 2010; IPCC, 2012). This makes rainfed agriculture less predictable, more vulnerable and potentially more dependent on irrigation to sustain satisfactory yields, increasing the importance of sustainable management of land and water resources in Uruguayan agriculture.

The quantification of CWU for animal feed also highlights that the water requirement for livestock, and the type of water resource used, can be managed through changes in the composition of livestock diets and production system practices. In general, crops demand more water than grass, but they are also more efficiently converted into digestible energy by the animal (Mieres et al., 2004). The results of CWU in this study highlights the mixed beef production system as being the most water efficient in terms of CWU/kg of beef despite also having the largest total CWU, as the system is dominant in the country.

The CWU for the three production systems is also categorized over type of land where the water is consumed and indicate a potential conflict over water resource use between production of feed crops and pastures to feed livestock and production of food crops. The categorization of CWU over land illustrates that seven percent of CWU for mixed production and more than one third of CWU for intensive beef production is used on cropland, and it can be argued that this water could instead be used for production of food for direct consumption by humans, rather than used for production of animal feed.

It should be noted that reducing feed-food competition may not result in a reduction of CWU. The argument merely points to that, when resources becomes scare, it might be wise to consider allocating resources to production of foods that humans can consume directly rather than via animals, avoiding the large conversion losses in animal production.

2.4.2 Livestock management practices and effects on water-related ecosystem services

In addition to relating CWU estimates to water availability, impacts on ecosystem services are also not fully captured if water use is only quantitatively measured. The choice of agricultural management practices will not only affect the quantity of water required for crop or pasture growth but might result in cascading alterations of hydrological features on multiple scales, affecting a multitude of water-related ecosystem services over time (Bot and Benites, 2005; Deutsch et al., 2010; Falkenmark and Rockström, 2004; Keys et al., 2012a).

The large soybean expansion in the Rio De la Plata grasslands have induced two types of intensifying changes in livestock production systems in the area: 1) increased use of grains as feed to cattle on improved pastures and in feedlots, 2) increased stocking rates, as a result of increased competition over land (Chiara and Ferreira, 2012; Modernel et al., 2016) potentially causing overgrazing. Interviews with key actors in the beef and soy value chains confirmed this trend and revealed a number of related changes in Uruguay's agricultural management practices, i) conversion of grasslands and pastures to crop production, primarily soy used for livestock feed; ii) improvement of natural grasslands by seeding in ryegrass, clover and other species with addition of fertilizers; iii) removal of pastures from crop rotations and increase in continuous mono-cropping with soybean; iv) increased use of inputs such as fertilizers and pesticides; v) increased irrigation; vi) increased herd density on pastures; and vii) increased use of crops as animal feed, driven by the increasing demand for livestock products and animal feed crops. These findings were supported by similar observations in the literature (MGAP, 2004, 2010; MGAP and DIEA, 2015; Modernel et al., 2016)

Table 2 illustrates how identified drivers of intensification of livestock production systems in Uruguay may affect water-related ecosystem services in the country, as a result of agricultural changes in management practices associated with such drivers. For example, removing grazing in crop rotations may result in a decrease in SOM (e.g. Bot and Benites 2005; Dogliotti et al. 2003; Latawiec et al. 2017), in turn decreasing SWHC, and ultimately affecting water cycling in the landscape (Bot and Benites, 2005; Gordon et al., 2008).

Thus, despite the positive effects on provisioning ecosystems services, such as increased crop and livestock production, this development can be expected to negatively impact supporting and regulating ecosystem services, for example soil formation and erosion control. As Uruguay is expected to experience an increased frequency of dry conditions (Eleftheratos et al., 2010; IPCC, 2012), the importance of actively managing SOM will increase in order to maintain the SWHC of soils. High stocking rates and overgrazing of native grasslands have already resulted in increased soil erosion and carbon losses, i.e. climate regulation, in the Rio de la Plata grasslands region, (Modernel et al., 2016; Overbeck et al., 2007). In addition, land use change, for example transforming native grasslands to croplands, have resulted in decreasing SOM levels (Diaz-Zorita et al., 2002; Sala and Paruelo, 1997).

Table 2: Identified drivers and their potential effects on water-related ecosystem services

Drivers of ecosystem change	**Effects in indicators on ecohydrological and other biophysical processes in the landscape**[1]	**Potential effects on water-related ecosystem services**[1]
Conversion of grasslands and pastures to crop production	Infiltrability / runoff. Risk for reduced SOM and increased erosion.	Changes in *water cycling,* reduced *soil formation* and *habitat provisioning*.
Increased use of inputs such as fertilizer and pesticides	Leaching of nutrients to downstream waterbodies and groundwater	*Water cycling (water quality)*
Improvement of natural grasslands	Increases nitrogen content in the grassland, reducing the need for fertilizer addition. Well-managed grasslands prevent run-off of nutrients and pesticides and preserve riparian areas	Increased *primary production*. No negative effect on *water quality*. Positive effect on *habitat provisioning*.
Removal of pasture / Continuous cropping	Decreased SOM, and increased risk of erosion	Decreased *soil health* and *erosion control*
Increased use of inputs such as fertilizer and pesticides	Leaching of nutrients to downstream waterbodies and groundwater	Decreased *water quality*.
Consumptive water use	Large consumptive water use may reduce both green and blue water availability in the landscape	Decreased *water quantity*
Increased irrigation	Increased withdrawal of water from water bodies and groundwater (see fig 6)	Decreased *water quantity*
Increased herd density on pastures	Soil compaction reducing infiltration rates and soil porosity and resulting in a loss of carbon from the soil	Decreased *soil formation*, *erosion control* and *climatic regulation*
Increased use of crops as animal feed	Increased conversion of grasslands to croplands resulting in larger herds on smaller pasture areas. Croplands may also require an increased use of fertilizers and pesticides. A change in land cover is potentially decreasing SOM, thus soil-water holding capacity and risk compaction of soil.	Positive effects on provisioning ecosystem services; *livestock and crop production,* increasing productivity. Decreased *water quality* as a result of fertilizer and pesticide use. Decrease in *soil formation*.

[1] Dogliotti et al. (2003); Latawiec et al. (2017); Modernel et al. (2016); Picasso et al. (2014); field interviews

It is, however, important to note that intensification of agriculture is not a threat *per se* to sustained long-term productivity of livestock systems in Uruguay. As illustrated in this chapter, a certain degree of intensification, as in the mixed system, will in fact result in higher water use efficiencies compared to extensively managed systems. With adequate management, for example crop production with pasture in rotation, soil compaction and erosion can be controlled. In order for production systems to be sustainable agricultural management must see to the entire bundle of ecosystem services associated with production systems, not only focusing on improving provisioning services.

2.5 Conclusion

The livestock revolution is expected to continue with increased global demand for livestock products, much of which will originate from the developing world. Thus, increased productivity in livestock and animal feed production will become even more important. One of the key challenges for livestock production will be to enhance and increase the provisioning services of food production and water availability, without degrading supporting or regulating ecosystem services such as favourable soil formation and erosion control.

As a response to higher international demand for meat, livestock production systems are gradually being intensified. An example from Uruguay illustrates how this intensification is expressed as higher stocking rates, and an increased use of grains as feed to cattle on improved pastures and in feedlots. This study showed how changes in agricultural management practices associated with intensification of beef production have resulted in higher food production in the short term. However, by gradually reducing supporting ecosystem services such as soil formation (higher erosion, increased soil compaction, and reduced soil organic matter) and consequently green water availability, there is a risk that agricultural productivity will be lost over time. Or that agricultural systems will depend on larger inputs to maintain key ecosystem services such as erosion control and the capacity of soils to withhold water.

Zooming in on the use of water for livestock production, estimates of CWU for beef production in Uruguay show that mixed production is the most water-efficient system in terms of total consumptive water use; only a small fraction of the CWU is from croplands, and the system is high yielding compared to more extensive systems. In addition, the feed-food competition is lower than for the more intensive production systems. Therefore, if sustainably managed, mixed production appears to be the preferable production system; however, this is dependent on stocking density, the feed composition (e.g. type and amount of grain used as fodder) and agricultural management.

In an era of global change, policymakers must balance short-term positive economic effects of intensified crop and livestock production with efforts to mitigate long-term negative environmental impacts. By highlighting the interconnections of water and livestock production, the findings in this paper increase the understanding of the complex hydrological processes linked to agricultural management practices. For future research, the analysis of water in relation to livestock production must be extended beyond the actual CWU estimates for feed towards an ecohydrological context. Identifying the role of water in generating water-related ecosystem services is a key research area where further knowledge is needed.

2

Chapter 3

Assessing water resource use in livestock production: A review of methods

Ylva Ran[a,b], Mats Lannerstad[c] , Corina E. van Middelaar[a], Mario Herrero[d], Imke J.M. de Boer[a]

[a] Animal Production Systems Group, Wageningen University, PO Box 338, 6700 AH Wageningen, The Netherlands

[b] Stockholm Environment Institute, Linnégatan 87D, 11523 Stockholm, Sweden

[c] International Livestock Research Institute, PO Box 30709, 00100 Nairobi, Kenya

[d] Commonwealth Scientific and Industrial Research Organization, Bioscience Precinct Carmody RD, St. Lucia, 4067, Queensland, Australia

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Abstract

This paper reviews existing methods for assessing livestock water resource use, recognizing that water plays a vital role in global food supply and that livestock production systems consumes a large amount of the available water resources. A number of methods have contributed to the development of water resources use assessments of livestock production. The methods reviewed in this study were classified into three categories: water productivity assessments, water footprint assessments and life cycle assessments. The water productivity approach has been used to assess benefits of livestock production systems related to their consumptive water use; the water footprint approach has raised awareness of the large amounts of water required for livestock production; and life cycle assessments highlight the important connection between water resource use and local impacts.

For each of the methods we distinguish strengths and weaknesses in assessing water resource use in livestock production. As a result, we identify three key areas for improvement: 1) both green and blue water resources should be included in assessments and presented separately to provide informative results; 2) water quality should not be summarized within quantitative assessments of water resource use; and 3) methods for assessing water use in livestock systems must consider the alternative uses, multiple uses and benefits of a certain resource in a specific location.

3.1 Introduction

The demand for animal-source foods is expected to double by 2050 (IAASTD, 2008), driven by population growth, urbanization, and rising incomes (Delgado et al., 1999). The major part of the increase in the production and consumption of animal products will take place in developing countries (Alexandratos and Bruinsma, 2012). It will be imperative to limit agricultural expansion into vulnerable ecosystems and avoid irreversible undermining of agroecosystem resilience (Naylor, 2009; Rockström et al., 2009c). There is a broad consensus among agricultural scientists that a large part of the expected increase in demand for animal-source food must be met by a sustainable intensification of agriculture, that is, production of more food without using more natural resources, such as land and water, and without increasing emissions into water, air and soil (Herrero et al., 2010; Tilman et al., 2011).

At present, global livestock production demands about 30% of the global agricultural water requirement, including rain and irrigation water used for the production of feed and withdrawals for livestock husbandry (Mekonnen and Hoekstra, 2012). A major part of freshwater withdrawals already take place in basins suffering high water scarcity and the pressure on water resource availability is expected to increase (FAO, 2007; Molden, 2007a; 2007b; Kummu et al., 2014). The number of people living in regions with absolute water scarcity, i.e. with annual renewable freshwater less than 500 m^3 per capita per year (Rijsberman, 2006), is expected to increase from 1.2 billion today to 1.8 billion by 2025. Two-thirds of the world population is projected to be suffering from water stress by 2025 (FAO, 2007; Molden, 2007a).

3.1.1 Water resource use in agriculture

To properly account for different and competing uses of limited water resources it is important to define different types of water use. Two fundamentally different water uses are non-consumptive water use and consumptive water use (CWU). Freshwater withdrawals for domestic and industrial purposes normally have large return flows that, although often degraded as a result of pollution, can in principal be reused downstream. Consumptive water use, most notably evapotranspiration during use, primarily during plant growth of irrigated and rainfed crops and pastures, on the other hand, results in vapor flow leaving the basin that is not available for reuse (Falkenmark and Lannerstad, 2005).

Traditionally, assessments of water use in agriculture have focused on withdrawals from water bodies and aquifers for irrigation, industry, and municipal or domestic uses (e.g. Shiklomanov, 2000). These assessments did not initially account for the agricultural appropriation of huge amounts of naturally infiltrated rainfall in the soil. To illustrate the importance of both soil moisture and water withdrawals for sustainable agricultural production, water resources can be divided into green water, which refers to soil moisture available to plant growth, and blue water, which refers to liquid water stored in water bodies (Falkenmark, 1995). The important role that green water resources play in agricultural production was highlighted at the end of the 1990s (Falkenmark and Lundqvist, 1997; Falkenmark et al., 1998; Rockström, 1999, 1999. Today the concepts of green and blue water are widely used to describe and assess water use in agriculture, including livestock production (e.g. Molden, 2007a; 2007b; Hoekstra and Mekonnen, 2012; Mekonnen and Hoekstra, 2012). Gray water is a third water volume concept that has been introduced to capture the quantities of water being made unavailable for use due to pollution, i.e. the volume of freshwater that is assumed to be required to assimilate the load of pollutants (Hoekstra et al., 2011). From a hydrological perspective, the distinction between green and blue water is not always ideal, since these two water resources are not always clearly distinguishable from each other. Water flows across the landscape and can change from one resource to the other. However, the distinction between green and blue water is useful for assessing and improving water use since they are managed differently and affect the environment in different ways (Keys et al., 2012b). Blue water can be managed in both time and space, for example in reservoirs and through canals and pipes, and is used both for irrigation in agriculture and for domestic and industrial services. Green water, on the other hand, is coupled to land use and primarily supports plant growth on cropland or grassland, and other terrestrial ecosystem services (Schyns et al., 2015).

Green water dominates water use in agricultural production and globally accounts for about 80% of the CWU on agricultural land (e.g. Molden et al., 2007a; Rockström et al., 2014). In livestock production, green water accounts for 90% of total CWU (Mekonnen and Hoekstra, 2012), since livestock production also depends on rainfed grazing land. In total about 98% of the total CWU, green and blue, in livestock production can be attributed to evapotranspiration during plant growth, e.g. feed crops, roughage and pastures. Only about 2-8% of the CWU originates from blue water used as drinking water, for servicing and as feed-mixing water (Steinfeld et al., 2006; Mekonnen and Hoekstra, 2012; de Boer et al., 2013). Estimates of the total global agriculture water footprint indicate that livestock appropriates 29%, with pasture alone accounting for almost 14% of global agricultural green water use (Hoekstra and Mekonnen, 2012; Mekonnen and Hoekstra, 2011a, 2012).

Given the levels of blue water scarcity in many regions, future challenges related to water use and water availability in agriculture will be linked to more efficient, but also increased, use of green water resources (Rockström et al., 2009a). This is particularly true for livestock production, which is largely rainfed. Changing dietary preferences for an increasing share of animal source foods (e.g. Delgado et al., 1999; Lal, 2013) underline the need to find pathways to increase water productivity in both crop and livestock production (Molden et al., 2007a; 2007b; 2010). Improved efficiency will be important in this context, but the expected increase in demand for food, and animal-source foods in particular, will require additional water quantities to be appropriated (Falkenmark and Lannerstad 2010; Lannerstad et al., 2014). This development will increase the global competition for the scarce water resources available for agriculture and result in local environmental impacts such as agricultural horizontal expansion, dwindling rivers and falling ground water levels (Rockström et al., 2007).

3.1.2 Water resource use in livestock production

In the past decade, a number of papers have proposed different approaches to relating water use in livestock production to local impacts on the environment and ecosystem functions (Milà i Canals et al., 2009; Deutsch et al., 2010; Ridoutt and Pfister, 2010; Ran et al., 2013; Ridoutt and Pfister, 2013). The life cycle assessment (LCA) network developed a water stress-related water footprint (Pfister et al., 2009; Ridoutt and Pfister, 2010, 2013) and expanded the LCA methodology to include water in environmental impact assessments of livestock production (de Boer et al., 2013). Other studies emphasize impacts of livestock production systems on water-mediated ecosystem functions. One example is assessments of potential changes in water partitioning, like impacts of heavy grazing pressure on vegetation cover and soil composition, influencing water infiltration (e.g. Deutsch et al. 2010).

To grasp the impacts on water use associated with each specific livestock production system, assessments should consider temporal and spatial differences in water, land and animal management, and how these affect the local hydrology (Deutsch et al., 2010).

Although several different approaches have been proposed, there is no clear or consistent method for assessing water resource use in livestock production. A comparison between published studies is often hindered by differences in terminology and system boundaries, as well as in impact assessment methods and indicators. Thus, stakeholders might find it hard to identify whether differences in water use between livestock products and livestock production systems really exist, or only appear to exist because of a different method of calculation. In addition, this often makes it difficult for stakeholders to identify whether there is a direct link between water use and environmental impacts, such as water scarcity and local water quality impacts.

Existing water assessments of livestock products present significantly larger water footprints, particularly for cattle meat, than assessments of crop production (e.g. van Breugel et al., 2010; Hoekstra and Mekonnen, 2012; Mekonnen and Hoekstra, 2012). In order to optimise total water use in global food systems it is imperative to ensure that methodological differences are understood and how results from different studies can be compared. Approaches must take into account that, in contrast to food cropping systems, water use in livestock systems result in competition over water resources between food and feed production.

Reviews of water use in agriculture have been published prior to this study (e.g. Kounina et al., 2013; Schyns et al., 2015) but despite the dominance of green water resource use in agriculture, studies have continued to emphasize blue water resources and developing methods to relate blue water use to water scarcity (e.g. Ridoutt and Pfister, 2010). The recent review by Schyns et al. (2015) highlights the importance of developing indicators that also consider green water use, but the study is not specifically related to livestock or to the comparison of results from different studies. Thus far, livestock production systems have received little attention for their freshwater use (Mekonnen and Hoekstra, 2012). As a result of the limited attention paid to green water resources and water use by livestock systems, competition for water resources from multiple users (e.g., feed-food competition) has been overlooked and the discussion around the opportunity cost of green water resources has not been addressed.

Livestock production systems are a major contributor to the world food system and a large consumer of water resources. This study reviews existing methods of assessing livestock water use, recognizing water as a limited resource in global agriculture. The review highlights different methodological aspects of water use in livestock systems. It has two aims: first, to identify the differences, strengths and weaknesses of existing methods of assessing water use in livestock production; and, second, to identify a number of key areas in which water assessment methods can be further developed in order to better inform decision makers about the complexity of water use in livestock production systems.

Assessments of beef cattle systems were chosen to compare methods for assessing water use in livestock production, in order to compare a system that relies on both water resource use on cropland and grassland for the production of animal feed.

3.2 Review of methods

A first step in quantifying water resource use in livestock production is an assessment of actual CWU during the production of animal feed crops and grass, as well as for drinking and servicing water. There are various methods for assessing CWU related to plant growth, and these are briefly described in the next section of this review. The CWU is then related to the comparative unit (animal products), the production system or a nation. Assessment methods depend on the intended use of the assessment of water resource use in livestock production. This review has divided methods that assess water resource use in livestock production into three broad categories. All three categories have been developed for different purposes, resulting in methodological differences. The paper first discusses differences in methods to estimate water requirement for crops and grasses. Following, the three categories of methods are described and discussed to provide insights into the wide variations in water resource use assessments for livestock products and identify key areas for improvement.

3.2.1 Approaches to assessing water use for feed crops and forages

Part of the large variation in water assessments is related to methodological differences in assessing the water resource use of livestock systems (see Tables 1 and 2). The majority of the water used in livestock production is associated with the production of animal feed. Studies often combine several methods for quantifying CWU in feed production. For example, many studies use hydrological modelling at different spatial resolutions to estimate evapotranspiration during the cultivation of feeds (Bondeau et al., 2007; Zhuo et al., 2016). This generates a bias, however, since every analysis has to rely on a number of assumptions linked to the different models, which are not necessarily tailored to the system being studied.

Assessing CWU linked to grazed biomass from pastures is a specific methodological challenge, particularly for ruminant production systems. A simplified approach is to attribute all, or a fixed share, of the evapotranspiration from all the biomass that grows in a pasture area to the livestock grazing on these lands (Pimentel et al., 1997, 2004). Another approach is to base the estimate on the feed required to produce a certain amount of animal outputs and estimate the corresponding evapotranspiration for that feed. Such calculations are more accurate, since they generate water estimates directly related to the quantity of biomass consumed by animals rather than the production of biomass over an area.

The uncertainties in the different methods and models are a general problem in livestock-water assessments. Although hydrological models can generate the necessary data to estimate the CWU, they often operate at a higher spatial resolution than is required to provide insights into local environmental impacts following changes in CWU. Moreover, the more precise a model is, the higher the demand for accurate input data becomes. More easily accessible web-based models generally require less input data but rely more on both general and less spatially explicit assumptions and data e.g. Aquacrop (Steduto et al., 2009). It is interesting to note that none of the studies included in this review performed any sensitivity analysis.

Remote sensing is another way to estimate the evapotranspiration of different land covers. In combination with other methods, such as farmer surveys, hydrological modelling and secondary data on animal feed requirements and feed intake (van Breugel et al., 2010; Nosetto et al., 2012), remote sensing can be used to estimate CWU related to livestock production. Estimates using remote sensing can be used to calculate CWU and water stress related to feed production and, in combination with secondary data, to better account for irrigation performance at multiple spatial scales (Ahmad et al., 2009a, 2009b). However, remote sensing does not accomplish differentiation between appropriated green and blue water resources. Nor does it account for the competition for water resources between crops used to produce livestock feed and crops that can be directly consumed by humans. Instead, the focus is mainly on quantifying the evaporation from different land cover types to illustrate the impact of land cover change.

The methods for estimating the animal feed requirements and intake in a given production system also differ greatly between studies and range from animal to herd up to system level. The RUMINANT model (Herrero et al., 2008, 2010) predicts the voluntary intake of feed and nutrition by cattle, sheep and goats at the animal level, while the Global Livestock Environmental Assessment Model (GLEAM) developed by the Food and Agriculture Organization of the United Nations (FAO) operates at the herd level (Gerber et al., 2013). The agricultural land and biomass (ALBIO) model is a physical model of global agricultural systems that predicts land use and feed requirements for animal production systems (Wirsenius, 2000, 2003; Wirsenius et al., 2010). By combining feed requirement estimates from such models with hydrological models it is possible to generate global, regional and national livestock water analyses that rely solely on secondary data (e.g. Ran, 2010). Data on animal feed requirements can also be obtained from field studies, interviews and household surveys, or based on reference values found in the literature (NRC, 2000; Hoekstra et al., 2011; Ridoutt et al., 2012a).

The wide range of available methods that can be used to assess the CWU requirement for plant growth generates differences in CWU assessments for both feed crops and grazed biomass. However, the different hydrological assessment models do not explain the variations identified in the results from the different livestock water use methodologies reviewed for this study. The fundamental differences in results are highlighted below, in a comparison of three categories of methodological approach to livestock CWU assessment.

3.2.2 Comparing three categories of methods for livestock water assessment

This review divides the methods commonly used to quantify water use in livestock production into three categories: water productivity assessments, water footprint assessments and LCAs. These three categories are based on the fundamental differences between the methods, in terms of the water assessment data compared, their purpose and the information gained from the results generated. The three categories reflect the fact that the methods have been developed to meet different needs.

Water productivity assessments were developed to assess and improve water use in agriculture. Early assessments of benefits per unit of blue water withdrawal for irrigation have later been expanded to benefits per total consumptive water use, which also includes green consumptive water use. The water footprint methodology was developed to increase knowledge of the human pressure on water resources and provide a consistent way to inform consumers and producers about their water use. Life cycle assessments aim to connect resource use to local environmental impacts and exist for many different products and indicators.

Each of the three categories is reviewed with regard to methodological approach, strengths and benefits, and relevant areas of application. A summary of the key attributes of each category is presented in Table 1 and results from the three different method categories are compared in Table 2.

Table 1. The three method categories for assessing water use in livestock production

Method Category	Methodological Approach	Benefits	Drawbacks	Application Livestock Production
Water productivity (WP)	Calculates the ratio of net benefits per depleted water quantity. The method is appropriate at the river basin, watershed and community scale.	Can include multiple benefits derived from livestock. Requires relatively little data.	WP normally do not present separate figures for blue and green consumptive water use.	WP has been used for comparison of different livestock production systems to identify potentials to increase water productivity for smallholders in water-scarce areas and areas with poor water resource development.
Water footprint (WF)	Calculates the volume of consumptive water use and water quantities assumed to be required to dilute generated pollution (blue, green and grey) for the production of a product or for a process.	Identifies different water resources, i.e. blue, green and grey. Blue water scarcity indices aim to address local impacts of blue CWU	WF figures are difficult to relate to other assessments as they combine quantity and quality. No elaborate method for relating green and blue water use to local impacts.	WF has been used for global assessments of the water footprints of various livestock products.
Lifecycle assessments (LCA) and revised water footprints	Calculates consumptive water use along the entire value chain to produce livestock products. The water is assessed in relation to local water stress in the area where it is used.	Precise results along the entire value chain. Local effect of impacts included in assessments, e.g. blue water scarcity indices and potential eutrophication.	LCA is very data intensive. Normally only assess blue water use, thus excludes green water use.	LCA has been used to assess livestock water use in relation to water stress for different livestock production systems; to prevent increased local water stress and water scarcity.

Water productivity

Water productivity is the ratio of the net benefits from crop, forestry, fishery, livestock and mixed agricultural systems to the amount of water depleted to produce those benefits. The benefits can either be measured as the physical agricultural outputs or the economic value of these outputs. The amount of water depleted is either; the consumptive use during production, the water quantity incorporated in a product, water flows to a location where it cannot be readily reused, or heavily polluted water quantities not available for further use (Molden et al., 2010).

Studies that specifically consider livestock water productivity (e.g. Peden et al., 2007) build on previous water accounting studies (e.g. Kijne et al., 2003; Molden et al., 2007a; 2007b) and water productivity research (e.g. Pimentel et al., 1997) developed to provide information about how to improve water use in agriculture. Most of the papers focusing on livestock water productivity assess the water productivity against the total consumptive water use, and the generated figures have contributed to the debate on the differences in environmental impact between industrial agricultural production and smallholder production in developing countries (Peden et al., 2007).

Livestock water productivity can be used to assess the efficiency of water resource use and to identify possible efficiency gains, and can be calculated for a product, an entire production system or a specific area, such as a river basin. Generally, the ratio of total evapotranspiration during production, of both green and blue water, to the amount of livestock produced or benefits is calculated and expressed in terms of kg of product or monetary outputs per m^3 or litre of depleted water (see Table 2). Assessments for livestock have been applied at many different scales, and for different livestock products and production systems (Bossio, 2009; Haileslassie et al., 2009; Bossio et al., 2010; Descheemaeker et al., 2010). However, most studies are generally limited to a specific area and a limited number of crops (Mekonnen and Hoekstra, 2013). Many studies focus on mixed crop-livestock systems in sub-Saharan Africa at the farm-level scale, and some include a wide range of multiple benefits such as livestock produce, draft power, manure, transport, nutrient cycling and socio-cultural value (Descheemaeker et al., 2010; Kebebe et al., 2015).

Livestock water productivity results are generally presented as benefit produced divided by an aggregated CWU figure, without distinguishing between green and blue water resources. As Table 2 illustrates, for beef meat production the results display a large variation and range from 14,286 to 200,000 L/kg of meat. The climatic condition in a region of production and the methods for assessing evapotranspiration explain a large part of the variation between studies, since evapotranspiration can vary hugely between different areas and climates across the world. The variation in livestock water productivity for beef can also be explained by differences in feed quality, digestibility, and feed conversion efficiency between production systems. For example, the study by van Breugel et al. (2010) considers smallholder cattle and small ruminant systems in different regions of Africa, which have low to very low meat production per animal. This results in significantly higher CWU per kg of livestock output compared to intensive systems utilizing biomass with higher digestibility and better-performing breeds. Consequently, studies of such intensive systems as by Ran et al. (2013) and Molden et al. (2007a; 2007b) present much lower estimates, ranging from 10,000 to 30,000 L/kg beef.

Water productivity studies often include multiple benefits from livestock, by accounting for livestock outputs in kilograms or the economic value of livestock outputs divided by the amount of depleted water (Rockström et al., 2007; Descheemaeker et al., 2010; Molden et al., 2010; Kebebe et al., 2015). Haileslassie et al. (2009), for example, relate water use to the economic value of animal source foods and the manure used to produce biofuel or to fertilize crops. The inclusion of the multiple functions of livestock, however, is not limited to this method, but can also be applied in water footprint and LCA studies.

The possibility to compare different livestock benefits related to their corresponding CWU enables farmers, and other water resource managers, to make rational decisions about management changes based on the connections between production benefits and specific resource use. In most cases water productivity estimates are presented as a summarized value without distinguishing between green and blue water resources, making it difficult to optimize water resource uses of blue and green water resources based on this method. Water productivity results also do not provide guidance on how livestock water uses affect competitive uses for the water resources unrelated to livestock outputs, such as the production of food and fibre, or other ecosystem services and functions like maintaining soil fertility. Other limitations of using a comparative unit of benefit in terms of monetary value include the problem of assigning exact monetary values to livestock outputs such as draft power, and the fact that in smallholder systems animals constitute insurance against difficult periods such as drought. Finally, water productivity studies do not distinguish between the quantity of water used and the quantity of water polluted (Mekonnen and Hoekstra, 2013).

Water footprint

In this review, we assess the water footprint approach developed within the Water Footprint Network (Hoekstra et al., 2011). The water footprint is defined as the total amount of water required to produce a product or service during all or part of the product's life cycle and constitutes of a summarized value of green, blue and gray water quantities. The methodology can be applied at different scales and has even been used to quantify the "water footprint of humanity" (Hoekstra and Mekonnen, 2012). It is not a traditional water accounting approach, as it combines actual CWU estimates of green and blue water with an assumed requirement for blue water, denoted gray water, to dilute an estimated pollution caused during the production of a product, as if this pollution were being released into a recipient. Since it is difficult to assess the actual water volume needed to dilute multiple polluting substances, the gray water estimate is often calculated only as the water volume needed to dilute nitrogen leaching to the maximum allowable concentration in free flowing surface water bodies (Hoekstra et al., 2011).

A coherent water footprint approach was first introduced in the early 2000s (Hoekstra and Huynen, 2002) to inform companies and consumers about the pressure on water resources in the production of different products, including animal source foods (Chapagain and Hoekstra, 2003; Mekonnen and Hoekstra, 2010a, 2012). The water footprint concept has gained wide approval among different organizations and companies, and has, for example, been adopted by the World Wildlife Fund (WWF, 2010) in striving to reduce the global human pressure on limited water resources.

The estimates obtained from water footprint studies in Table 2 show water footprints either as global averages or according to the three water resource categories: green, blue and gray. The global averages show very small variations, from 15,415–15,497 L/kg of beef, including green, blue and gray water estimates. In contrast, the study by Mekonnen and Hoekstra (2012) presents relatively large variations between both regions and production systems. Grazing systems, for example, have a range of 16,353–26,155 L/kg of beef, mixed systems of 11,744–16,869 L/kg of beef and industrial systems of 3,856–13,089 L/kg of beef.

The green and blue water footprint estimates are within the same range of other studies that include total CWU of crops and grass feeds (e.g. Molden et al., 2007a; 2007b; Deutsch et al., 2010; Ran et al., 2013), apart from water productivity studies that consider low-productivity smallholder systems (e.g. van Breugel et al., 2010) which generally result in higher values. Studies that include both green and blue water use, however, show significantly higher values than studies that exclude green water use that is not the result of irrigation, such as the LCA studies on beef (e.g. Ridoutt et al. 2012a; Zonderland-Thomassen et al., 2014). The gray water footprint, or equivalent measure of water quality, is not calculated in any of the reviewed studies apart from water footprint studies. Thus, the results for gray water assessments are not comparable.

To make the footprint a relevant measure of local environmental impact, a water footprint sustainability assessment was developed in the most recent water footprint standard (Hoekstra et al., 2011). All three contributing water resources are viewed separately and have their own impact assessment. A water scarcity index relates blue water footprints to local blue water availability and the impact assessment of a gray water footprint is related to the local waste assimilation capacity. It suggests estimating the green water scarcity index as the ratio of the green water use for a defined area (and crop or product) to the effective rainfall, the total evapotranspiration of rainwater from land minus evapotranspiration for natural vegetation and the amount that cannot be made productive, over the same area. It should be noted that a green water scarcity index has not yet been applied in a water footprint network publication, due to the difficulty in obtaining data on effective rainfall (Hoekstra et al., 2011). In addition, many water footprint studies are not spatially connected to local characteristics and thus local impacts in the landscape.

Water footprint estimates differ from water productivity assessments in several ways. One obvious difference is that water footprint values are presented inversely to water productivity, that is, as a water quantity per benefit and not a benefit per water quantity. Another major difference is the water quantities that are included. The water footprint approach is built up using separate estimates for blue and green water use, although they are aggregated in the final footprint. Water productivity generally uses total CWU without separating green and blue water. The most important methodological difference between water productivity and water footprints is that water productivity studies only include actual depleted water quantities, while the water footprint methodology combines the CWU with a theoretical estimate of the gray water volume required to dilute the load of pollutants generated during production of a product or service. Thus, even if the values from each methodology are inverted in order to be presented next to each other, the figures are not comparable, as recent water footprint studies generally include gray water figures.

LCA and revised water footprints

LCA studies aim to assess the environmental impact of a product along the entire value chain, and to quantify that impact based on the location of resource use. The methodology presents a direct connection to local environmental impacts by calculating water use based on stress water indices. Existing LCA studies of water use in livestock production focus mainly on consumptive and degradative blue water use along the entire production chain and their associated contributions to local water stress. These LCA studies include spatial information about water scarcity, resulting in water stress-related water footprints (Pfister et al., 2009) referred to as LCA revised water footprints. LCA revised water footprints directly couple consumptive blue water use to local blue water scarcity indices to give spatial environmental relevance to the water resource outtake (Ridoutt and Pfister, 2010; Zonderland-Thomassen and Ledgard, 2012; de Boer et al., 2013; Ridoutt and Pfister, 2013). These footprints can also be used to assess and measure water quality by quantifying eutrophication and the ecotoxicity potential of a product along the value chain (de Boer et al., 2013d).

A number of studies compare LCA revised water footprints with standard water footprints for different products (Ridoutt et al., 2009; Zonderland-Thomassen and Ledgard, 2012; Sultana et al., 2014). These generally argue that green and blue water should not be given equal importance. The focus should be on blue water resources, due to the larger local implications in, for example, water stressed areas. In addition, a few attempts have been made to link green water flows to environmental impact. Núñez et al. (2013b), for example, use the same principle proposed by the water footprint network (Hoekstra et al., 2011). The exclusion of green water in LCA livestock-water estimates, and that assessments account for water stress, means that LCA results generally fall well below other estimates based on total evapotranspiration for the production of animal feed (green and blue water use), as can be seen in Table 2.

Only accounting for blue water use that contributes to local water stress results in a lower figure than accounting for all water resource use even if it does not appear to affect or cause water stress. This is also the explanation for the relatively large variations in the results from LCA studies, ranging from 0.18 to 117 L of H2O-equivalent/kg of beef. The higher estimates are from Australian beef production, where the water stress is significantly higher than in regions with low water stress, such as New Zealand. This result in a higher consumptive water use per kg of beef produced (Zonderland-Thomassen et al., 2014).

By excluding the major part of consumptive green water use, the LCA approach does not capture the major water use related to livestock, i.e. green consumptive water use. This is a shortcoming if the intention is to analyse total water efficiency in agriculture, or in livestock production in particular, including how to allocate water between other competing production and ecosystem services, such as the production of food crops instead of animal feed crops.

Table 2: Water use estimates for meat production for different methods (in litres of water/H_2Oe per kg meat)

Reference	Aggregated value of green and blue water	Green	Blue	Grey	Data collection and analysis	Spatial scale and production system
	Water productivity					
Pimentel et al. (1997); Pimentel et al. (2004)	100,000–200,000 l/kg				Uses rough estimates of grassland ET from Falkenmark (1994)	Beef production
van Breugel et al. (2010)[b]	All systems except MI HYP* (14,286 l/kg) had a water productivity of 100,000 l/kg or more				Livestock model for livestock feed requirements developed by Herrero et al. (2008) and van Breugel et al. (2010); water for feed crops computed using FAO data (Allen et al., 1998); rangeland evapotranspiration calculated using actual evapotranspiration GIS layers (Ahn and Tateishi, 1994).	Cattle, sheep and goat production systems across the Nile basin
Molden et al. (2007a)	Varied between 10,000 and 3,300				Presents results from the literature	Beef production
Haileslassie et al. (2009)	0.42USD/m^3				Regional and agricultural survey data used for livestock produce. Penman-Monteith method used to estimate observed ET and reference ET from FAO (Allen et al., 1998)	Cattle farming systems in Ethiopia
	Revised water productivity					
Ran et al. (2013)[c]		18,900 l/Kg (I); 15,700 l/kg (M); 19,300 l/Kg (E)	42 l/Kg (I); 7 l/Kg (M); 0 l/kg (E)		Hydrological modelling with LPJmL (Bondeau et al., 2007); field interviews	Extensive, mixed and intensive conceptual beef production systems in Uruguay
Deutsch et al. (2010)	12,000 l/kg (zero allocation for grazing); or 40,500 l/kg (allocation for grazing)				Uses a water productivity approach but develops a matrix to illustrate ecosystem effects of hydrological alterations due to livestock production. Allocation of ET from grazing in intensive and mixed systems is based on share of feed basket. For grazing systems, allocation of ET is determined by whether grazing sustains or enhances other ecosystem services (zero allocation), or negatively affects other ecosystem services (full allocation)	Average of three conceptual beef production systems; intensive, mixed and grazing
	Water footprints					

Mekonnen and Hoekstra (2012)[d]	15,415 l/kg	China: I:10,922, M: 13,227, G:16,140; India: I: 12,412, M: 16,192, G: 25,913; Netherlands: I: 3,934, M: 10,319; USA: I: 2,949, M: 12,726, G:19,102 l/Kg	China: G: 213, M: 339, I: 933; India: G: 242, M: 533, I: 1,471; the Netherlands: M: 349, I: 761; USA: G: 525, M: 546, I: 356 l/Kg	China: 1,234; India: 0–866; Netherlands: 225–664; USA: 243–712 l/Kg	Water footprint method; feed basket composition based on FAOSTAT (http://faostat3.fao.org/home/E), Hendy et al. (1995) and Wheeler et al. (1981)	Global average. Beef production
Hoekstra and Chapagain (2007)	15,497 l/kg				Water footprint method; feed basket composition based on FAOSTAT (http://faostat3.fao.org/home/E), Hendy et al. (1995) and Wheeler et al. (1981)	Global average. Beef production
	LCA and revised water footprints					
Ridoutt et al. (2012a)[c]	12–117 l H_2Oe/kg				This study uses the methodology developed by Ridoutt and Pfister (2010), creating an inventory of CWU to relate production system water use to freshwater availability. The revised water stress-related water footprint was used in this example (Ridoutt and Pfister, 2010).	Geographically different beef production systems in Australia
Zonderland-Thomassen et al. (2014)[e]	0.18 l H_2Oe /kg (average New Zealand water scarcity footprint)				Uses a water scarcity-related water footprint based on the method developed by Pfister et al. (2009). Secondary data consulted where primary data not available.	Compares different beef production systems in New Zealand

[a] All water productivity studies have been inverted to enable easier comparison with the remaining studies using the same format. Revised water productivity studies were already calculated in litres of water per unit of output.
[b] Studies climatic variations—hyper arid (HYP), arid (A), temperate (T) and humid (H)—of three different production systems—grazing (G), mixed irrigated (MI) and mixed farming (MR)—for cattle, goats and sheep. For mixed systems with * water for residues were not included in the calculations
[c] Studies three different systems, intensive (I), mixed (M) and extensive (E), referring to the feed composition for each system
[d] Studies three different systems, industrial (I), mixed (M) and grazing (G), referring to the production system
[e] Recalculated as kg beef instead of kg live weight based on FAO assumptions of carcass weight as a percentage of live weight (FAO, 2003)

3.3 General discussion

As noted above, this review groups existing approaches to assessing water resource use in livestock production into three method categories: water productivity assessments, water footprint assessments and life cycle assessments. One main finding of the review is that the methods for calculating water use for livestock vary in different ways and thus often yield dissimilar insights regarding water use in livestock production systems.

Many of the disparities between methods are the result of differences in intended use and purpose, which also makes it difficult to compare results. For example, water productivity assessments have been developed to highlight areas of possible productivity increase to produce more crops from the same amount of water. Livestock water productivity studies have mainly focused on improving the water productivity of smallholder farmers in water-scarce areas. In contrast, the water footprint methodology aims to provide insights about the pressure on water resources caused by production of goods or services and presenting figures that are easy to comprehend for decision makers and consumers. LCA assessments focus on linking assessments of water use to impacts on the local environment, such as increasing water scarcity. Because the methods were developed for such widely different purposes, they should be expected to answer different questions, even while investigating the same resource use issue. This increases the difficulty of comparing results and harmonizing methods, since the differences are embedded in methodological choices not evident to policymakers, consumers or researchers. This discussion highlights a number of key methodological choices that might induce such differences between the results of studies of livestock water use.

3.3.1 Accounting for water quality

Many of the methods and studies reviewed aim, at least to some extent, to measure both water quantity and water quality. The water productivity method refers to the assessment of water quality as one type of depleted water, that is, a water quantity too degraded to be available for further use (Molden et al., 2007b). The water footprint method calculates the gray water footprint, which refers to the volume of water assessed to be required to dilute pollutants (Hoekstra, 2009, 2010; Mekonnen and Hoekstra, 2010a, 2012). This approach has been criticized on three counts. First, the quantity of water needed to dilute pollutants depends on downstream user needs, e.g. if the water will be used for hydropower or serve as drinking water. Second, there is no standardized measure of water quality, so the quantity required to dilute pollutants will differ according to what is defined as acceptable water quality (Perry, 2014). Finally, as described above, the gray water footprint is a virtual water amount not a consumptive water use, which makes interpreting water footprint figures problematic from a water quality perspective, as well as from a water quantity perspective. LCA methods generally use indicators of eutrophication and eco-toxicity to measure water quality, thus, focus water use assessment on consumptive water use (Milà i Canals et al., 2009).

However, actual assessments of water quality are highly data intensive and thus often difficult to measure, which results in generalizations and the use of simplified methods in many studies. For example, Mekonnen and Hoekstra (2012) assess the gray water footprint of farm animals, but only in relation to nitrogen leakage, which means that all the other nutrients, such as phosphorous, or chemicals used in the product value chain are not considered. Even many LCA studies fail to fully capture the complexity of water quality and the effects of all polluting substances that can cause water deprivation (Kounina et al., 2013). The integration of water quality indicators, such as eco-toxicology potential, is generally lacking due to problems with data availability (de Vries and de Boer, 2010).

3.3.2 The importance of including green water use

A key divide between the methods reviewed is the inclusion of green water resources. Table 2 shows that most of the variation in results is related to whether the methods include green water use in their assessments. The argument for excluding the majority of green water resources, over crop and grasslands, is that the consumptive green water use lacks a direct connection to local impacts, such as water scarcity (Ridoutt and Pfister, 2009). Another argument for excluding green water is that a large amount of the consumptive green water use is for the production of non-human edible biomass, such as grass, produced on non-cultivated fodder land and grassland. It is argued that the water evapotranspiration over these grasslands would be required to support grass growth regardless of whether animals were grazing it (Steinfeld et al., 2006; Peden et al., 2007; Deutsch et al., 2010).

However, the exclusion of all green water resources means that the potential use of that water for other competing purposes is not accounted for. Certain grasslands have alternative uses, and green water resources could support the production of food, fuel, fibre or other provisioning and regulating ecosystem services instead of grass growth for livestock grazing. These opportunity costs of green water resources are not captured if green water is excluded from CWU assessments. In addition, consumptive green water use can severely affect water partitioning, altering short- and long-term soil water and water availability in the landscape. This alteration of water availability should be considered an impact on the functioning of that particular ecosystem (Milà i Canals et al., 2009).

Green water availability is also closely linked to other resource use, because land use and land cover change affect the soil moisture (Kounina et al., 2013). The efficiency of green water use is of interest with regard to sustainable intensification to meet the increasing global demand for food. More efficient green water use implies a reduced need for additional blue water resources, in terms of irrigation, or to expand rainfed crops into other terrestrial ecosystems to appropriate additional green water resources (Molden, 2007b; Rockström et al., 2007).

Water footprints that include all green water resources, however, have been criticized for generalizing water requirements when summing all three water resources into a single final water footprint value (Ridoutt and Huang, 2012; Ridoutt et al., 2012b; Perry et al., 2013; Perry, 2014). Critics argue that consumers are faced with a measure that provides no information on whether a product has a large water footprint due to its high blue CWU, its high green CWU, or assumptions about its severe impact on water pollution. The spatial scale of assessment is still generally focused on higher levels: the national or global (Hoekstra et al., 2016). Only four of the 33 studies published in 2014–2015 focused on a lower spatial scale (waterfootprint.org/publications), which means that the local impacts of assessed CWU are not considered appropriately.

The LCA network has provided input into and assisted with the development of an ISO standardized method for water footprint assessments that considers both water consumption and the pollution of water resources (Ridoutt and Pfister, 2013). However, the method developed (ISO, 2014) does not account for the majority of green CWU, treating it as an indicator of land use rather than water use (Pfister and Ridoutt, 2014). There have been attempts within the LCA community to account for the use of green water (Milà i Canals et al., 2009; Núñez et al., 2013b) and for green water scarcity (Núñez et al., 2013a), arguing that it is crucial to further integrate quantitative green water assessments into LCA for systems where dependency on green water resources is high. However, Pfister and Ridoutt (2014) state that the methodology should not be used for LCA water footprints and indicate that consumptive green water use should be incorporated only if it can be directly linked to causing human or environmental harm (Ridoutt et al., 2009). For example, green water evapotranspired on irrigated land should be considered, since it can be directly linked to water stress as opposed to green water evapotranspired over purely rainfed crop and grasslands.

Accounting for the competition for water resources between, for example, the provision of feed and food, is relevant to the sustainable intensification of agricultural systems. Sustainable intensification implies not only improving agriculture and livestock productivity per ha, but also increasing the number of human beings nourished per ha (van Zanten et al., 2016). In other words, the use of green water resources should be seen from a competition perspective when analysing the current trend for an increase in the global demand for animal-source foods (Steinfeld et al., 2006; de Fraiture et al., 2007).

3.3.3 Consideration of environmental impacts and other potential uses

The real impacts on ecosystem functioning and ecosystem services are not captured if only quantitative measures, such as the CWU associated with the production of animal feed, or from a water footprint estimate, are considered. Agricultural management practices in combination with local system characteristics will affect the quantity of water required for crop or pasture growth, and could also alter hydrological features on multiple scales, affecting a number of water-related ecosystem functions and services (Falkenmark and Rockström, 2004; Bossio et al., 2007; Gordon et al., 2008; Bossio et al., 2010; Deutsch et al., 2010; Keys et al., 2012b; Ran et al., 2013).

This review found that several methods include environmental impacts of livestock water consumption, for example, the relationship to local water scarcity in the revised water footprints and LCA assessments (Pfister et al., 2009; Descheemaeker et al., 2010; Ridoutt and Pfister, 2013). However, most approaches fail to calculate the potential loss of other ecosystem services beyond agricultural production. The exception among the reviewed methods is Deutsch et al. (2010). In this paper, CWU from pastures in grazing systems is not accounted for when grazing is assumed to sustain or enhance other ecosystem services. When grazing is assumed to dominate the function of the system at the expense of other ecosystem services, however, the CWU from pastures is entirely allocated to the grazing system. Thus, the study considers competition in the form of preventing the provision of ecosystem services, focusing on ecosystem degradation. However, the suitability of using the resources for grazing as opposed to other potential uses, such as the provision of food, fuel or fibre, is not considered if the grazing system is assumed to be sustainable. In other words, the method enables resource competition to be included in the assessment but does not substantially cover the potential for the water and land resources to be used for other applications than the current one.

Differences and trade-offs

The three categories of methods reviewed have all made important contributions to the development of water resources use assessments for livestock. The water productivity methodology is the only approach that quantifies multiple benefits of livestock, not merely for animal sourced food, and identifies potential water efficiency gains in smallholder agricultural systems. Water footprints have raised awareness of the large amounts of water required for livestock production and consumption of livestock products. LCAs highlight the importance of connecting water resource use to local impacts and local water stress. All three methods, however, display certain limitations. For example, water productivity studies refer to total CWU, thus loose relevance to management options and impacts related to blue and green CWU, whereas both water productivity studies and many water footprints studies have no sophisticated connection to local environmental impacts and LCA studies do not include the majority of consumptive green water use. Finally, all of the three methods lack a clear connection to landscape interactions and hydrological basin system dynamics. The potentially important role that both green and blue water resources could play in improving agricultural productivity and ecosystem functioning is not properly captured by any of the methods reviewed. All but one of the studies fails to include competition for resources or the different outputs and benefits to the landscape (Deutsch et al., 2010).

Methodological opportunities

This review has identified a number of key aspects that should be considered when assessing water use in livestock production. First, assessments should include both green and blue water resources. However, it is crucial that the results are presented separately, because they have different alternative uses, causes different environmental impacts and play different roles in causing water scarcity, for competing uses and in sustaining different aquatic and terrestrial ecosystem services. By keeping green and blue water resources separate, it is possible to identify the complementary roles they can play. For instance, a small addition of blue water to a system that is highly dependent on green water can significantly increase crop water productivity, and thus also livestock water productivity, and make unproductive green water flows productive. This is verified by the findings of a review of freshwater assessment methods (Kounina et al., 2013), which calls for methods to fill the knowledge gap on quantifying the link between green water use, and the identification of indicators to characterize the relationship between green water resources and land use.

Second, gray water measures, which are a virtual water proxy for the amount of water required to assimilate pollutants and abate water quality degeneration, should not be summed with consumptive green and blue water uses. That does not mean that water quality should not be addressed, which is vital in terms of water quality indicators. It is merely to point out that such results should not be presented together with quantitative data on green and blue water resource use.

Third, livestock-water resource assessments need to consider the competition for the use of different water resources and to highlight the importance of green water use in agriculture. The competition for water resources between the production of human food or animal feed can occur in two ways: directly, where animals are fed crops and crop products that can be directly consumed by humans; and indirectly, where animal fodder and grazing are produced on land and using water that is also suitable for the production of food crops, forestry, energy crops or other ecosystem services. Animal feed production can also be produced without causing any increase in competition with other production, for example, in systems where animals are grazed on marginal land that has few alternative uses and little socio-ecological value.

Methods can take account of direct resource competition by considering green and blue water coupled with their respective land uses, as in the assessments by Ran et al. (2013); and identifying the direct competition between human food production directly from crops or production through livestock keeping. In order to provide useful results, however, livestock CWU assessments also need to address the potential for indirect competition with the production of fuel, fibre and other ecosystem services and socio-economic values. Such an assessment method would successfully capture the environmental impact and the ecosystem functioning related to consumptive water use for livestock production.

3.4 Conclusions

Water resources are a limiting factor in the ability to feed a growing world population. Livestock production systems are a major contributor to the world's food systems and a large consumer of water resources. This study reviewed existing methods of assessing livestock water use, recognizing water as a limited resource in global agriculture.

Existing methods for assessing water use in livestock systems were classified into three categories: water productivity assessments, water footprint assessments and life cycle assessments. Methodological differences and differences in the intended uses of the methods hamper the interpretation and comparison of results. The review identified three key methodological points that would improve assessments of freshwater use in livestock production.

First, water resource use assessments should include the use of green water resources, thus assessing and recognizing the importance of green water. Blue and green water resources should be presented separately in order to achieve policy relevant results and to identify improvement options.

Second, gray water is a water quality measure generally calculated as a proxy of the volume of water required to abate pollution, it should not be summed with blue and green water, which are quantitative measures to account for consumptive water use.

Third, for assessments to be useful to consumers, producers, policymakers and decision makers, the competition for water resources between users and the local environmental impact of consumptive water use need to be taken into account.

The link between consumptive water use and the impact on local ecosystems is not properly captured in current methods for assessing water use in livestock systems. Considering the competition for water resources between local users is an imperative contribution to sustainable intensification of livestock production, and of the agricultural sector as a whole. Sustainable intensification implies improving the number of human beings that can be fed per unit of resource, such as water, rather than simply increasing livestock productivity. In order to do this, methods for assessing water use in livestock systems must consider the alternative uses, multiple uses and benefits of a certain resource in a specific location.

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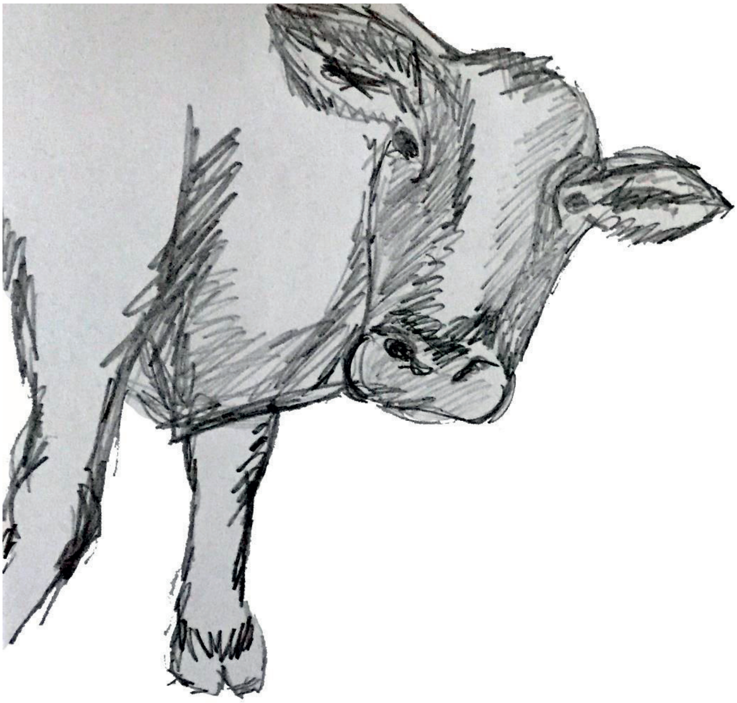

Chapter 4

Freshwater use in livestock production – To be used for food crops or livestock feed?

Ylva Ran[a,b], Corina E. van Middelaar[a], Mats Lannerstad[c,2], Mario Herrero[d], Imke J.M. de Boer[a]

[a] Animal Production Systems Group, Wageningen University, PO Box 338, 6700 AH Wageningen, The Netherlands

[b] Stockholm Environment Institute, Linnégatan 87D, 11523 Stockholm, Sweden

[c] International Livestock Research Institute, PO Box 30709, 00100 Nairobi, Kenya

[d] Commonwealth Scientific and Industrial Research Organization, Bioscience Precinct Carmody RD, St. Lucia, 4067, Queensland, Australia

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[2] Present address: Welanders väg 7, 112 50 Stockholm, Sweden.

Abstract

Current approaches to estimate freshwater use in livestock production systems generally fail to consider the competition for water resources with alternative uses, such as production of food crops food or other ecosystem services. This article presents a new method to account for the competition for freshwater use between food crops and animal feed, while assessing freshwater use in livestock production systems. The developed water use ratio (WUR) is defined as the maximum amount of human digestible protein (HDP) derived from food crops from the consumptive water use (CWU) appropriated to produce 1 kg of animal-source food (ASF) over the amount of HDP in that 1 kg of ASF. The CWU for livestock production is first categorized according to the land over which it is consumed, based on the suitability of that land to produce food crops. Then, the method assesses feed-food competition by determining the amount of HDP that could have been produced from food crops, using the same CWU currently used to produce ASF. The method enables identification of livestock production systems that contribute to global food supply without competing significantly over water resources with food production, based on their CWU. Three beef production systems in Uruguay are used to illustrate the method. During the backgrounding and the finishing stages, which are analysed in this study, cattle can be kept on natural pasture (NP), seeded pasture (SP) or in feedlots (FL). The following three systems were analysed: i) NP-NP, ii) SP-SP and iii) SP-FL. Results show that the NP-NP system uses the largest amount of water per kg of beef output. However, results also show that the SP-SP and SP-FL systems can potentially produce more HDP by growing food crops than by producing beef. Based on the traditional measure for water productivity, i.e. the quantity of CWU per kilo of beef produced, we would conclude that the NP-NP system is least efficient, whereas based on the WUR the NP-NP system is the only system producing HDP more efficiently than food crops. Sustainable intensification not only implies improving agriculture and livestock productivity per unit of resource used, but also improving the number of human beings nourished. Results from this study illustrate the importance of considering competition and trade-offs with other uses when evaluating water use efficiency of livestock systems to promote sustainable intensification.

4.1 Introduction

A growing world population, estimated to reach nine billion people by 2050, is increasing the pressure on global agricultural production to ensure food security for all. Between 2005 and 2050 the demand for meat and milk products is projected to increase by around 70–80% and the demand for crop protein by 100–120% (Tilman et al., 2011; Alexandratos and Bruinsma, 2012).

Livestock production requires large amounts of natural resources, including water and land, and the expected rising demand for animal sourced foods (ASF) can potentially amplify environmental impacts related to livestock (Delgado et al., 1999; Godfray et al., 2010; Bouwman et al., 2013; Westhoek et al., 2014; Herrero et al., 2015).

At present, the global livestock sector uses about 75% of all agricultural land (Foley et al., 2011), and is responsible for about 30% of global agricultural water requirements, including rain and irrigation water used for production of feed and withdrawals for animal husbandry (Mekonnen and Hoekstra, 2012). At current productivity levels, the expected rise in demand for animal products will result in a doubling of the land and freshwater requirement, increasing the water resource use competition (Rockström and Barron, 2007; Rockström et al., 2007). An amplified water use for livestock and crop production can, in turn, locally increase the risk of water stress (Ridoutt and Pfister, 2010). At present, more than 1.2 billion people already suffer conditions of physical water scarcity (Molden, 2007a).

Livestock require water for e.g. drinking and cleaning services, and for the cultivation of feed crops or for grass growth (Figures 1 and 2). In this paper, we focus on consumptive water use (CWU), which refers to water that is withdrawn from a watershed, and not discharged to the same watershed because it evaporates, is embodied in plants or animals, or is discharged to a different watershed (Falkenmark and Lannerstad, 2005). As a general rule, > 98% of the total CWU in livestock production can be attributed to evapotranspiration from feed crops and pastures. Only 2–8% of livestock CWU is drinking, servicing and feed-mixing water (Steinfeld et al., 2006; Mekonnen and Hoekstra, 2012; De Boer et al., 2013).

To acknowledge the importance of both soil moisture and water withdrawals from water bodies, water resources can be divided into green water, which refers to soil moisture available to plant growth, and blue water, which refers to liquid water in water bodies, as rivers, lakes and aquifers (Falkenmark, 1995). Green and blue water resources, however, are interchangeable states, and water can shift from one state to the other, and back. Green water use does not only affect the availability of soil moisture, but could also affect the availability of blue water, since part of the soil moisture, if unused, could drain out of the soil and re-charge water bodies as blue water. Thus, both green and blue water uses may ultimately alter water availability in the landscape in different ways, impacting local ecosystem functioning and, should therefore both be considered in water use assessments (Milà i Canals et al., 2009).

As illustrated in Figure 1, livestock products, e.g. beef meat, can be produced in a variety of production systems that use a wide range of different feeds, which in turn can be grown using different natural resources and management practices. The use of water resources, and primarily green water, is tightly connected to the land that is used by a particular livestock production system. Green water is directly linked to a specific area, available as soil moisture for plant growth, while blue water is linked to water bodies, thus the ability in the landscape to store liquid water. Since the majority of water consumption in livestock systems relates to the cultivation of feed, water resource use and land use should be considered together, rather than separately (Ran et al., 2016).

Animal feed can be produced on grasslands such as natural pastures (grazing livestock) and cropland (all livestock). Grasslands, especially natural pastures, require primarily green water. However, some pastures are irrigated, thus using additional blue water resources, and some are even cultivated and occupy land suitable as cropland. All animal feed crops require cropland for growth, however, some feed crops are rainfed, and thus depend entirely on green water, while others require irrigation water depend on both green and blue water.

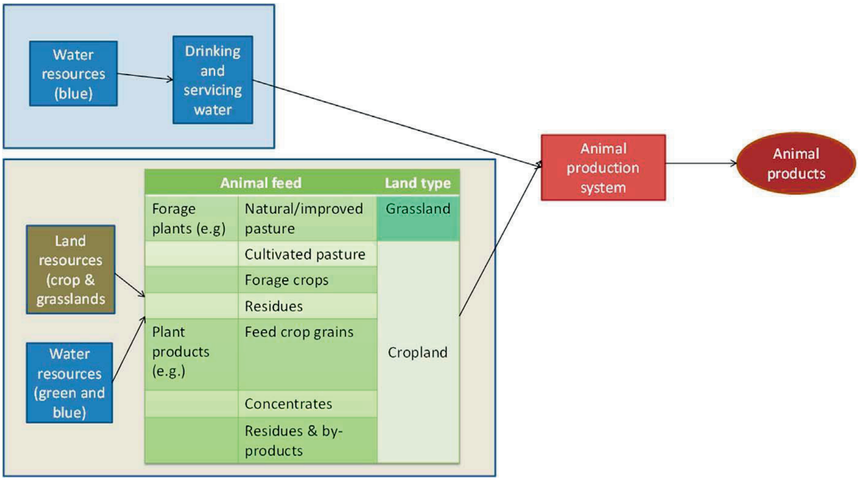

Figure 1. Conceptual flow chart of land and water resource requirement in livestock production.

To prevent unsustainable use and management of water resources, there is a need to describe the linkages between livestock production and freshwater use. Understanding and quantification of these links is imperative in order to increase water productivity in livestock production, and to identify trade-offs and synergies between livestock production and other competing water uses, such as food crop production. The focus on increased feed efficiency for livestock to improve resource use efficiencies, and changing consumer preferences towards more pork and poultry products, has led to a larger share of human edible plant material in animal feed (De Vries and De Boer, 2010; Eisler et al., 2014). An increased use of high-quality croplands to cultivate animal feed, in preference to food crops, will further proliferate resource use competition between food and feed production.

Current estimates of both water and land resource use by livestock generally fail to consider the competition for resources between the production of food crops and animal feed (van Zanten et al., 2016). To address such knowledge gaps, van Zanten et al. (2016) developed a method that accounts for the competition for land resources between food and feed production. Based on a land use ratio (LUR), the land use efficiency of livestock systems is defined as the maximum amount of human digestible protein (HDP) derived from food crops on all land used to cultivate feed required to produce 1 kg of ASF, over the amount of HDP in that 1 kg of ASF.

Considering that livestock production systems, in addition to land, use large amounts of water resources we further develop the method presented by van Zanten et al. (2016) to investigate feed-food competition over water resources in livestock production systems. This requires careful consideration of the complexity of the hydrological cycle, recognising that water is a dynamic resource with a strong connection to landscape dynamics and multiple users competing for its availability.

The method presented in this study, focuses on water resource use in livestock production systems and the competition between food and feed production. Different from other water assessment studies, water resources are categorized with regard to land use, taking into account the opportunity costs of land for food crop production. In this way livestock production is compared to food crop production based on their contribution to the production of human digestible proteins per unit of water resource used. The method enables identification of livestock production systems that contribute to global food supply without competing significantly over water resources with food production. In this paper, the method is first described in generic terms and subsequently illustrated, using three beef production systems in Uruguay.

4.2 Conceptualization of the method

4.2.1 Generic description of the method

The developed method is illustrated in a flowchart in Figure 2. The method calculates CWU during plant growth of feed crops on cultivated land and grass growth on pastures. Water used for feed production is either green water, i.e. rainwater on crop or grasslands, or blue water, e.g. groundwater or surface water used for irrigation of primarily cropland (Figure 2). Cropland and associated green and blue water resources can be used directly to cultivate feed crops, food crops or other crops (e.g. fuel or fibre), whereas water resources used on grassland that is suitable for crop production could provide animal feed as grass but could also support crop growth. Thus, water use for feed production on cropland competes directly with food crop production, whereas water use on grasslands suitable for crop growth illustrates indirect feed-food competition (Figure 2). The developed method calculates and differentiates the CWU between water resources evapotranspired over land suitable for crop production, and land that is assumed to be unsuitable for crop production.

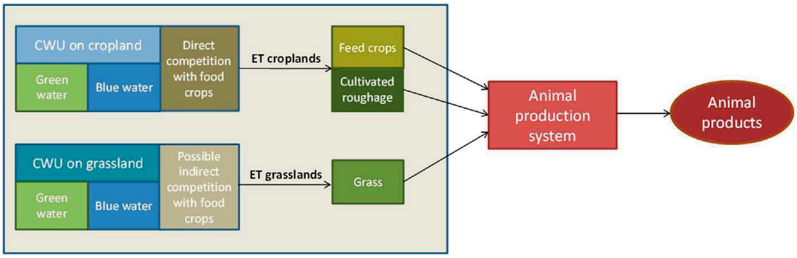

Figure 2. Water use in livestock production categorized, considering differences by feed composition for different animal type and production systems and possible trade-offs between feed and food crops.

The proposed methodology is a four-step process. First, green and blue CWU during production of animal feed is quantified, for example by using a hydrological model or from field measurements. The division into green and blue water highlights to what extent CWU for animal feed constitutes of soil moisture from naturally infiltrated rainfall, and to what extent it is water abstracted from water bodies. Second, the green and blue water required for production of feed is categorized according to the two agricultural land types over which it is evapotranspired, i.e. croplands and grasslands (see Figure 2). Third, the opportunity cost of land and water resources, with regard to feed-food competition, is identified by assessing the suitability of the land and water resources to produce food crops. The green and blue CWU on crop land could have been used directly to produce food crops on that land and represent direct feed-food competition over water resources. Indirect competition refers to the CWU over grasslands that are partly or fully suitable to support crop cultivation. The suitability can be assessed by using statistical data, like the global agroecological data base (FAO, 2016; van Zanten et al., 2016), or by field observations. Finally, the water use ratio (WUR) is calculated to provide a measure of how efficient a production system uses water resources to produce HDP comparing food crops against livestock products.

4.2.2 Water use ratio

The WUR is calculated according to Equation (1):

$$\frac{\sum_{i=1}^{n} \sum_{j=1}^{m} (CWU_{ij} \times \mathrm{HDP_j\ m^{-3}\ y^{-1}})}{\mathrm{HDP\ of\ one\ kg\ of\ ASF}}$$

Eq.1

CWUij is the consumptive water use in m^3, evapotranspired over land suitable to produce food crops that is required to produce feed ingredient i (i=1,n) in country j (j=1,m) used to produce one kg of ASF. HDPj is the amount of human digestible protein (HDP) that can be produced in country j, using the same water resources, by direct cultivation of suitable food crops in country j per year. The denominator is the amount of HDP of one kg of ASF. A ratio larger than 1 implies that the water resources for that production system can generate a larger amount of HDP by producing food crops instead of ASF. Correspondingly, if the ratio is below 1 the production of

HDP through livestock is more efficient than cultivating food crops using the same water resources.

4.2.3 Case study description: Uruguayan beef production

Three beef production systems in the Rocha region in the southeast of Uruguay are used as a case study to illustrate the new method, with data adapted from previous studies (Modernel et al., 2013; Ran et al., 2013; Picasso et al., 2014). The region is a good representation of Uruguay, with 78% of the land use dedicated to beef compared to 77% for the entire nation (MGAP, 2011; Modernel et al., 2013). For each of the three systems, the two final stages of the production cycle; back-grounding and finishing are analysed (Picasso et al., 2014). During backgrounding and finishing, beef cattle grow from about 150 kg to their final slaughter weight of around 500 kg. The dressing percentage (i.e. carcass weight / live weight × 100%) of beef produced in Uruguay was assumed to be 52% (FAO, 2003). The protein content of beef was assumed to be 17.6 g protein per 100 g of meat, whereas protein digestibility was assumed to be 94% (Young and Pellet, 1994; USDA, 2015). The cow-calf system was not included in this study.

The backgrounding can be based on either natural pasture (NP) or seeded pasture (SP), and the finishing system can be based on NP, SP or a feedlot (FL) system (Picasso et al., 2014). The three analysed systems, each have a different combination of a backgrounding and a finishing systems and are defined as follows; 1) NP-NP, 2) SP-SP, 3) SP-FL (Table 1). The relative area for each system to produce required animal feed was calculated from animal nutritional requirements (NRC, 1996; AFRC, 1993) which are based on initial and final animal weight, daily weight gain, feed composition and nutritional characteristics of forages and concentrates (first reported in Mieres et al. (2004)), as described in Modernel et al. (2013) and Picasso et al. (2014).

In Uruguay, most beef cattle are finished on pasture; only about 10% of the cattle are finished on feedlots. Natural pastures are assumed to be unsuitable for crop production, because agricultural expansion in Uruguay, primarily for soybean production, has reduced the grazing area in the country and pushed grazing animals to marginal lands (Picasso et al., 2014). Seeded pastures are cultivated with a crop-pasture rotation, where a crop is sown at least every fourth year (Modernel et al., 2013), and, therefore, this land is suitable for both crop and grass growth.

In a global comparison, all three Uruguayan beef systems are rather extensive; largely depending on grass as animal feed (Seré and Steinfeld, 1996). In this study, the animal diet constitutes of grass or a combination of grass, grain of sorghum (Sorghum bicolor (L.) Moench), rice bran, rice husk and rice hay (Table 1). Uruguayan grasslands were assumed to be rainfed, since there was no available data indicating the existence of irrigated pasture. The category "by-products", i.e. rice and sorghum straw, does not result in any corresponding water use since they are a rest product of, e.g. another food or feed production process. The CWU related to these by-products, therefore, is embedded in the CWU of the main product. In cases where by-products have a significant economic or functional value, the relative CWU can be calculated based on e.g. economic allocation or biophysical allocation using the HDP or the energy value of the different products.

Table 1: Dietary composition and characteristics for the backgrounding (B) and finishing (F) stages of Uruguayan beef production systems combined as NP-NP, SP-SP and SP-FL, where NP is natural pasture, SP is seeded pasture and FL is feedlot (Modernel et al., 2013; Picasso et al., 2014).

Beef cattle system	NP-NP		SP-SP		FL-FL	
Dietary composition (%)	B	F	B	F	B	F
Natural pasture	100.0	100.0	30.0		30.0	
Seeded pasture			61.0	93.0	70.0	
Sorghum grain			9.0	6.5		60.5
Rice bran				0.5		12.0
Residues, vitamins and minerals						27.5
System characteristics						
Dry matter intake (kg animal^{-1} day^{-1}	9.9	12.0	7.9	8.4	7.9	13.2
Days to achieve final weight[a]	486	366	285	214	285	102

[a] 350 kg in B and 500 kg in F

Water use assessment

The water requirement per feed ingredient was calculated as the total CWU for a specific feed type. The CWU per type of vegetation used as feed was computed by the Lund-Potsdam-Jena managed Land (LPJmL) model (Gerten et al., 2005; Bondeau et al., 2007; Haberl et al., 2007; Rost et al., 2008a; Fader et al., 2010), accounting for area specific crop water requirement for each feed type, both irrigated and rainfed, vegetation growth and yield per pixel at a resolution of 0.5°. The CWU for production of crops was based on CWU during the growing season. The CWU for production on natural and seeded pasture was assumed to be evenly distributed throughout the year.

The LPJmL model yields results on CWU per type of vegetation, in m^3 per ton of fresh matter for crops, or per ton of dry matter for grasses. The CWU per ton of feed crop was multiplied by the amount of feed crops used to produce 1 kg of HDP from beef for each beef production system. All crops were assumed to have a dry matter content of 85%. For results to be comparable, livestock CWU were also calculated as litres/kg of beef, as presented in the results section in Figure 3.

Crop suitability index and maximum HDP from food crops

To determine the amount of HDP from food crops, the spatially defined crop suitability index (CSI) for global agroecological zones (GAEZ) (FAO, 2016) was used to define the suitability of both land and associated green water resources for food crop production. The GAEZ database operates at a 0.5-degree resolution. In this study the CSI for cultivated land was determined for baseline climate conditions (1961–1990) and a “high input level” situation, referring to a market-oriented farming system with well managed agricultural production (FAO, 2016) to reflect Uruguayan crop production. The CSI is based on input data of climate (i.e. frequency of wet days, temperature and sunshine), crop water requirements, soil conditions (i.e. pH, soil water holding capacity and total exchangeable nutrients), applied soil management, slope, elevation, terrain, land cover, protected areas and administrative areas (FAO and IIASA, 2012). In this study, land with crop suitability of either “good”, “high” or “very high” (i.e. a CSI > 55) for cultivation of food crops was regarded as suitable.

For the case study of Uruguay, CSI was assessed for the four major food crops produced in the country: wheat (Triticum spp.), rice (Oryza sativa), barley (Hordeum vulgare L) and maize (Zea mays) (MGAP, 2012). Crop suitability was determined for the Rocha region as all feed was assumed to be produced within that region. The CSI shows that all four major food crops have a suitability of > 55. For the water and land resources, used for pastoral biomass growth, that are also suitable for crop production, we account for the maximum amount of HDP that could be produced. This is calculated by combining crop yields per hectare of suitable food crops, i.e. wheat, maize, barley and rice, with protein content and human digestibility (van Zanten et al., 2016). Ideally, data on crop suitability should be obtained at the lowest possible spatial resolution. However, this case study is based on three beef production system spatially defined to a region of production and all feed is assumed to be produced within the same region. For this case study, there is no data available on crop suitability below regional level, thus the relative suitability of the four identified suitable food crops is based on their relative cultivated area within Uruguay.

The crop- and grasslands used by the beef systems in this analysis were preliminary rainfed and only used small amounts of irrigation water for feed production. However, rice cultivation in Uruguay requires some irrigation water for production. Since the objective of the WUR is to calculate the maximum amount of HDP that can be produced using the same amount and type of water currently used for production of beef in Uruguay, the requirement of additional blue water resources to cultivate rice will impact the suitability for rice production in this particular case study. A larger amount of blue water is required to cultivate rice than what is required for beef production in any of the three production systems. We therefore did not assume that rice can be produced satisfactory, only using the CWU currently used in the analysed beef production systems. Thus, even though rice has a CSI > 55 based on the GAEZ database, rice was assumed to be unsuitable for production considering the water resource availability, and accordingly excluded from further analysis.

The CWU on cropland and grassland used for beef production was divided into blue and green water resources. All blue and green water resources consumed on croplands and seeded pastures (i.e. as compared to natural pastures) were assumed to be suitable for crop production. The amount of HDP that can be produced from food crops was determined by dividing the CWU suitable for crop production with spatially explicit crop water requirements for the suitable food crops, which were assessed with the LPJmL model. Crop yields were subsequently multiplied by protein content and digestibility to determine HDP yield. National production and yield data for food crops were used since regional data was not available. Production and yield data were derived from the Uruguayan ministry of livestock, agriculture and fisheries (MGAP, 2012). Protein content and digestibility for selected crops were obtained from literature (Young and Pellet, 1994).

4.3 Results

Figure 3 illustrates the CWU to produce 1 k of beef for each of the three Uruguayan beef production systems, categorized per type of water and land and expressed as litres of water per kilo of meat. All production systems depend almost entirely on green water, with blue water resources only representing about 1% of total CWU. The NP-NP system requires the largest amount of water; 28,000 l of green water per kilo of beef and no blue water. The SP-FL system requires 13,800 l of green water and 430 l of blue water per kilo of beef and the SP-SP system requires the least water; 13,500 l of green water and 20 l of blue water per kilo of beef.

Direct competition over water between food and feed crops is illustrated in Figure 3 by the categories green and blue water from cropland. The NP-NP system does not include feed from croplands so there is no direct competition over water in this system. In case of the SP-SP system, however, croplands constitute about 16% of the total CWU, in comparison to 53% for the SP-FL system. Direct competition with production of human food crops, therefore, is highest in the SP-FL system.

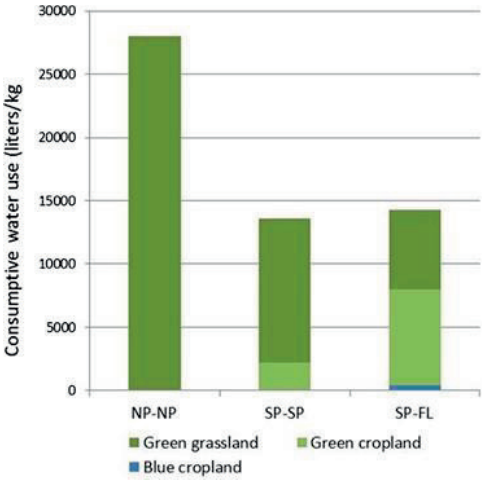

Figure 3. Consumptive water use (CWU) in litres per kg of beef for the three Uruguayan beef production systems in categories of green and blue water over crop and grasslands. Cattle in the systems are fed on different combinations of natural pastures (NP), seeded pasture (SP) and feedlot (FL). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

In Table 2 the CWU bars in Figure 3 are further disaggregated and present the green and blue resources behind each feed type used in the three beef production systems. Results in Table 2 indicate that the relative distribution of CWU between feed composition and production system vary greatly. For example, while the CWU for the NP-NP system entirely comes from green water on natural pastures, green water on seeded pastures corresponds to almost 70% of the total CWU in the SP-SP system. In the SP-FL system, seeded pastures only constitute 30% and green water on cropland for production of sorghum, instead constitutes a major part of the total CWU for the system.

The WUR was calculated to account also for indirect competition over water resources, i.e. competition where water is currently consumed over grasslands that could potentially support crop growth (Figure 4). Results show that the NP-NP system has a WUR of 0. This implies that the CWU to produce 1 k of beef yields no HDP from food crops, which is logical because the NP-NP system does not use any cropland or grassland suitable for crop cultivation. This livestock system, therefore, produces more HDP per litre of CWU than a crop system could have done. The SP-SP system had a WUR of 2.4, whereas the SP-FL systems had a WUR of 2.7, implying that the water required to produce 1 kg of HDP from beef could yield 2.4 kg of HDP from food crops in case of the SP-SP and 2.7 k of HDP in case of the SP-FL system.

Table 2: Consumptive water use for three Uruguayan beef production systems combined as NP-NP, SP-SP and SP-FL, where NP is natural pasture, SP is seeded pasture and FL is feedlot, categorized according to dietary composition of each system and summarized for green and blue water on crop and grassland.

	Beef cattle system	NP-NP	SP-SP	SP-FL	Water productivity[a]
CWU (l/kg beef)	Dietary composition	l/kg beef	l/kg beef	l/kg beef	l/kg crop/grass
Green water	Natural pasture	28 014	2 056	2 056	533
	Seeded pasture		9 269	4 180	533
	Rice		31	642	786
	Sorghum		2 182	6 947	1 195
Blue water	Rice		20	429	469
	Sorghum		<1	<1	<1
Green water on grassland		28 014	11 325	6 236	
Green water on cropland			2 213	7 589	
Blue water on cropland			20	429	
Total		28 014	13 558	14 254	

[a] Crop and grass water productivities has been inverted to litres of water per kg of output to enable easier comparison with CWU estimates.

This depends on that these two systems, as can be seen in Table 2, although still using mostly green water resources, to a large extent use green water evapotranspired over crop and grasslands that are suitable to support crop growth, and thus can be used for HDP production directly. Based on the traditional measure for water productivity, i.e. litres of CWU per kilo beef produced, we would conclude that the NP-NP systems is less efficient than the other systems, whereas based on our new WUR, the opposite conclusion can be drawn (Figure 4).

Our WUR results show a comparable pattern to results based on the land use ratio (LUR; Figure 4) (van Zanten et al., 2016). Both the WUR and LUR results indicate that it is more efficient to produce HDP from food crops than from livestock for the SP-SP and SP-FL systems, and that livestock production is the most efficient way to produce HDP in the NP-NP system. However, for the two more intensive production systems, the LUR results (5.7 for SP-SP and 6.2 for SP-FL) are significantly higher than the WUR results (2.4 for SP-SP and 2.7 for SP-FL) (Figure 4). A higher LUR in relation to WUR indicates that the system can yield a higher amount of HDP based on its land use relative to the amount of food crops it can yield based on its water use. WUR and LUR results also indicate that the SP-SP system use both water and land resource more efficiently than the SP-FL system, when considering feed-food competition.

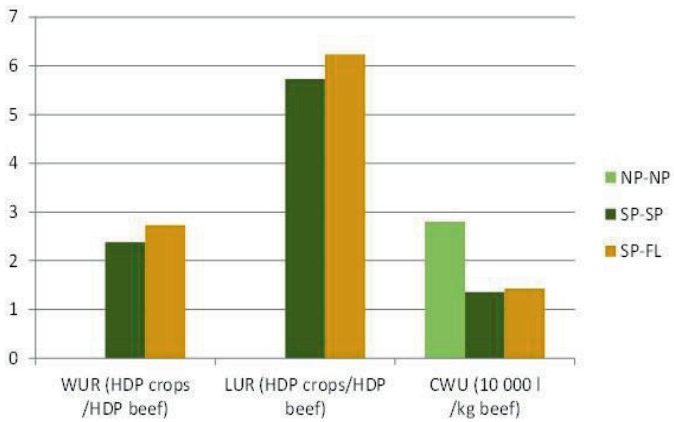

Figure 4: Water use ratio (WUR), as kg human digestible protein (HDP) from food crops/kg HDP in ASF, compared with land use (LUR) in HDP from food crops/HDP in ASF and consumptive water use (CWU) in 10 000 litres per kilo of beef calculated for three Uruguayan beef production systems. The NP-NP system does not appear in the WUR and LUR results, because they are equal to 0.

Since natural grasslands are assumed not suitable for crop production, the NP-NP system generates a WUR and LUR of 0. The results from this study indicate that the NP-NP beef production system could be important from a food security perspective, since it does not compete, directly or indirectly, with human food production. This finding is not identified using traditional CWU assessment methods where, as in this case study, the NP-NP system seems to be the least efficient beef production system in terms of water use.

It should be noted that this is because the NP-NP system relies entirely on natural grassland with a crop suitability index well below the minimum level of "good" that was used to determine if land was suitable or unsuitable for crop production in this study. In reality, all land has some suitability for producing crops and would thus have a LUR/WUR above zero. Extensive systems that use concentrates and cultivated roughage, although to a very small extent, would also generate a ratio above zero.

4.4 General discussion

This study aimed to investigate water resource use in livestock production and the competition over water resources for production of food crops. In the past, water use assessments primarily focused on withdrawals from water bodies and groundwater, for agriculture, industry, municipal or domestic uses (Shiklomanov, 2000). These assessments did not account for the large amounts of green water, i.e. naturally infiltrated rainfall in the soil. For livestock production, green water resources constitute 90% of the total CWU on a global average, looking at grazing, mixed and industrial livestock production systems (Mekonnen and Hoekstra, 2012). Today, the concepts of green and blue water are widely used to describe and assess water use in agriculture, including livestock production (e.g. Mekonnen and Hoekstra 2012; Ran et al. 2013; 2016).

4.4.1 Capturing the complexity of green water use

Most of the variation in results from water use assessment studies of livestock relates to whether or not green water is included, partially included or excluded from assessments (Ran et al., 2016). Some studies argue that all, or most of the green water use should be excluded (e.g. Ridoutt et al. 2012a; Ridoutt and Pfister 2013), because a large amount of the consumptive green water is used for production of human non-edible biomass, such as grass, produced on non-cultivated fodder land and grasslands. The water evapotranspiring over such land areas would be consumed for biomass growth regardless if the biomass was used as animal feed, or not (Deutsch et al., 2010; Ran et al., 2013). Others highlight the importance of looking at both green and blue water resources to identify areas of improvement (e.g. Hoekstra and Mekonnen, 2012; Mekonnen and Hoekstra, 2012). They argue that the location of where the blue and green water is consumed does not matter significantly, since the focus for policy-relevant water use studies should be on decreasing the total water use for food production globally (Hoekstra, 2014).

Recent studies also indicate that we should account for the local and environmental impacts associated with water use (De Boer et al., 2013; Schyns et al., 2015), for example by using water stress-related indexes (e.g. Ridoutt et al. 2012a; Ridoutt and Pfister, 2013; Zonderland-Thomassen et al., 2014) and water scarcity assessments (Mekonnen and Hoekstra, 2016). These measures, however, only focus on the scarcity of blue water resources, with the aim to assess environmental impacts (i.e. direct impacts on river flow or aquifer levels), and not green and blue resource use efficiency caused by abstractions of blue water.

This perspective does not include the role of surplus soil moisture contributing to blue water recharge, and the opportunity costs of both blue and green water usage (Deutsch et al., 2010). Recent reviews of freshwater use in agriculture argue that there is a need to further develop methods that deal with the efficiency of green water use, as well as the scarcity of green water resources and indicators to measure that scarcity (Kounina et al., 2013; Schyns et al., 2015).

In this study we, therefore, seek to capture the green water efficiency and complexity by developing and applying the WUR. This is a measure that addresses different debated aspects, in particular for green water resources, related to water productivity and green water use, such as water use efficiency of feed production, the ability to convert human non-edible feed products into food, and the opportunity cost of resource use for agricultural production (van Zanten, 2016).

The CWU estimations for livestock production in this study are well within the range of previous estimates, ranging from 13,000 l to 30,000 l of water per kg of beef (e.g. Molden et al. 2007a; 2007b; Mekonnen and Hoekstra 2012; Ran et al. 2013). The hydrological model and crop suitability data set both operates at a 0.5 spatial degree resolution. Ideally, for a regional analysis, water modelling as well as crop suitability data should be collected for a lower spatial resolution to deliver more precise national/sub-national results on crop and grass water requirements and spatial variability within the region.

Another future improvement would be to include herd dynamics of the analysed livestock production systems. Due to insufficient data sources, the cow-calf system was not considered in this study. Since the cow-calf phase is similar for all three systems and constitute only a small fraction of total CWU for feed, an inclusion would not largely impact the comparison of CWU for different systems. However, it may impact the WUR of each system and should therefore be included in further analyses.

4.4.2 Considering multiple resources and competitive uses

The results shown by the WUR method are similar to the results generated when using the LUR approach to assess the feed-food competition related to the use of land resources. A comparison of the WUR and LUR results in the studied Uruguayan beef production systems show that the potential contribution to HDP by producing food crops is higher when based on land resources, than when based on water resources. This indicates that land, rather than water, is the limiting resource in the compared systems under prevailing conditions. These differences highlight that the feed-food ratio depends on the natural resource under study. However, it is the natural resource that limits production under prevailing conditions that will determine the actual feed-food competition of that particular system and time. This will differ from system to system, and will change dependent on management practices and resource availability at the point of analysis. Therefore, resource use assessments should preferably consider multiple resources, since efficient use of one resource is not necessarily efficient use of another resource.

The opportunity cost of resource use can also change with altered parameters in a production system. Access to irrigation water and nutrients can transform currently unproductive land to suitable farming land for food crops. Such change could also be captured in WUR/LUR calculations to enable comparison between different points in time and identifying opportunities to increase the number of human beings that can be nourished per unit of input, e.g. water resources (van Zanten et al., 2016). This cannot be achieved only by increasing production efficiency. Optimizing resource allocation by identifying alternative uses, multiple users and multiple benefits can be crucial.

The methodology presented in this paper can help identify opportunities to feed the world sustainably in several ways. Primarily the method can be used to identify livestock production systems that use natural resources with low opportunity costs for other uses. This will make it possible to identify and value production systems that use crop residues, food waste or grass produced on marginal lands, in comparison to systems that increase their efficiency by using nutritious feed crops that can be directly consumed by humans.

The methodology can also help to identify unsustainable uses of e.g. blue water resources. Through alterations in feed composition, such use can be shifted to a more sustainable use of green water resources. Lastly, the method can highlight potential situations where livestock production systems could benefit from an additional use of blue water resources to increase the water use efficiency in the system.

Studies of environmental impact of beef production do not successfully capture all ecosystem benefits (Eshel et al., 2014). Thus, the method presented in this study should be developed to also capture other competitive uses, such as fuel and fibre production, and potentially competition with ecosystem functions, e.g. by including competition with regulating and cultural ecosystem services, moving the concept of sustainable intensification to also include eco-efficiency, that is to produce more value with less impact (Tittonell, 2014). For example, overgrazing by cattle may impose a threat to the ecosystem in terms of land degradation, which may also cause large water losses in the long term (Saxton and Rawls, 2006; Bossio et al., 2007). Such an extended method could also be used, for example, to assess potential benefits of integrated crop-livestock systems in comparison with intensive agriculture (Lemaire et al., 2014).

4.4.3 Production of food crops or livestock feed

The methodology presented in this paper adds two new features to the concept of quantifying CWU of livestock products: 1) a categorisation of water resources in classes, defined by land use, which enable identification of how much of the total CWU could have been used for human food production directly, 2) identification of indirect competition over resources, by calculation of a WUR, based on the potential of the water and land resources that are currently used by the livestock system, to be used for another, more beneficiary way to produce HDP. The developed method enables identification of livestock systems that use large amounts of green water with a low opportunity cost for the production of food crops, and thus appear to be efficient in comparison to systems that use water resources with higher opportunity costs. As livestock is increasingly fed on human edible products such as grains, and productive cropland is dedicated to animal feed production, it is important to demonstrate the efficiency of water use for livestock in terms of food supply, e.g. by showing how much of the total CWU could have been used to produce human food crops more productively. This is not properly captured if water resources are just quantified, even if they are categorized into blue and green water.

A study by Cassidy et al. (2013) indicate that the global calorie availability could be increased by up to 70% if crops are directly consumed by humans rather than used as animal feed and biofuel production. This study also suggests that shifting meat consumption away from beef towards more poultry and pig meat could potentially nourish more people per ha. The results presented in our study, however, indicate that ruminants can play an important role in future food security as they convert human non-edible biomass to nutritious food. In terms of maximizing HDP per unit of water or land, such production systems can be regarded as resource use efficient.

The WUR results presented in this paper, however, also show that, in regard to water resources, it would be more efficient to produce HDP from food crops rather than livestock for the two more intensive beef cattle systems, SP-SP and SP-FL. The results are mostly dependent on the large amount of sorghum that is fed to cattle in the feedlot systems and the use of water for grass production on seeded pasture lands that are also suitable for crop production. Only the livestock system relying entirely on natural pasture, produce HDP more efficiently than food crops could. Results indicates that alterations in feed composition may change the resource competition significantly, which can be of great local importance and contribute to more sustainable resource use in agriculture.

4.5 Conclusion

This study argues that water resource use for livestock should be analysed and considered based on three criteria. First, water resources use should be distinguished as blue and green water resource use. Second, they should be categorized according to the land over which they are evapotranspired. Third, the competition over resources should be included in assessments to bring significance to the large use of green water in livestock systems.

To tackle these issues, we developed a water use ratio that enables comparison of livestock production and plant production systems for best use of water to produce human edible proteins. Results from this study show that based on the traditional measure for water productivity, i.e. litres of CWU per kg beef produced, we would conclude that the most extensive Uruguayan beef production systems use water resources less efficient than the more intensive systems, whereas based on our new water use ratio, the opposite conclusion would be drawn.

This study shows that livestock, and livestock production systems that produce HDP from human non-edible biomass appropriating CWU from land with none, or very low suitability for crop cultivation, can play an important role in food security. It also indicates that some livestock production systems use resources that may be more suitable for competing purposes, and that multiple resources should be considered, in order to contribute to the identification of trade-offs and opportunities for improvement and sustainable intensification.

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4

Chapter 5

Towards more spatially explicit assessments of virtual water flows: linking local water use and scarcity to global demand of Brazilian farming commodities

Rafaela Flach[1,2,5], Ylva Ran[3,4], Javier Godar[3], Louise Karlberg[3] and Clement Suavet[3]

[1] Department of Thematic Studies, Linköping University, 581 83 Linköping, Sweden
[2] International Max Planck Research School on Earth System Modelling (IMPRS-ESM), 20144 Hamburg, Germany
[3] Stockholm Environment Institute, 104 51 Stockholm, Sweden
[4] Animal Production Systems Group, Wageningen University PO Box 338, 6700 AH Wageningen, The Netherlands
[5] Author to whom correspondence should be addressed

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Abstract

Global consumption of farming commodities is an important driver of water demand in regions of production. This is the case in Brazil, which has emerged as one of the main producers of globally traded farming commodities. Traditional methods to assess environmental implications of this demand rely on international trade material flows at country resolution; we argue for the need of finer scales that capture spatial heterogeneity in environmental variables in the regions of production, and that account for differential sourcing within the borders of a country of production. To illustrate this, we obtain virtual water flows from Brazilian municipalities to countries of consumption, by allocating high-resolution water footprints of sugarcane and soy production to spatially explicit material trade flows. We found that this approach results in differences of virtual water use estimations of over 20% when compared to approaches that disregard spatial heterogeneity in sourcing patterns, for three of the main consumers of the analysed crops. This discrepancy against methods using national resolution in trade flows is determined by national heterogeneity in water resources, and differential sourcing. To illustrate the practical implications of this approach, we relate virtual water flows to water stress, identifying where global demand for water coincides with high levels of water stress. For instance, the virtual water flows for Brazilian sugarcane sourced by China were disproportionally less associated to areas with higher water stress when compared to those of the EU, due to EU's much higher reliance on sugarcane from water scarce areas in Northeast Brazil. Our findings indicate that the policy relevance of current assessments of virtual water flows that rely on trade data aggregated at the national level may be hampered, as they do not capture the spatial heterogeneity in water resources, water use and water management options.

5.1 Introduction

Freshwater resources are becoming scarcer globally (Falkenmark, 2013). Conflicts around freshwater use are rising and already create strong tensions between countries, regions and sectors (industry, agriculture, urban demand and conservation). Agriculture production today requires about 70% of global freshwater resources, compared with only 10% for households and the industrial sector respectively (Molden et al., 2007a). As such, global trade of agricultural commodities is one of the main drivers of impacts on water availability and land use change (Hoff, 2009; Rockström et al., 2014). A growing population and changing diets associated to rising incomes and urbanisation are set to increase pressure on water resources even further (WWAP, 2012).

Although basin-scale analyses and governance still shape most of the water research and development agenda, there is a need for better understanding of scale interdependencies, linkages and teleconnections in the global water system (Vörösmarty et al., 2013; Rockström et al., 2014). Moreover, there is an expressed demand for indicators of water use in supply chains that are policy relevant and contribute to ensure sustainable resource use, linking consumers to producers (Hoekstra et al., 2012; Hoekstra and Wiedmann, 2014).

The concept of water footprints, i.e. the amount of water consumed per unit of produced item (Hoekstra et al., 2011), aims primarily at measuring the human appropriation of global water resources (Ercin and Hoekstra, 2014). It also strives to increase awareness about global water resource use for consumption by under-pinning assessments of virtual water flow, i.e. the volume of virtual water that is being transferred from one area to another as a result of trade of goods and services (Hoekstra et al., 2011). A large number of studies link water footprint accounts to trade aiming to assess international dependency on external resources (e.g. Hoekstra and Hung, 2005; Chapagain and Hoekstra, 2008; Hanasaki et al., 2010; Ercin et al., 2013), opportunities of sparing resources in a location through trading of goods and services from elsewhere (e.g. Chapagain et al., 2006; Fader et al., 2011; Konar et al., 2013; Biewald et al., 2014), or to assess pressures to local water resources (e.g. Dong et al., 2014).

There is a mismatch, however, between the national scale at which trade analyses are traditionally assessed, and the sub-national scales at which consumptive water use, impacts on water resources and water governance occur (Ridoutt and Pfister, 2010; Biewald et al., 2014). Both the accuracy and spatial resolution of water footprint and water use accounts have steadily improved over time (e.g. Liu et al., 2007; Rost et al., 2008b; Siebert and Döll, 2008; Mekonnen and Hoekstra, 2011a). However, this high resolution is lost in virtual water flow assessments which aggregate the original detail of the water footprint accounts with trade data at the national scale, both for when trade is estimated by physical accounting of traded material flows (e.g. Hanasaki et al., 2010; Fader et al., 2011; Ercin et al., 2013; Zhang et al., 2016) as well as by input–output analyses (e.g. Lenzen et al., 2013; Kastner et al., 2014; Lutter et al., 2016).

The aggregation of trade data to the national scale is a result of basing calculations on nationally reported global trade data with national resolution (e.g. COMTRADE or FAOSTAT) or other datasets instead of subnational trade data (Godar et al., 2015; Godar et al., 2016; Jiang et al., 2015). Therefore, all consumer countries account for the same amount of virtual water from the producer country per consumed unit, regardless of if they are sourcing from different regions within the production country and/or rely on production systems with different water resource endowments and water use management. Aggregation at the national scale result in that key sub-national parameters, such as local water scarcity issues, or precipitation differences between regions within a country, are not captured in virtual water trade accounts. Consequently, identifying key actors along a supply chain that may have a large impact on water consumption in the specific region of production, and thus may be important stakeholders to consider in water management decisions, is currently difficult. This study is part of recent seek to consider sub-national scales in virtual water trade assessments (Biewald et al., 2014; Dong et al., 2014; Jiang et al., 2015).

In addition, concerns about the capacity of consumptive water accounts (e.g. virtual water estimates and water footprints) to provide policy relevant information on local pressures, or to help establish a direct causality between demand drivers and pressures on the ground, have been raised previously (e.g. Ridoutt and Huang, 2012; Perry, 2014; Wichelns, 2015). Water footprint analyses do not provide information about the impact of the consumed water for ecosystem functions or other competing water users, or alternative uses (Ridoutt and Huang, 2012; Wichelns, 2015; Ran et al., 2016). The focus on 'total water removal' in a country scale hampers an informed decision on sustainable sourcing for cost-efficient production and consumption and ignores the complexity of water resource use and allocation. To address such criticism several studies attempts to estimate the sustainability of water footprints at local (Gleeson et al., 2012; Wada and Bierkens, 2014) and global scales (Hoekstra and Wiedmann, 2014), and comparing global water use with the planetary boundaries for freshwater use (Steffen et al., 2007; Gerten et al., 2013). Along with a meaningful progress in conceptualizing and estimating water scarcity as a local and global issue (Falkenmark, 1989; Smakhtin et al., 2004; Pfister et al., 2009; Hoekstra et al., 2012), water scarcity assessments have recently received increased attention in several water footprint studies (Hoekstra et al., 2012; Biewald et al., 2014; Mekonnen and Hoekstra, 2016).

This study aims to bridge the existing knowledge gap in virtual water assessments related to trade by improving spatial explicitness in trade flows and relates this to virtual water accounts and local water scarcity. We use the global supply chains of Brazilian soy and sugarcane to conceptualize the developed method and illustrate how improved spatial explicitness and accounting for local conditions of water scarcity enables an identification of major water users along the supply chain (in this case exemplified by the EU and China) in critical areas of water scarcity. Based on these findings, we elaborate on a new approach to assess pressures of water use related to traded commodities, allowing for more policy relevant and actionable information on the ground to support improved sustainability measures along water-demanding international supply chains.

5.2 Method

The method developed in this paper is based on linking detailed assessments of traded material flows to water footprint and water scarcity estimates for two main Brazilian crop commodities, sugarcane and soy. The method consists of a step-wise process; first, the spatial explicit water footprints of sugarcane and soy are estimated. Second, the production of sugar and soy, and their associated water use are linked to trade flows at a high spatial resolution. Finally, the tradeflow related water footprints are coupled to data on local water scarcity at the municipality level.

5.2.1 Water footprint accounting

This study assesses the consumptive water use of internationally traded products, thus, an abstracted water volume with no return flow to the same basin. We focused on surface and groundwater withdrawal, i.e. blue water, rather than rainwater or soil moisture, i.e. green water (Rockström et al., 2009a), since the use of blue water resources can be directly related to water scarcity. The results from the global model by Mekonnen and Hoekstra (2011a) were used. The water footprint model quantifies the water footprint of global crop production for the period 1996–2005, estimating the water footprints of 126 crops. It takes into account the daily soil water balance and climatic conditions for each grid cell. The data was first regionalized to the municipality level and then extrapolated to each year within the period 2001–2011, accounting for changes in the distribution of crop production, harvested area and yields at the municipal scale (IBGE, 2015), as described in Appendix A, Table A1.

5.2.2 Trade flow modelling

The SEI-PCS model6 (Godar et al., 2015; 2016) allows for tracing global consumption of farming products to the sub-national regions of production (e.g. municipalities in Brazil), thereby enabling an assessment of associated pressures of international consumption on sourcing regions. The tool uses a combination of sub-national production, domestic allocation, custom declarations and international trade data to estimate the physical amounts of goods exported from each production area to all countries of consumption (further described in Appendix A). Using the traded products defined by the Harmonized Commodity Description and Coding System from the World Customs Organization, including soybeans, soy cake, soy oil and soy sauce for the soybean crop, and sugar and ethanol for the sugarcane crop (see Appendix A), this tool was applied for all identified consumer countries of Brazilian production. For the sake of clarity, soybean and sugarcane equivalents are used throughout this paper (Godar et al., 2015).

5.2.3 Water stress

In order to assess the implications of global consumption of traded commodities on local water stress in regions of production, a use-to-availability indicator was calculated. The indicator was estimated by dividing the total water demand at the micro-basin level (166 843 sampled micro-basins covering the vast majority of the Brazilian territory) by the available water flow in the same area, as estimated by the Brazilian Water Agency (ANA, 2013). The thresholds for each class of water stress, i.e. high, intermediate and low, were based on the classes of Raskin et al (1996) and are described in Appendix A. The water availability is defined as the Q95%, i.e. the flow in cubic metres per second which was equalled or exceeded for 95% of the flow record, summed to the regularised flow in case of existence of upstream dams, and the total water demand comprises industrial, domestic, agriculture and rural demands (ANA, 2013).

5.3 Results

5.3.1 Virtual water use of Brazilian soy and sugar cane

The analysis of sub-national differences in virtual water flow for various consumer countries reveals marked differences. We focus on the main consumers, China and the EU, for the sake of clarity. Figure 1 illustrates the virtual water flow of soy and sugarcane, distributed by Brazilian municipalities related to consumption in China and the EU in year 2011. For soybeans, the total virtual water flow amounted to 67 Mm^3 of blue water, predominantly originating from Southern Brazil7. This water was consumed in order to produce 75 Mton of soy, with an average associated water footprint of 0.89 m^3 ton−1, ranging from an average of 0.22 m^3 ton−1 in Northern Brazil to 16 m^3 ton−1 in the South.

Virtual water flow for sugarcane production was substantially higher than for soybeans and amounted to approximately 3350 Mm^3 of blue water. About 75% of the virtual water flow for sugarcane consumption occurred in the Central-West region, but some also originated from the coastal regions in the East and North- east regions. In total, 734 Mton of sugarcane were produced, with an average associated water footprint of 4.5 m^3 ton−1, ranging from an average of 0.25 m^3 ton−1 in the South to 27 m^3 ton−1 in the Northeast region.

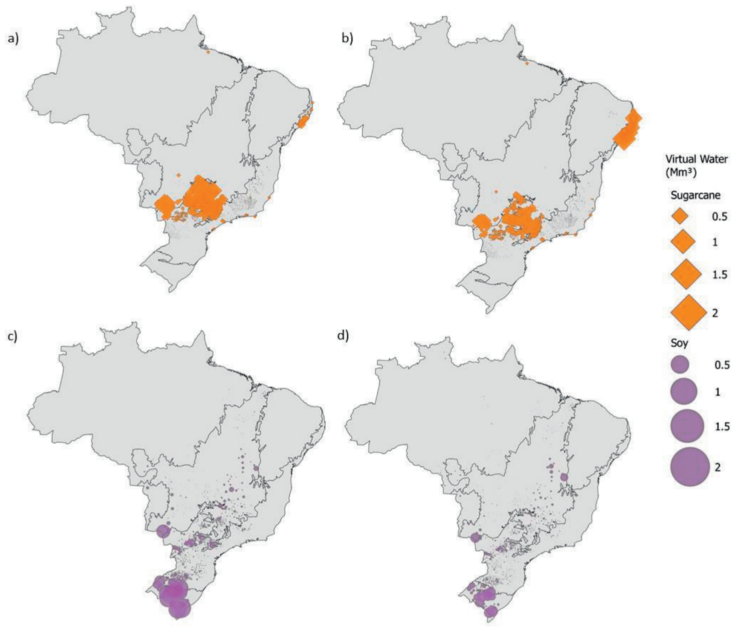

Figure 1: Blue virtual water flow in 2011 at the municipal level, for (a) Brazilian sugarcane consumed in China, (b) Brazilian sugarcane consumed in the EU, (c) Brazilian soy consumed in China, and (d) Brazilian soy consumed in the EU, in Mm^3 of water.

Figure 1 also illustrates the large spatial variation of Brazilian sourcing between the consumer countries. The aggregated virtual water flow for Chinese consumption of soybeans was almost three times larger than that of the EU in 2011 (34 Mm^3 and 12 Mm^3, respectively). This is partly explained by the fact that China consumed almost twice as much Brazilian soybeans as the EU (24 Mton compared to 13 Mton). However, the virtual water flow related to Chinese consumption of soy was also strongly linked to its relative preferential sourcing of soybeans from municipalities in the South region (ANA 2013), resulting in comparatively high associated water footprints (1.4 m^3 ton−1 on average). In comparison, the EU consumed more soy from municipalities in other areas with relatively small associated water footprints (0.9 m^3 ton−1 on average).

Regarding sugarcane, the virtual water flow of China was higher than that of the EU (60 Mm^3 and 47 Mm^3, respectively) (figure 1). Although China consumed considerably more sugarcane than the EU in 2011 (16 Mton and 11 Mton, for China and the EU respectively), the virtual water flow of China is proportionally lower than for the EU. This is explained by the fact that China is primarily sourcing from municipalities in the Southeast region with comparatively low associated water footprints (3.7 m^3 ton−1 on average), while the EU consumes comparatively much more sugarcane from municipalities situated in the dry areas of the Northeast region (4.1 m^3 ton−1 on average).

5.3.2 Discrepancies of spatially-explicit versus nationally aggregated virtual water accounts

Accounting for sub-national high-resolution sourcing of crops for different consumer countries enables considering differences in water footprints between regions where consumer countries source traded goods. Figures 2(a) and (b) shows the observed discrepancies between municipal-scale and nationally aggregated virtual water accounts, which range between overestimations of up to 188 Mm^3 or 7.4% (Brazil) and underestimations of 13 Mm^3 or 38% (United Arab Emirates) of virtual water use for sugarcane between different consumer countries. Underestimations are especially relevant for the two major consumer regions, China and the EU, with 21% and 10% respectively. Overall, a large part of the underestimations for global consumers was masked by an overestimation for the main overall consumer of Brazilian soy and sugar cane, which is Brazil itself[3].

[3] Here we included exclusively the soy consumed or traded as soybeans or one of its primary processed products (soy oil, soy meal and soy sauce), as well as the sugar cane consumed or traded as sugar or ethanol. The inclusion of embedded soy and sugar cane in third products that are heavily exported (Kastner et al 2014) would certainly decrease total Brazilian consumption and therefore its virtual water flow, because Brazil is a major exporter of products such as poultry, processed food and a large diversity of other commodities in which soy and sugar cane are embedded (Godar et al 2015).

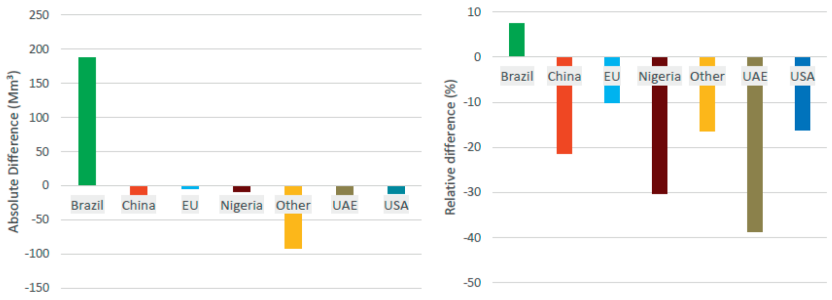

Figure 2: Difference between traditional virtual water assessments (country resolution) and spatially explicit (municipal resolution) virtual water assessments per country for sugarcane, year 2011, in (a) total amount of water (Mm^3), and (b) relative difference between the two approaches (%). Positive values indicate an overestimation with respect to traditional country-to-country approaches.

5.3.3 Global sourcing from high water scarcity regions

In Brazil, water scarcity levels vary greatly in space, as illustrated in figure 3. In particular, there are three critical areas: (1) the Southern region, with high irrigation demand, such as water-intensive rice crops (ANA, 2013), (2) large metropolitan areas like Sao Paulo with high pressure on water resources due to high demographic, urban and industrial use, and (3) the Northeast region, which presents climate-related water scarcity resulting from a semi-arid climate and occurrence of drought periods.

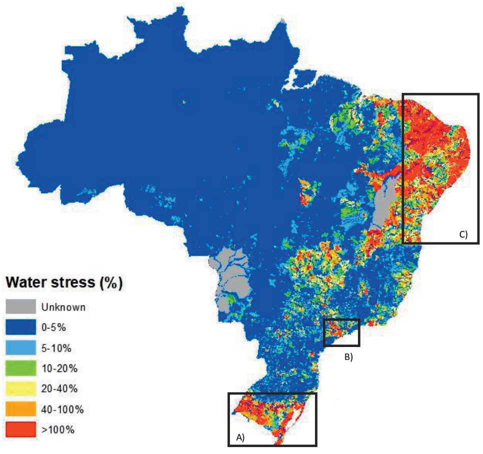

Figure 3: Water stress (%) per micro-basin. The rectangles highlight three critical regions for water stress: (A) the intensive rice irrigated areas in the South, (B) the highly populated metropolis of Sao Paolo in the Southeast, and (C) the semi-arid and water scarce Northeast. Based on data from 166843 micro-basins (ANA, 2013).

By spatially linking water scarcity and virtual water flows, we observe that the risk for undesirable impacts on water resources caused by global consumption of Brazilian soy and sugarcane varies considerably between crops. For instance, 17% of the total virtual water flow related to Brazilian sugarcane consumption occurs in regions with medium and high water stress, while this figure drops to just 8% for soybean consumption. Thus, the aggregated virtual water flow for sugarcane is not only larger in quantity, but also is associated to higher pressures on water stress areas, in comparison to soy.

Similarly, because different countries source their crops from different regions, their virtual water flow also may have different local impacts on water resources. For instance, as illustrated in figures 4(a) and (b), 36% of the virtual water flow for the EU consumption of sugarcane originates from municipalities with high levels of high stress, predominantly in the coastal regions of the Northeast, while the corresponding share for China is only 4%, mostly related to sourcing from municipalities in the Southeast region. On the other hand, the pressures of their soy consumption on water-stressed areas appears to be rather similar for both regions; 7.8% and 8.3% of the virtual water flow was sourced from municipalities with intermediate water stress levels, for EU and China respectively (figures 4(c) and (d).

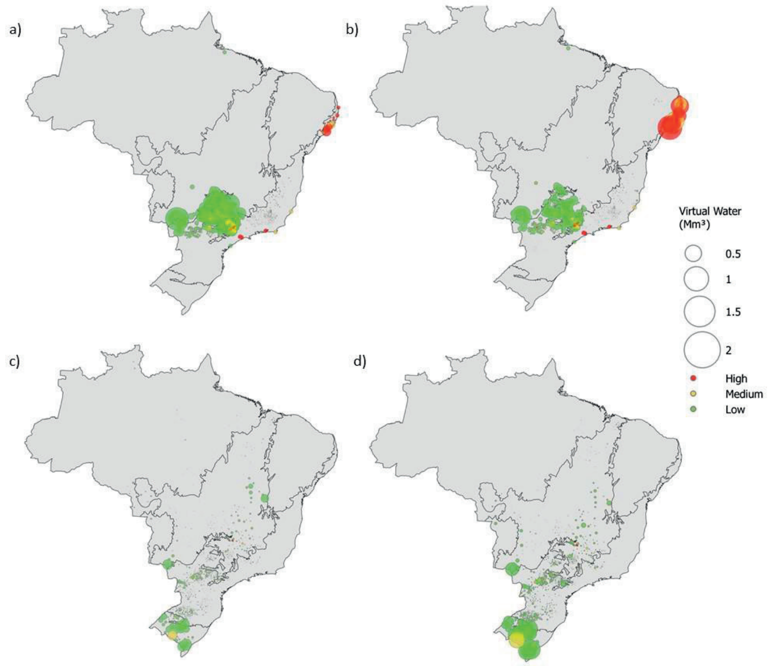

Figure 4: Virtual water use at the municipal level in 2011 in low (green), intermediate (yellow) and high (red) water stressed areas for (a) Brazilian sugarcane consumed in China, (b) Brazilian sugarcane consumed in the EU, (c) Brazilian soy consumed in China, and (d) Brazilian soy consumed in the EU.

5.3.4 Understanding global virtual water use dynamics and their impact in production regions

It is possible to link varying dynamics of consumption and trade of specific countries, to the dynamics of virtual water flow mediated by global trade in production regions of interest. For instance, the virtual water flow of Brazilian soybeans consumed in China has increased considerably (1100%) since 2008, mainly due to increased consumption (figure 5). Conversely, it is possible to analyse the opposite, i.e. how distortions in local conditions affect virtual water flows. This was the case during the infamous drought in 2005 that had an effect on crop yields in some regions of the country (USDA, 2006). While the overall virtual water flow of soybeans clearly increased with time (+197% from 2001 to 2011, as opposed to a 70% increase in production), the drought resulted in an increase in the virtual water flow by 89% only in 2005 when compared to the average of the studied period. However, a closer look at the data reveals that while most countries increased their virtual water flow in 2005, the water flow for domestic soy consumption in Brazil actually decreased, which was related to a significant decrease in the consumption of domestically produced soy (Godar et al., 2015), probably caused by drought driven poor yields. Consequently, the virtual water flow of countries traditionally sourcing from drought-affected areas increased considerably.

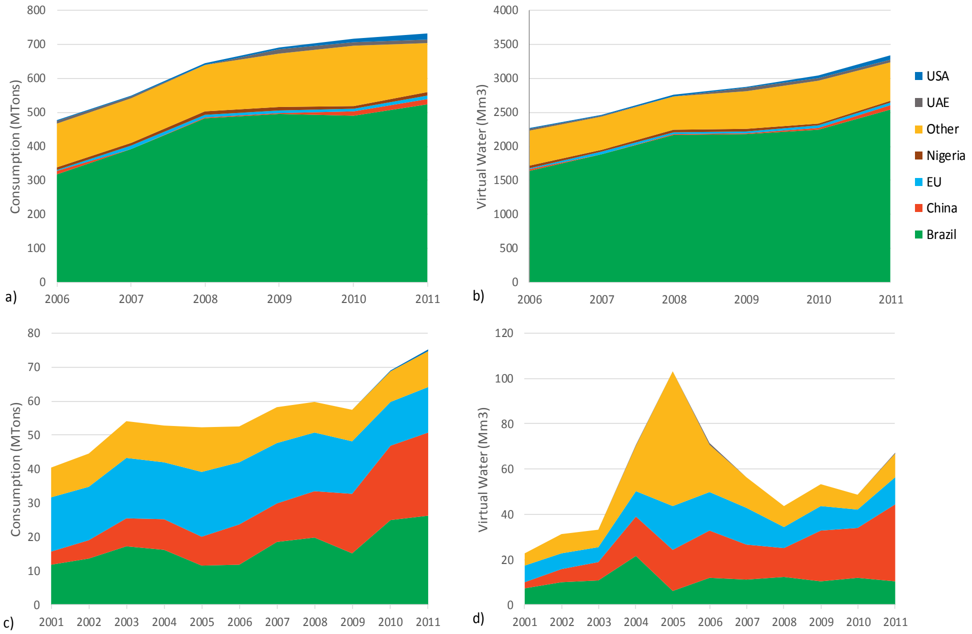

Figure 5: Global consumption of Brazilian soy and sugarcane, and associated virtual water trade, in the period 2001–2011: (a) soy consumption (b) sugarcane consumption, (c) annual virtual water trade per consumer country for Brazilian soy (Mm^3), and (d) annual virtual water trade per consumer country for Brazilian sugarcane (Mm^3).

5.4 Discussion

Due to its availability of arable land and water resources, Brazil is becoming an increasingly important player supporting food security for a growing world population (Lathuillière et al., 2014, Flachsbart et al., 2015). At the same time, this role brings about trade-related concerns such as trade-offs of resource use between various actors (including food security among smallholder producers), or the local impacts and risks that need to be considered by policy-makers and all stakeholders involved in global supply chains. A spatially explicit and high-resolution linkage between all actors in a supply chain and the regions of production from which they depend is a key entry point to address these issues (Godar et al., 2016). This is particularly important for water resource use given its criticality and local relevance (as opposed to for example GHG emissions whose impacts are shared globally) (Wichelns, 2015).

The water footprint estimates used in this study for estimating virtual water footprints (Mekonnen and Hoekstra, 2011a), fall well within the range of other global water footprint accounts (e.g. Hanasaki et al., 2010; Liu and Yang, 2010), although other studies that used a bottom-up approach to water footprint accounting in Brazil found diverging values (da Silva et al., 2015; Lathuillière et al., 2014). However, the aim of this study is not to present absolute numbers of virtual water use of crops, but to illustrate the importance of considering trade flows at a sub-national spatial scale to obtain accurate virtual water footprints and water scarcity linkages.

Our results highlight two key aspects to be considered in virtual water flow assessments of farming commodities. First, increased transparency in product value chains enables an identification of actors directly linked to virtual water use at the local level, by connecting them to sourcing regions and impacts at sub-national scales. We found that virtual water footprints for Brazilian soy and sugarcane were clearly distinct, and varied significantly between regions, countries of consumption and over time. For instance, the aggregated virtual water flow for sugarcane sourced by China was found to be disproportionally low when compared to that of EU consumption. This is explained by the fact that China imported sugarcane from municipalities with lower crop water footprints per consumed unit compared to the EU. Unless virtual water assessments are linked with trade analysis at relevant sub-national scales, it is not possible to identify key actors along the supply chain with the highest associated water use. While for the sake of clarity we have focused only on linking country consumers with regions of production, it is possible to identify the companies (exporters, importers) that are linked to those regions of production as well (see https://ttp. sei-international.org/ and Godar et al 2016).

Second, potential impacts of virtual water flows vary between regions of production. There is a growing concern that global consumption may exacerbate water stress in the regions of production of farming commodities. For instance, our results indicated that more than a third of the virtual water flows associated with sugarcane consumption in the EU originate from highly water-stressed areas predominantly in the coastal regions in the northeast of the country, in comparison to just 4% for Chinese consumption. Moreover, our results indicate that the aggregated virtual water flow for sugarcane is not only larger in quantity, but also has a higher pressure related to water stress, in comparison to soy. The different sourcing regions for both consumer regions (EU and China) vary in hydroclimate and water demands, therefore giving rise to different allocation of water resources and associated socio-economic impacts. Linking virtual water use to water scarcity data and other information relevant to social and environmental issues is thus essential for the detection of critical hotspots to focus interventions, zoning and other types of spatial planning and water management.

To enable relevant attribution of virtual water flows to imports and exports and inform traders and retailers of the water demand of a product at the site of production, spatial-explicitness is imperative. Our approach enables an identification of actors along the supply chain sourcing farming commodities with high virtual water content from critical hotspots of water stress that may be exacerbated by global consumption, revealing potential needs to prioritize between alternative water uses. Underlying our specific findings, this paper thus argues for the use of spatial-explicit trade information that links subnational regions of production at a scale that is relevant to understand national heterogeneity in water resources and water management. Compared with an approach that does not account for differential sourcing within the country of production, our approach generated results that differed by over 20% for three of the main consuming countries of Brazilian sugarcane. These results indicate that ignoring sub-national variation in sourcing of produce may (i) generate significant errors in estimations of virtual water flows because of large variations in water footprints at the sub-national level, and (ii) considerably reduce the policy relevance of virtual water accounting, because without sub-national granularity leverage points for practical interventions by decision makers are strongly limited. Given the urgent need to embed the water dimensions in global and national sustainability agendas more efficiently (e.g. Agenda 2030) there is a strong demand for tools that address local impacts on water resources of global trade.

Spatially explicit information on the impacts of water use is especially relevant to support decision makers at local and regional levels to prioritize and implement cost-effective management practices, and in assessments of socio-environmental trade-offs between alternative water uses. For instance, the information generated by our proposed approach could support a better understanding of the role of global agricultural demand in the ongoing water scarcity in the region of Sao Paulo (ABC, 2014). Moreover, the methodology contributes to increased understanding about to which extent local food security and basic access to water may be compromised by water use for commercial plantations in the Northeast of Brazil. For actors along the supply chain, such as traders and the finance sector, our approach illustrates risks associated to sourcing from high water stress areas, i.e. potential disruptions in production, and reputational risks. The increased supply chain transparency can also contribute to design contingency plans ahead of periods of extreme water stress to guarantee their supply, for example by delineating a more diversified sourcing portfolio. This is progressively important in view of ongoing climate change.

Increased transparency, however, does not inform consumers and producers about how they should make their decisions. There are a number of reasons for why producers grow a certain crop in a given location, regardless if this is the most optimal way to use water, or other resources (Wichelns, 2015). Thus, changing consumer behaviour to choose goods and services with low virtual water does not necessarily solve local water management issues. Increased transparency, however, enables an identification of critical hotspots of water stress that are linked to specific supply chain actors and traded commodities. This kind of transparency reveals potential needs to prioritize investments and policy focus between alternative water uses. Furthermore, it also scans the existence of hidden hotspots in remote areas that are far from the consumer's and government's concerns.

The approach presented in this study can improve the understanding of linkages between dynamics of consumption, trade and production systems in the context of water use demands. However, we have focused on conceptualizing and illustrating this approach instead of analysing in-depth the concrete implications for a set of crops, municipalities and even policies in Brazil. Beyond that possibility, this approach could be successfully applied to other countries of production with large spatial heterogeneity in water resources, to other crops for which very different water management practices occur even in the same region of production, or to other environmental dimensions that show a large spatial dependency and heterogeneity. The latter is the case of, for example, linking sub-national material flows with local biodiversity impacts, for which global demand that leads to tropical deforestation may result in several times more embedded biodiversity loss than if consumption is linked to non-forested areas with poor biodiversity values. Green water assessments were not included in this study as green water use cannot be directly related to the water scarcity indicator applied, but moreover because of current methodological and data limitations for accurately assessing green water scarcity (Schyns et al., 2015). In any case, the application of this type of approach to water resources and scarcity should preferably rely on locally adapted water modelling, as well as to include green water assessment and linkages to local environmental impacts of water partitioning and soil moisture availability.

5.5 Conclusion

In this paper we illustrate how improved spatial explicitness and accounting for local conditions of water stress enables an identification of major water users along the supply chain, exemplified by the EU and China, in critical areas of water stress. These estimates were obtained by linking material trade flows from municipal scale sourcing regions, a water footprint model of blue water use and a high-resolution mapping of blue water stress in Brazil.

We argue that by accounting for subnational heterogeneity in virtual water use and water scarcity, it is possible to identify potential trade-offs and regions of concern, linking local pressures to various actors along global supply chains and therefore facilitating multi-stakeholder dialogue to find solutions to water resource management conflicts. Overall, this paper makes a strong case for a more holistic and joint consideration of methods and data allowing to obtain detailed water scarcity and virtual water footprint assessments. This allows for increasing the policy relevance of water assessments and to better support improved sustainability along water-demanding global supply chains. Our proposed approach is well suited to capture spatial heterogeneity in water resources and management in the regions of production; to account for differential sourcing within the borders of a country of production to different regions of consumption; and to relate virtual water flows and local conditions of water stress and demand.

Acknowledgments

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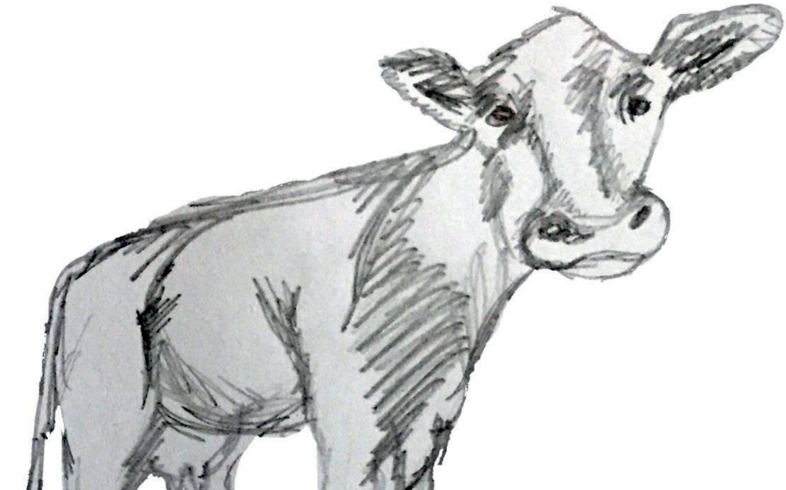

Chapter 6

Consumptive water use for beef production in the Brazilian Cerrado: past and future trends

Ran, Y.[1,2], De Boer, I.J.M.[1], Lannerstad, M.[3], Van Middelaar, C.E.[1]

[1] Animal Production Systems group, Wageningen University & Research, PO Box 338, 6700 AH Wageningen, the Netherlands
[2] Stockholm Environment Institute, Stockholm, Sweden
[3] International Livestock Research Institute, Nairobi, Kenya
[4] Commonwealth Scientific and Industrial Research Organization, Brisbane, Australia

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Abstract

Brazil is a top producer of meat and feed crops in an increasingly global livestock sector. The Brazilian Cerrado hosts about 40% of Brazils cattle herd and has undergone a rapid transition as a result of a large increase in soy production. Beef production is known for its high demand for land and water resources and the sector is expected to grow still further in the coming decades. These trends will put increased pressure on already scarce land and water resources in the Brazilian Cerrado.

To explore potential pathways for beef production to use water in a more sustainable way, this study first estimated and analysed trends in water use for beef production in the Brazilian Cerrado for the period 2010—2016. Second, maximum potential beef production and associated water uses , without requiring additional land resources, were estimated for four distinct production systems to enable exploration of sustainable development of the Brazilian beef sector. The four Cerrado beef production systems were a natural pasture system (NP), an improved pasture system with legumes (IP_{leg}), an improved pasture system with supplementary feeding (IP_{supp}) and a feedlot system (FL).

Results illustrate that water requirements are relatively similar across all systems. The NP system, however, requires the largest amount of water per kg of beef produced, while the FL system is the most water efficient. Analysing the maximum potential beef production on current pasture area in the Cerrado states shows that the FL system can contribute a significant increase in beef production, but also consumes a significant amount of water over cropland that would be suitable for producing more human edible protein from food crops. In contrast to all other three systems, the NP system does not consume any water over cropland and, thus, does not contribute to increased competition over land and water resources with food production. Results from this study show that there are multiple pathways for increasing beef production without significantly increasing feed-food competition over land and water resources, and that low-opportunity cost feeds, such as pasture, could contribute effectively to the sustainable development of the food sector in areas where resources are scarce.

6.1 Introduction

Agriculture today requires approximately two thirds of global freshwater withdrawals for irrigation, and also dominates water use in periods and areas of water scarcity (Poore and Nemecek, 2018). Per capita and total consumption of meat are increasing globally. By 2050, the global population is expected to reach 9 billion, and demand for livestock products is expected to continue to grow as a result of population growth and increasing average incomes (Godfray et al., 2018). At the same time, trade in agricultural commodities constitute a main driver of water availability and land use change today (Hoff, 2009; Rockström et al., 2014) and agricultural and livestock value-chains are becoming increasingly global (Galloway et al., 2007; Meyfroidt et al., 2013). All these factors are inevitably contributing to even greater pressure on global water and land resources.

In contrast to greenhouse gas emissions, water resource use has impacts on local rather than a global scale. The impacts of water resource use largely depend on what type of water is used, that is, groundwater and surface water or rainwater. Furthermore, impacts are specific to the local context that applies during use, such as whether a river basin is experiencing water scarcity or a shortage of rainfall affects crop growth. Brazil, which is one of the largest agricultural producers in the world, is a country that is considered water abundant. More than 70% of available water resources, however, are located in the Amazon basin in the Amazon biome (Figure 1), which is host to just 5% of the Brazilian population (da Silva et al., 2016). Flach et al. (2016) identified key areas of water stress in Brazil using a water stress index to represent a use-to availability ratio including irrigation (see Figure 1). The assessment showed that, even though Brazil receives enough precipitation to be considered water abundant, there are areas outside of the Amazon where water stress is already an area of concern.

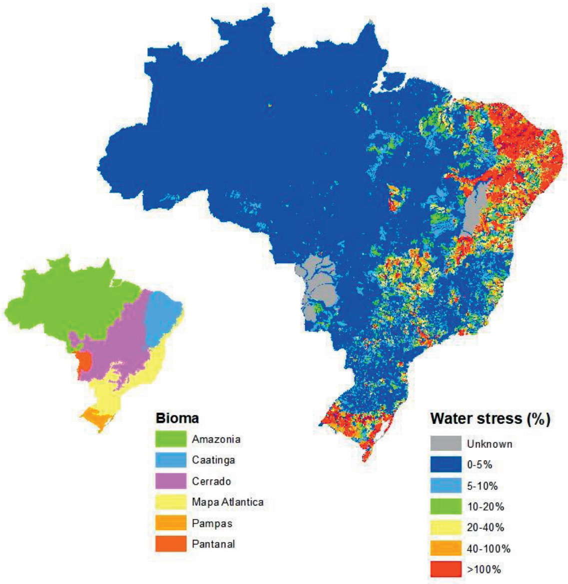

Figure 1: Estimated water stress for Brazilian municipalities. Adapted from Flach et al. (2016)

Brazil is a top global producer and exporter not only of beef but also of soy, which is primarily used for animal feed (FAOSTAT, 2018). Latin America is increasingly supplying agricultural commodities to global markets (Flachsbarth et al., 2015). Initiated by the soybean expansion, in the 1990s the Cerrado region underwent a rapid land use change (e.g. Arima et al., 2011; Beuchle et al., 2015; le Polain de Waroux et al., 2017; Macedo et al., 2012). This land use change has had large biodiversity impacts, increased greenhouse gas emissions and, affected water partitioning, and ultimately affected the hydrological cycle (e.g. Castello and Macedo 2016; Chaplin-Kramer et al. 2015; Coe et al. 2011; Galford et al. 2010).

The Brazilian cattle industry is also currently undergoing intensification (Millen et al., 2011; Palhares et al., 2017; Radobank, 2014), as a result of stricter land use policies, increased competition over land resources and the increasing global demand for livestock products (Millen et al., 2011). As a result, the number of Brazilian cattle finished in feedlots has increased rapidly over the past decade (ABIEC, 2017). Almost 95% of Brazilian cattle are destined for beef production, and the intensification of production has already decreased the length of the cattle cycle, increased stocking rates and efficiency and decreased greenhouse gas emissions per kg of produced beef (Latawiec et al., 2017; Millen et al., 2011).

Despite the increase in beef production, total pasture area in Brazil is decreasing (Dias et al., 2016) and this trend is expected to continue (ABIEC, 2017). Natural pasture area has steadily decreased since the 1940s and it is being replaced by planted grasslands and croplands. Even though the area of Brazilian pasture area is decreasing on average, total pasture area has increased significantly in some states as a result of a large increase in planted pasture, as illustrated in Figure 2.

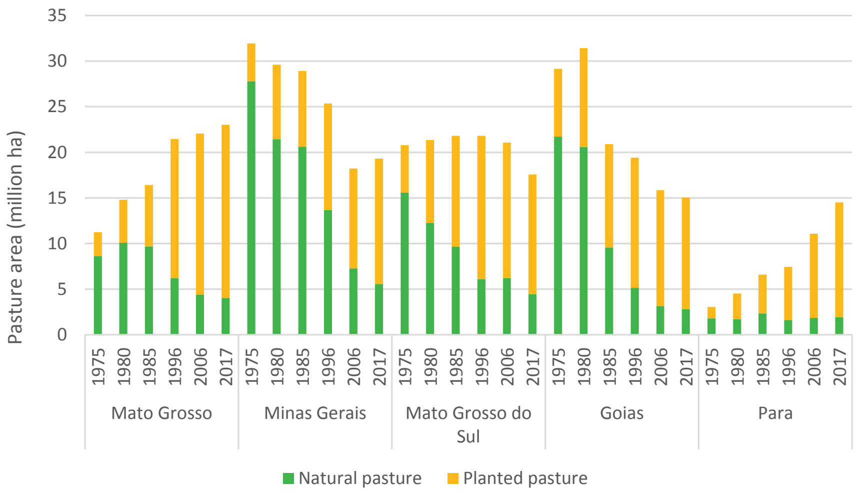

Figure 2: Natural and improved pasture areas in the five largest beef producing states in Brazil: Mato Grosso, Mato Grosso do Sul, Minas Gerais, Goias and Para between 1975—2017.Source: IBGE (2018).

Animals kept in more intensive beef production systems require a larger amount of high protein feed crops, such as soy and maize. These are also suitable for direct human consumption, and require water resources that could be used to produce food crops instead of feed crops. Such animal feed production is therefore in direct competition with food production for available resources, such as water and land van Zanten et al. (2016) and Ran et al. (2017). In addition, indirect competition over water resources can occur when pasture areas appropriate land that is also suitable for crop production (Ran et al., 2017). The intensification of the Brazilian beef sector will therefore affect water use for beef production in terms of the volume and type of water resource used and have potential impacts on the landscape with regard to competition over water resources.

To further investigate the effects of an intensifying beef sector on water resources, the aim of this study is two-fold. First, we investigate trends in water use for Brazilian beef production in the period 2010-2016. Second, we explore the potential pathways for beef production to use water in a more sustainable way, without requiring additional land resources. To this end, we estimate maximum potential beef production in four different production systems and quantify the consequences for water requirements in the Brazilian Cerrado.

6.2 Methods

To further emphasize that water use can refer to use of widely different types of water resources that has significantly different local and regional impacts on an ecosystem, this study separates water resources into *green water*; naturally infiltrated soil moisture available for plant growth, and *blue water*; liquid water in water bodies as rivers, lakes and aquifers (Falkenmark, 1995). Green and blue water, however, are not static pools of water but interchangeable states, and water can shift from one state to the other. The focus, furthermore, is on consumptive water use (CWU), i.e. water withdrawn from a watershed and not discharged to the same watershed because it evaporates, is embodied in plants or is discharged to a different watershed (Falkenmark and Lannerstad, 2005). Since feed production requires approximately 92—98% of the total CWU for livestock production (De Boer et al., 2013; Mekonnen and Hoekstra, 2012), this study considers CWU for livestock feed only.

Figure 3 illustrate the origins of water consumed in the production of livestock feed. The CWU of feed is divided into green and blue water over crop and grasslands. Figure 3 further illustrates how water use, although attributed to exported products such as beef and animal feed crops, is consumed at the location where the production takes place. Thus, the potential local impacts of CWU on the socio-ecological landscape only matter in regard to the region where the water is used. The focus of water resource use estimates has primarily been to identify areas of blue water scarcity (e.g. Quinteiro et al. 2018; Schyns et al. 2019). To exemplify the scarcity of green water, this study also highlights competition over water resources between food and feed production (Ran et al., 2017; Schyns et al., 2015). If green water is used to grow crops, that water is no longer available for other purposes. In other words, the water resources required to produce food, fuel and fibre are limited in a landscape.

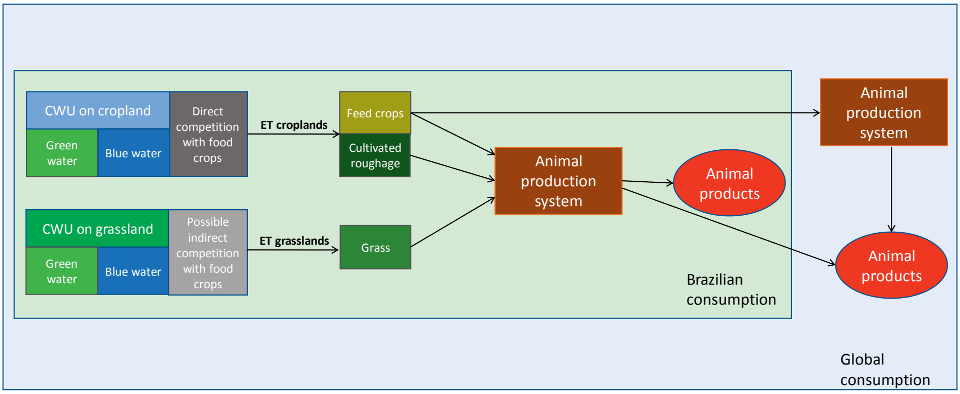

Figure 3: Consumptive water use for livestock feed in Brazilian beef production. Source: Adapted from Ran et al. (2017).

To address our first research aim, we analysed the CWU required to produce all the feed used in the four main beef production systems in 11 states of the Brazilian Cerrado between 2010 and2016 (Mato Grosso (MT), Mato Grosso do Sul (MS), Minas Gerais (MG), Goias (GO), Pará (PA), Bahia (BA), São Paolo (SP), Tocantins (TO), Maranhão (MA), Piauí (PI) and Distrito Federal (DF), see Figure 4)). The Cerrado region is the second largest of six biomes and cover about 24% of Brazil. It also comprises of three major river basins. The area is naturally covered by tropical grasslands and savannah (Rada, 2013). We assume that all the feed used for beef production is produced within each state. Subsequently, we determined the total maximum potential production of beef in these 11 states, based on the availability of pasture land and the specific pasture land requirement for each production system, and quantified its associated water use in 2017.

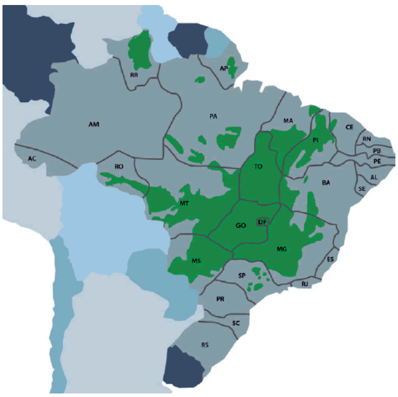

Figure 4: Map of the Brazilian Cerrado (highlighted in green) and the Cerrado states. Key: Mato Grosso (MT), Mato Grosso do Sul (MS), Minas Gerais (MG), Goias (GO), Pará (PA), Bahia (BA), São Paolo (SP), Tocantins (TO), Maranhão (MA), Piauí (PI) and Distrito Federal (DF). Source: Adapted from Lopes and Guilherme (1994).

The four production systems included in this study are described in Cardoso et al. (2016). They are categorized according to their feeding regimes: one natural pasture system (NP), two improved pasture systems — one with pasture vegetation improved with legumes (IP_{leg}) and one with improved pasture and supplementary feeding (IP_{supp}) — and one feedlot system (FL) (see Table 1). These four production systems correspond to feeding regimes 2 to 5 in Cardoso et al. (2016). Depending on the production system, cattle had a starting weight of 30—40 kg and reach a finished state at 420—490 kg (Appendix B, Table B1) and are generally the type of Nellore. Pasture in the region is dominated by the tropical forage grass Brachiaria. The rainy season lasts between November and April, and is followed by a cooler dry season during which precipitation patterns can vary considerably (Cardoso et al., 2016).

Table 1: Relative distribution, indicators of livestock units (LU), stocking rate and land use for beef production systems in the Cerrado.

Production systems Feeding scenarios[b]	% of cattle herd	Supplements (maize and soybeans) (kg)	Stocking rate (LU/ha)	Grassland area[a]	Cropland area[a]
NP	81		1.00	679.5	
IP_{leg}	<1	28.8	1.70	432.1	5.8
IP_{supp}	7	40.3	2.50	293.8	10.7
FL	11	104.0	2.75	267.1	27.2

[a] For a herd based on 400 reproductive females as described in Cardoso et al. (2016)

[b] NP=Natural pasture system; IP_{leg}=Improved pasture system with legumes; IP_{supp}=Improved pasture system with supplementary feeding; FL=Feedlot system.

Natural pasture systems

In the NP production system, feeding is based on Brachiaria pasture only. Animals are given occasional mineral supplements and the pasture is renewed every 10 years by ploughing and liming. No other fertilizers are added. This system is estimated to have applied to an average of 75% of the cattle herd in the Cerrado between 2010 and 2016 (Cardoso et al., 2016).

Improved pasture systems

The IP_{leg} system is based on pasture that has been improved by the introduction of a forage legume, *Stylosanthes spp*. Pastures are renewed every five years by ploughing, including the application of lime and P and K fertilizers. Animal reproduction is not controlled. The pasture is improved by legumes better adapted to local conditions, which enables increased pasture carrying capacity and feed of higher nutritional value (Latawiec et al., 2017). The practice requires careful management of the pasture and has a low adoption rate in the region; less than 1% of the cattle herd is kept in IP_{leg} systems.

In the IP_{supp} system, feeding is based on *B. brizantha* pastures, which also increases pasture productivity. Lime and K fertilizer are applied at the planting stage, while N fertilizers are applied three times during the rainy season. The pasture is renewed every five years by ploughing, including the application of lime and N and K fertilizers. Animal breeding is controlled in this system. It is assumed to have been applied to about 11% of cattle in the Cerrado between 2010 and 2016.

Feedlot system (FL)

In this production system, feeding at the calving and rearing stage is based on fertilized Guinea grass pasture with similar pasture management practices as in IP_{supp}. Cattle are finished in feedlots. Between 2010 and 2016 the system was estimated to be in use for 14% of the Cerrado cattle herd. The feed composition for cattle in feedlots is described in Table 2 (Cardoso et al., 2016).

The animal diet in all four systems, consists mainly of grass from natural or improved pastures. In the more intensive systems (IP_{leg}, IP_{supp} and FL) cattle are fed supplements comprising of maize, soybeans and maize silage, as well as mineral supplements. The feeding regime of each production system is further described below and in Table 2.

Table 2: Feed composition for the four production systems and production stages.

Phase	System	NP	IP_{leg}		IP_{supp}			FL				
	Feed type (%)	Grass	Grass	Maize	Grass	Maize	Soy-beans	Grass	Maize	Soy-beans	Silage	Lime
Calving		100	100		100			100				
Rearing	Heifer	100	100		100			100				
	Steer	100	100		100			100				
Finish	Heifer	100	87.0	13.0	78.8	12.7	8.5		49.5	7.0	40	3.5
	Steer	100	87.6	12.4	79.3	12.4	8.3		49.5	7.0	40	3.5

Source: Adapted from Cardoso et al. (2016)

The dry matter intake (DMI) per animal in each production system was based on Cardoso et al. (2016). They calculated the DMI using the Intergovernmental Panel on Climate Change Tier 2 method (IPCC, 2006), based on body weight and estimated net energy requirement for the different cattle cycle stages and specific to each of the four systems. Estimations follow a developed herd model based on 400 reproducing females which is further described in Appendix B, Tables B1 and B2. DMI and feed composition are then used to calculate the total feed requirement per feed item. Results are calculated to represent an annual cycle.

To enable comparison with previous estimates of water use in beef production, we expressed water resource use per kg of beef produced in each production system and year, using carcass weights as illustrated in Table 3.

Table 3: Carcass weight and yield for four cattle production systems.

System	NP	IP_{leg}	IP_{supp}	FL
Male carcass weight (kg)	240	250	250	265
Female carcass weight (kg)	210	220	220	235
Male carcass yield %	0,51	0,52	0,52	0,54
Female carcass yield %	0,5	0,5	0,5	0,52

Source: Cardoso et al. (2016)
Key: Natural pasture (NP), improved pasture with legumes (IPleg), improved pasture with supplementary feeding (IPsupp) and feedlot finishing (FL).

6

6.2.1 Calculating consumptive water use for Cerrado beef production systems

To quantify the CWU required to produce feed in each state in the period 2010—2016, we first determined the CWU for each feed ingredient. For animal feed crops we used the state specific crop water requirements from Mekonnen and Hoekstra (2011b), which calculated a global estimate of crop water footprints on a 5 by 5 arc minute resolution (as further described in Appendix B Section B1 and B2). The estimates from this model account for climatic and local conditions and are widely applied, and therefore provide estimates that are comparable with other studies. For the pasture water requirement, we used the approach described in Zhang et al. (2001) for estimating pasture evapotranspiration (equation Eq B1 in Appendix B). Spatial precipitation data was obtained from the CROPWAT 8.0 model (FAO, 2010b, 2014) and the CLIMWAT database (FAO, 2010a), based on 30-year averages of precipitation patterns.

Subsequently, the CWU per ton dry matter of feed crops, silage and grass were multiplied by the amount of feed required to produce 1 kg of beef in each of the four production systems, yielding an average estimate of CWU per kg of beef. This CWU per kg of beef distinguished between green and blue water resources use, and the amount of green water used associated with grass production or feed crop production (e.g. maize, soy etc.).

Finally, the total CWU per system and state for the 2010—2016 period was calculated based on the CWU per kg of (slaughtered) beef, the number of beef cattle slaughtered in 2010—2016, (IBGE, 2018; SIDRA, 2018), and the relative distribution of beef cattle across production systems. Cattle distribution between systems per state was calculated using data from ANUALPEC (2018)which showed number of cattle in feedlots and improved pastures and that the IP_{leg} system covers less than 1% of beef cattle in the region (Cardoso et al., 2016). ANUALPEC data covers nine of the 11 Cerrado states. For the two remaining states, Maranhão and Piauí, we estimate an average distribution of cattle based on the remaining Cerrado states.

6.2.2 Maximum potential beef production and associated consumptive water use

The second aim of this study was to explore potential pathways for beef production to use water in a more sustainable way, without requiring additional land resources. To address this, we calculated how much beef could be produced in each state if all beef production was carried out according to one of the four systems, based on current availability of pasture in that region, and estimated the CWU associated with such production.

We quantified this maximum potential production for each system in a state, which was determined by the available grazing area per state and literature values of system-specific stocking rates, grassland requirement, slaughter rate and carcass weight ratio (see Tables 1, 3 and Appendix B Table B2). State-specific pasture areas in 2017 were derived from the agricultural census (IBGE, 2018). Finally, we calculated the water required to produce that beef, categorized over crop and grassland to illustrate the competition over land and water resources between feed and food production.

To enable validation of our results with the agricultural census data for 2017, which is reported as live animals per state, we calculated the potential maximum cattle herd for each state and system based on the carcass weight percentage of live weights, as stated in Cardoso et al. (2016).

6.3 Results

First, the CWU per kg of beef for each production system is illustrated. Second, total CWU by production system and state are presented for the period 2010—2016 to illustrate changes in total CWU over time. Third, maximum beef production and the corresponding CWU, categorized over crop and grassland for each production system and state are presented to enable a comparison and discussion of the potential effects on beef production volumes and CWU of future scenarios, varying the relative distribution of cattle between production systems.

6.3.1 Consumptive water use per kg of beef for each production system

Figure 5 presents the CWU in l/kg beef for the four main beef production systems in the Brazilian Cerrado. Total CWU is made up of green water on pasture (GW_P), green water on cropland (GW_C) and blue water on cropland (BW_c).

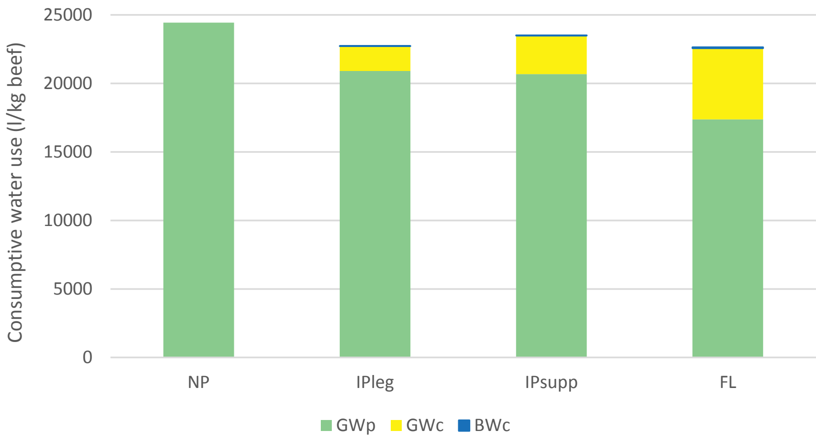

Figure 5: Consumptive water use for four beef production systems in the Brazilian Cerrado. Key: Natural pasture (NP), improved leguminous pasture (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and a feedlot system categorized as green water over pasture (GW_p), green water over cropland (GW_c) and blue water over cropland (BW_c).

6

All four systems largely rely on pasture to feed cattle. Thus, the majority of CWU for beef in the Cerrado region is GW_p. There are a number of notable trends in the water requirement for beef in these production systems. Although the differences between the four production systems are relatively small, the natural pasture system demands the largest amount of water, about 24,400 l/kg. The IP_{supp} system requires about 23,200 l/kg while the IP_{leg} and the feedlot system have about the same CWU per kg of beef produced: 22,700 l/kg and 22,600 respecitvely. The NP system relies only on green water on pasture. The IP_{leg}, IP_{supp} and FL systems, however, also require green water resources on cropland and a small amount of blue water on cropland. The blue and green water on cropland are significantly higher in the feedlot system, constituting 23% of total CWU, followed by the IP_{supp} at 12% of the total CWU and IP_{leg}, at about 8% of the total CWU on average. Blue water use constitutes less than 1% in all systems.

6.3.2 Consumptive water use for Cerrado beef production systems over time

Figure 6 illustrates the total CWU for beef production in the four main production systems in the Cerrado between 2010 and2016. There is a general increase in CWU for Cerrado beef production of 7% during the period, from almost 45 Mm^3 to just above 48 Mm^3, as a result of an increase in the size of the cattle herd in the region (IBGE, 2018). The NP system has a significantly larger total CWU than the other three systems, despite the small decrease in CWU over time, from about 32 Mm^3 in 2010 to about 31 Mm^3 in 2016. The NP system uses 2% less of the total CWU for Cerrado beef in 2016 than it did in 2010, corresponding with the fact that the system has a smaller share of the cattle in the region.

Alhtough the total CWU of the FL system is still relatively small in comparison with the total CWU of the NP system, the CWU of the FL system increased by almost 50%, from about 6 Mm^3 in 2010 to almost 10 Mm^3 in 2016. It comprised about 14% of total CWU in the Cerrado region in 2010 but more than 20% in 2016. The contribution of IP_{sup} to total CWU has also increased over time, corresponding with an almost 10% increase in the size of the Cerrado cattle herd between 2010 and 2016. The IP_{leg} remained small during the entire period.

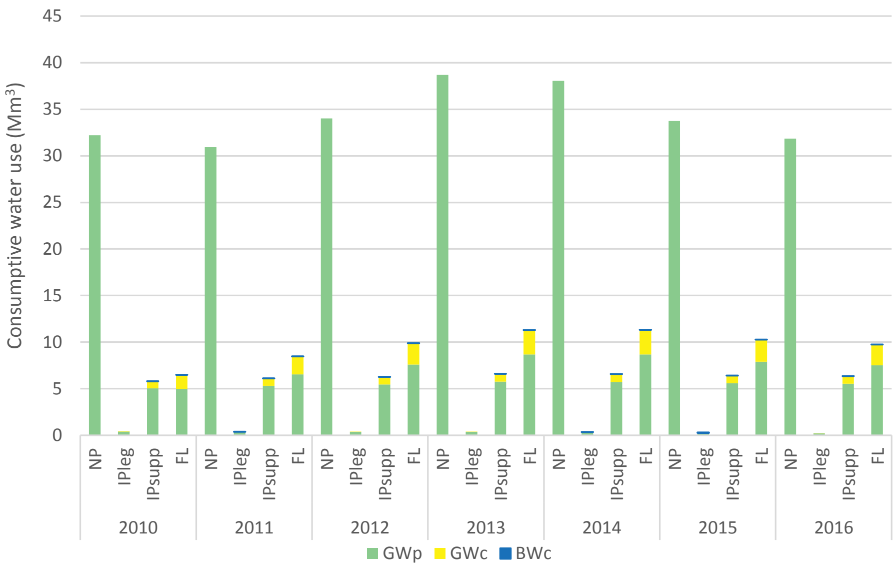

Figure 6: Total consumptive water use for beef produced in the Brazilian Cerrado states inbetween 2010—2016 for four livestock production systems

Key: natural pasture (NP), improved leguminous pasture (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and a feedlot system (FL).

Notes: Water resources are categorized into green water on pasture (GW_P), green water on cropland (GW_C) and blue water on cropland (BW_C).

As Figure 6 shows, the CWU in the NP system comprises only of the green water required to support the production of grass. The CWU of the IP_{leg}, IP_{supp} and FL systems, however, also constitute green and blue water use for the cultivation of maize and soybeans. Both blue and green CWU on cropland increase over time in 2010—2016, and the largest increase is associated with green water for cropland in the FL system. The demand for maize and soy for use as feed in the four systems studied, although increasing over time, remains low across all states. About 21% of the maize and between 35 and 95% of the soy produced in each state is currently exported and largely used as livestock feed elsewhere (Appendix B, Table B4).

6.3.3 Maximum potential beef production and associated consumptive water use

The maximum beef production per state for each of the four production systems is shown in Figure 7. Potential beef production is largest for the FL system in all states, followed by IP_{supp} and IP_{leg}, and lastly the NP system. Mato Grosso has the greatest potential for producing beef based on current pasture area; potentially almost 1.2 million tons (MT) in the feedlot system, compared with about 1.0 MT in Minas Gerais and almost 0.9 MT in Mato Grosso do Sul. If all cattle were reared in the NP system, Mato Grosso could potentially produce 0.35 MT of beef, compared to 0.29 MT in Minas Gerais and 0.27 MT in Mato Grosso do Sul.

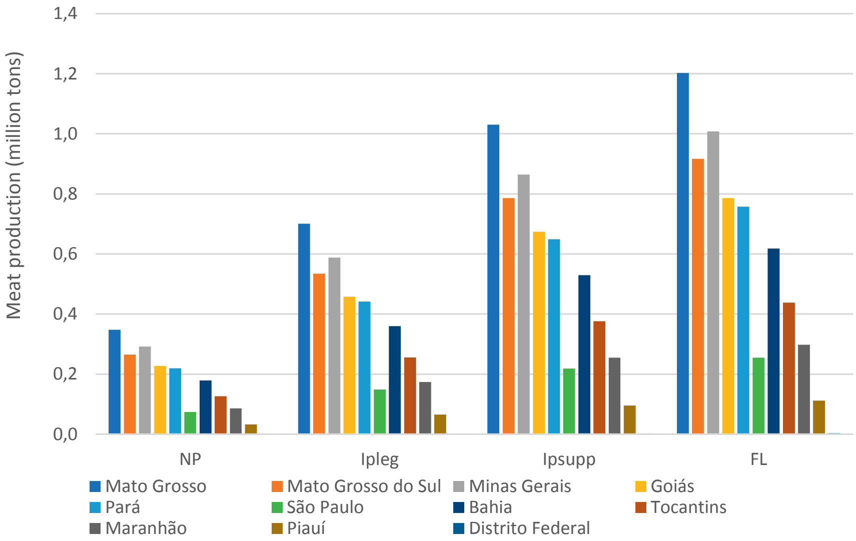

Figure 7: Maximum beef production by production system and state.
Note: The production systems are natural pasture (NP), improved leguminous pasture (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and a feedlot system (FL) in each state in the Brazilian Cerrado.

To validate our results, we compared calculations of the maximum potential cattle herd for each state with reported state cattle herds in 2017, provided in the agricultural census (IBGE, 2018). This comparison (Table 4) shows that our estimated maximum cattle herds for each system are generally aligned with reported herd sizes, with the exception of Bahia, Piauí and São Paolo. This result provides further confirmation that actual production is primarily carried out according to the NP system, but a significant share of the production is produced in more intensive systems, and that the distribution of cattle between production systems is similar to the distribution of cattle estimated in this study (see the methods section above).

In Bahia and Piauí the reported herd size is lower than our estimations for the NP system, indicating that cattle stocking rates may be lower in these states, or that they do not keep cattle on all available pasture land. In São Paolo, the reported herd size is higher than the largest maximum potential herd size according to the estimates in this study. This indicates that cattle in São Paolo are reared more intensively than in these four systems, which is verified by expert opinion and the fact that São Paolo has the largest number of cattle finished in feedlots (ANUALPEC, 2018), which means that they rely on feed production on cropland to a larger extent.

Table 4: Heads of cattle calculated for the four beef production systems by state.

Cattle numbers (million heads/state)	NP	IP_{leg}	IP_{supp}	FL	Census herd[a]
Mato Grosso	22.2	33.4	46.4	48.9	29.7
Mato Grosso do Sul	16.9	25.5	35.4	37.3	21.5
Minas Gerais	18.6	28.0	38.9	40.9	22.0
Goias	14.5	21.8	30.3	31.9	22.8
Para	14.0	21.0	29.2	30.8	20.6
São Paolo	4.7	7.1	9.8	10.3	11.1
Bahia	11.4	17.2	23.8	25.1	10.0
Tocantins	8.1	12.2	16.9	17.8	8.7
Maranhão	5.5	8.3	11.5	12.1	7.7
Piauí	2.1	3.1	4.3	4.5	1.6
Distrito Federal	0.1	0.1	0.1	0.1	0.1

[a] Data from the 2017 agricultural census (IBGE, 2018)
Key: Natural pasture (NP), improve pasture with legumes (IPleg), improved pasture with supplementary feeding (IPsupp) and feedlot (FL).

If all cattle were kept according to the NP system, there would be a general decrease in the size of the cattle herd of 27% on average, meaning that this system would not be able to achieve the estimated beef production volume for 2017. In contrast, a cattle herd based exclusively on the FL system would be between 30 and 50% larger than the current cattle herd on average. The two improved pasture production systems would generate an increase of the maximum cattle herd with 15% for IP_{leg} and almost 40% for IP_{supp} on average.

The total CWU for the maximum beef production, in each state is illustrated by system in Figure 8. The relative difference in the CWU required for maximum beef production is similar to the production difference for each system and state in the water consumed over grassland. While the FL system has the potential to produce significantly more beef overall, total green and blue CWU are also largest for this system in all states. However, if only water over pasture land is taken into account, the FL system requires somewhat less water, that is 31 Mm^3 in Mato Grosso, than the IP_{supp} system which requires 32 Mm^3 in the same state. The FL system, however, requires an additional 8 Mm^3 of green and blue water over cropland compared to only 4 Mm^3 for IP_{supp} in Mato

Grosso. Thus, total CWU still exceeds that of the improved pasture system with supplementary feeding in all states.

With the exception of Tocantins which has the highest CWU per ton of DM of grass of all the states, the variation in CWU between states and production systems is larger on cropland than on grassland. For example, the FL system, would require a larger volume of CWU on cropland in Mato Grosso do Sul compared to Mato Grosso, although Mato Grosso has a larger maximum beef production volume. With the IP_{supp} system, this trend is reversed; the CWU on cropland and the maximum production are larger in Mato Grosso than in Mato Grosso do Sul.

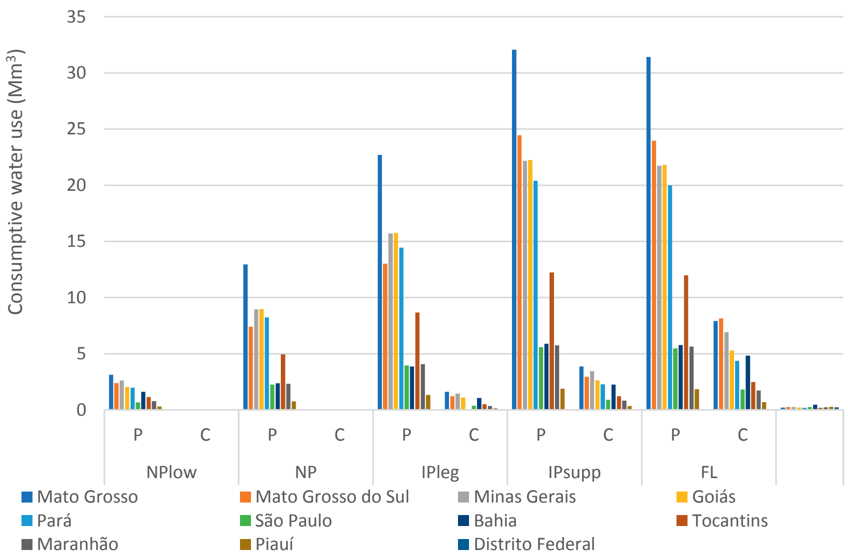

Figure 8: Total CWU requirement for maximum beef production, in each state and by system
Key: Natural pasture (NP), improved leguminous pasture (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and a feedlot system (FL).
Note: CWU is categorised according to whether it is consumed over pasture (P) or cropland (C).

Like the previous CWU estimates (Figure 5 and 6), the share of CWU that is evapotranspired over cropland is significantly larger in the feedlot system, at about 18—28% compared to 8—15% in the IP_{supp}, 6—10% in IP_{leg} and zero in the NP system. The demand for maize and soy for use as animal feed would also increase significantly for maximum production in the more intensive systems in comparison with the volumes required for Cerrado beef production in 2010-2016 (Appendix B, Table B3), sometimes exceeding the volume in each state that remains after exports (see Appendix B Table B4).

6.4 Discussion

This first aim of this study was to investigate CWU and current trends for CWU in beef production systems in the Brazilian Cerrado. The results show that all four beef production systems largely depend on green water use, while blue water constitutes less than 1% of total CWU for all systems. The NP system uses the largest amount of water per kg of beef, whereas the FL system is the most water efficient. The CWU estimates are in close proximity to other studies that have estimated CWU in beef production systems in Brazil (Gerbens-Leenes et al. 2013), and global and regional averages (Mekonnen and Hoekstra, 2012; Molden, 2007a; 2007b).

The differences between production systems in this study are, however, relatively small, within a range of 22,000—24,000 l/kg of beef, compared to other studies. For example, in the studies by Palhares et al. (2017) and Ran et al. (2017), which look at beef production systems in Brazil and Uruguay respectively, the more intensive production systems had only about half the level of CWU of the extensive production systems. The systems analysed in this study, however, are chosen because they are the most prevalent in the region. Thus, the intensive systems in this study are still largely pasture-based, compared to the intensive production systems studied, for example in Ran et al. (2017) and Palhares et al. (2017). This generates the small differences in total volumetric CWU between systems. The water productivity increase as a result of intensification is due to improved feed efficiency and results from the fact that supplementary feeding and better pasture quality improves feed efficiency and reduce the period required to reach the same final body weight.

Our second aim was to explore potential pathways for beef production to use water resources in a more sustainable way but without exhausting additional land resources. To this end, we determined the maximum potential to produce beef from each system in each state and quantified the associated water use. This maximum potential beef production was restricted by the amount of pasture area available in each state in 2017, defined in order to illustrate a natural resource management goal. We assumed that all pasture was used for beef production. Brazilian cattle production systems still largely rely on pasture and although the total pasture area is currently decreasing, pasture expansion has caused new areas to be explored for agricultural purposes (Dias et al., 2016), proving that pasture is still in demand for the large cattle sector. While Schyns et al. (2019) also estimated CWU with regard to a restriction in the land resources available for human appropriation, they aimed to maintain biodiversity and, thus, did not explore changes in water use productivity by optimising management, or the exploitation of resources in a landscape setting with a multitude of users.

Our estimates of current beef production volumes in the Cerrado states of between 1.8 and 6.4 MT carcass weight correspond to about half the total amount of beef produced in Brazil in 2017 (ABIEC, 2017; FAOSTAT, 2018). This corresponds well with estimates of the Cerrado's contribution to Brazil's beef production (Cardoso et al., 2016; Lahsen et al., 2016; Rada, 2013). The NP system is assumed to be the dominant production system in the region today (Cardoso et al., 2016). However, the estimated maximum beef production possible under the NP system would constitute only about 19% of the total beef produced in Brazil in 2017. This would fall below the assumed 40—55% of production that originated from the Cerrado (Cardoso et al., 2016; Lahsen et al., 2016; Rada, 2013). Thus, we can assume that beef production in the Cerrado cannot only be carried out using the NP system if current and projected production volumes are to be achieved.

If beef were produced only in the FL system, the Cerrado states could contribute up to 67% of total Brazilian beef production in 2017, using the same pasture area as the NP system.

We also validated our results by comparing the maximum potential cattle herd in each state to the actual cattle herd presented in the agricultural census of 2017 (IBGE, 2018). This comparison further strengthened the assumption that, in order to achieve current production volumes, beef cattle cannot only be kept according to the management practices of the NP system outlined in this study, but must also be reared on improved pasture and with some supplementary feeding. The actual cattle herd was, on average, about 25% larger than our calculations for the maximum cattle herd in the NP system, and 15% smaller than our estimated maximum herd for the IP_{leg} system. Production according to the FL system would result in a significant increase in productivity, resulting in a 40% larger cattle herd on average than the actual cattle herd across all Cerrado states. The productivity increase in the feedlot system is a consequence of a lower land use requirement, higher system productivity and higher density of cattle per ha of pasture.

There are large differences in maximum potential beef production volumes between states, due to differences in available pasture areas. Our calculated maximum potential production volume is largest in Mato Grosso, followed by Mato Grosso do Sul and Minas Gerais. These three states are also reported to have the largest cattle herds in 2017 (IBGE, 2018), which indicates that pasture area is indeed one of the factors that limit cattle herd sizes in the Cerrado states.

In line with the fact that the FL system can contribute significantly more beef than the other three production systems, this system would also require the largest total CWU to produce that beef, followed by the IP_{supp} system. The NP system would, by contrast, require a lower total CWU to produce its beef. However, the increase in CWU in the FL system is smaller than the potential increase in beef production in relative terms, indicating that water productivity would still increase despite having the largest total CWU. As is mentioned above, however, the FL system requires significantly more water over croplands than the other three systems, resulting in higher direct feed-food competition which is discussed further below.

In addition to more water being consumed over cropland with an increase in intensification, blue water resource use also increases (e.g. Flachsbarth et al., 2015), as feed crops are irrigated to a higher extent than pasture. This is verified by the results of our study although blue water use remains low across systems and states. In a recent study, da Silva et al. (2016) estimated water scarcity indices for different states in Brazil. They identified that even though the Cerrado region is not experiencing large-scale water scarcity as a whole, there are areas where the water scarcity index is already higher than 50. Examples of states with a high water scarcity index are Bahia and São Paolo, which indicates a sensitivity to increased competition over resources and higher vulnerability to climate change induced impacts on, for example precipitation patterns. Flachsbarth et al. (2015) identifies Tocantins as a critical region for water stress, where additional blue water use for irrigation may exacerbate water stress. In such regions, even though blue water use is currently low in the systems analysed, efficient use of green water is imperative in order to maintain agricultural productivity and minimise the required addition of blue water as irrigation (Rockström et al., 2007).

6.4.1 Increased feed-food competition with intensification

Brazilian grain production is expected to further increase by 20% over the coming decade and to require a 15% larger area for crop cultivation (MAPA, 2017). In addition, although the cattle herd is expected to remain about the same size between 2016 and 2026, there is a projected increase in production of about 20%, and beef exports are estimated to increase by almost 40% while the area of Brazilian pasture is projected to decrease by about 10 million ha in the same time period (ABIEC, 2017).

To achieve these positive projections for agricultural production in Brazil, the beef sector must increase its productivity (e.g. Latawiec et al. 2017; Palhares et al. 2017; Rada 2013; Soterroni et al. 2018). Soterroni et al. (2018) project that, in order to comply with the forest code — a legislation to stop illegal deforestation — the cattle productivity per hectare increase required is estimated at 56%, achieved through a combination of increased supplementary feeding and semi-intensive pasture management to avoid further pasture area expansion. The study further estimates that cropland area in the Cerrado must expand by more than 50% and, for the Cerrado region to comply with the forest code, the use of non-productive areas, currently not used for agricultural production, must double by 2050.

Thus, if we want to maintain the growing trend for beef production in and exports from the region, there is an obvious requirement for an intensification of the beef production systems most prevalent in the Cerrado region today. Although water productivity is increasing in the more intensive system in this study, there is also a higher volume of CWU evapotranspired over cropland, as a consequence of supplementary feeding with maize, soybeans and maize silage. This, in turn, results in competition over food and resources, for example water and land, between livestock and humans (Wilkinson and Lee, 2018). Such feed-food competition will increase further as livestock production systems rely more and more on feed crops suitable for human consumption (Damerau et al., 2019). This corresponds with the findings of Ran et al. (2017), which estimate that beef production systems that use large amounts of water over land that can be directly used for food crop production would produce human digestible protein more efficiently by cultivating food crops rather than crops for animal feed. Today, about 40% of global arable land is dedicated to producing feed for animal systems (Mottet et al., 2017), but there is currently no way to produce human edible food using croplands to feed animals that is as effective as directly producing food crops on the same land (Foley et al., 2011).

The FL system in this study uses significantly more of both green and blue, water over cropland than the other three systems per kg of beef, which means greater direct competition with alternative uses of water resources, such as food production. The proportion of Brazilian cattle finished in feedlots has already increased from about 8% in 2006 to 11% in 2015 (ABIEC, 2017), and the "boom" of feedlot operations is a continuing strong trend. All the systems in this study, however, are largely pasture-based in comparison with feedlot systems in, for example, the United States. Brazilian feedlots are expected to become larger and the percentage of roughages in animal diets is expected to decrease in favour of additional feed crops (Millen et al., 2011). Such development could result in an even higher CWU from feedlots in future, as Palhares et al. (2017) found that in order to minimise pressure on water resources, feedlot operations should increase the share of roughages from by-products in animal diets as they have a lower green water use than other types of feed. This would, however, be likely to result in decreased beef productivity. In addition, the body weight at which cattle enter feedlots has already decreased, which is expected to be a continuing trend (Millen et al., 2011; Millen et al., 2009). This means that cattle are confined for a longer period of time. In general, these factors indicate that even more crops will be required for animal feed in feedlot operations compared to today, generating higher CWU over croplands. Another way to intensify Brazilian cattle systems is to increase stocking rates, which have been generally low in Brazil (Lathuillière et al., 2012), but increasing over time (Dias et al., 2016). This intensification could negatively affect soil quality and ultimately decrease soil porosity and infiltration rates (Latawiec et al., 2017). Thus, it is important to consider the implications of intensification spatially, to consider the trade-offs between the use of several natural resources, and to identify competing uses and use dynamics of the same resources within the landscape to ensure sustainable agriculture management practices.

Natural pasture areas are currently decreasing in the Cerrado and being replaced with cropland areas and cultivated pastures (Beuchle et al., 2015; Dias et al., 2016; IBGE, 2018). This means that, in addition to an increase in direct feed-food competition, the large amount of green water used over pasture lands in all four systems may be used over land that is potentially suitable for crop production. These findings are verified by the study by Mottet et al. (2017), which shows that many pasture areas in the Brazilian Cerrado are indeed suitable for crop production, although compared to the areas currently in use for crop production, pasture areas are said to be marginal (McManus et al., 2016). If pasture areas in use for beef production are indeed suitable for crop production, the opportunity cost of using such water and land resources would be higher (Ran et al., 2017; van Zanten et al., 2016) and result in greater feed-food competition over CWU and land use.

Land use change could potentially result in greater green and blue water scarcity as a result of impacts on runoff, infiltration, erosion and evapotranspiration (Pradinaud et al., 2019). Such impacts will ultimately affect the hydrological cycle, in addition to other environmental impacts in the landscape. The Cerrado grasslands are biodiversity hotspots and the region provides a multitude of ecosystem services, such as climate regulation, clean freshwater, formation of key river basins, recharging of underground aquifers and hydropower electricity. The species of the Cerrado savannah have adapted to the arid climate and developed a deep root system that ensures that precipitation and surface water are infiltrated and recharge deep soil water reservoirs. If natural pasture areas are removed and replaced with croplands and planted pasture lands inhabited by species without such adaptation, this vital ecosystem service for agricultural production will be lost (Lahsen et al., 2016). This illustrates that aiming only to minimise feed-food competition in a landscape does not ensure that other vital ecosystem functions in the landscape are preserved. Instead, estimates of increased competition for natural resources should include not only competition between the production of food and feed, but also competition for the multitude of ecosystem services and functions that depend on the common pool of natural resources.

6.4.2 Future prospects for water use in the Brazilian beef sector

As illustrated by comparing estimates of maximum and actual beef production in the Cerrado states, there is an opportunity to increase beef production by moving a proportion of the large number of cattle currently kept according to the NP system, to one of the three more intensive systems. Productivity increases can be gained without having to increase the number of cattle in feedlots. For example, shifting 20% of the cattle from the NP system to the IP_{leg} system would result in a 20—30% increase in beef production and almost double water productivity compared to the NP system. In addition, such a shift would only result in a 2% increase in CWU over cropland. Shifting cattle to the IP_{supp} system would result in an even larger increase in water productivity but consequently also a much greater increase in water used over croplands.

The IP_{leg} system therefore provides an opportunity to intensify Brazilian beef production without a huge increase in supplementary feeding. This, however, would require producers to consider another breed, since the Nellore breed currently used does not have the potential to assimilate improved forage quality into much higher weight gain than today (Cardoso et al., 2016). A large increase in beef produced according to the IP_{leg} system is therefore highly unlikely under current conditions. However, a study by Latawiec et al. (2017) identified that adopting good management practices for pastures, that is improving pastures, resulted in improved productivity and ultimately in higher incomes for farmers. An increase in supplementary feeding with feed stuffs of low opportunity cost, such as by-products that humans cannot or do not want to consume directly, and grass (van Zanten et al., 2018) also provides an opportunity to intensify beef production without a significant increase in grains and cereals being fed to animals. Thus, low-opportunity cost feed stuffs result in a net contribution to overcoming food insecurity by turning human inedible waste streams into food products of high nutritional value.

To conclude, this paper shows that there are a number of potential ways to decrease CWU in beef production at the same time as increasing cattle production and productivity without a large complementary increase in feed-food competition. In a recent review, van Zanten et al. (2018), identified two dominant pathways for the debate on how livestock production can best contribute to increasing food security. The first, the production pathway, would mean that we have to produce more animal products to meet the increasing demand, and can reduce environmental footprints of animal products by sustainable intensification. The second pathway, the consumption pathway, argues that production of animal products is resource-intensive and should therefore be avoided, and promote vegetarian and vegan diets to reduce the environmental impacts of agricultural production. However, the study also identifies a third, less explored, pathway, where animals are fed products that human cannot or do not want to consume directly, or where animals are fed on grasslands with low ability to support other types of agricultural production (van Zanten et al., 2018). The results from this study support exploration of this third 'low-impact´ livestock' pathway for minimising competition over natural resources in agricultural areas such as the Brazilian Cerrado, where competition over resources tends to be high.

6.5 Conclusions

As a result of the expected increase in both livestock and agricultural production in the Cerrado region of Brazil, there is a drive to intensify beef production systems through increased supplementary feeding with crops. Although beef produced in more intensive production systems has a higher water use efficiency than beef from pasture-based systems, intensification of beef production could result in greater green and blue water scarcity as a result of increased irrigation, alterations to the hydrological cycle due to land use change, and increased feed-food competition.

To achieve increased productivity in beef production systems while at the same time avoiding unnecessary resource competition, there is a need to explore pathways that ensure sustainable use of water and land resources in a complex landscape setting with a multitude of potential uses across local and global markets. To avoid policy recommendations that improve water use along the value chain of beef but result in other negative costs in the landscape, the potential effects on other natural resources, ecosystem functions and human uses, should be considered at the same time.

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Chapter 7

General discussion

7.1 Introduction

The key challenge of today is to produce enough nutritious food for a growing and increasingly affluent global population while avoiding unsustainable use of natural capital that results in a loss of key functions of our global social-ecological system. Water is one of these key natural resources and both green and blue water resources are essential to the global food system and the livestock sector. Most studies that address water use in livestock production, however, do not address the local impacts and effects of both green and blue consumptive water use (CWU) in the landscape.

To address this knowledge gap, the two objectives of this thesis are to improve our understanding of the effects of CWU (both blue and green) in a landscape, and to develop and apply a method that better assesses these effects of CWU in livestock systems. We defined a landscape as an area with a multitude of functions and users that share the same land and water resources, such as production of food, feed, fuel, fibre and maintenance of biodiversity and ecosystem services.

The thesis begins with two chapters that contribute to the discussion on the complexity of assessing the CWU of livestock systems, and the different methodologies available for such assessments. I then propose a framework for how to better illustrate the effects of CWU in the landscape by addressing feed-food competition over water resources. The thesis further discusses the application and extension of such a framework to the multitude of users and functions that are dependent on freshwater availability in a landscape. This general discussion follows the same logic when discussing the thesis findings. Finally, the main conclusions from the thesis are presented.

7.2 Complexity of assessing consumptive water use in livestock systems

Water resources are complex in the sense that they can be seen as both renewable (Pradinaud et al., 2019) and limited in space and time (Schyns et al., 2019). The complexity of water resources and use also stems from differences in types of water use. For example, plants that use rainwater for growth will have different impacts in a landscape than irrigation water extracted from aquifers to increase agricultural productivity. To illustrate such differences, water resources can be differentiated into blue water, that is, freshwater in lakes, rivers and aquifers, and green water, that is, rainfall available as soil moisture for plant growth in the unsaturated zone (Falkenmark, 1995). Thus, combining them into a single volumetric estimate of water use, as was originally done in water use assessments for livestock, can obscure the huge complexity of CWU and the local impacts of the resources used.

Blue water consumption directly decreases the availability of blue water resources, thereby impacting other users of that water. The impact of green CWU, however, is directly linked to the landscape. The soil moisture that is consumed by one crop will no longer be available for use for purposes other than crop production in that area and at that specific time. Thus, green water resources are land bound (Falkenmark and Folke, 2010; Falkenmark and Rockström, 2006; Ridoutt and Pfister, 2010; Schyns et al., 2019), and spatially restricted in contrast to blue water resources that can be abstracted in one location and applied elsewhere.

Green water will, in contrast to blue water, be used over land in spite of human activities. To assess the sustainability of green water use, therefore, it is important to also consider the opportunity costs of the associated land. In terms of food production, for example, the relatively large use of green water associated with ruminants, grazing native grasslands with low opportunity costs for food production, might be more efficient than a smaller use of green water by monogastrics if that water is used to produce feed on land suitable for food crop production. In the latter situation, there is a competition over resources, e.g. water and land, between livestock (feed production) and humans (food production) (Wilkinson and Lee, 2018). Such feed-food competition will increase further as livestock production systems come to rely more and more on feed crops suitable for human consumption or produced on land suitable for food production (Damerau et al., 2019).

Moreover, land cover change and land use management can alter how much green water is evapotranspired (e.g. Pradinaud et al. 2019; Schyns et al. 2017). For example, converting native grassland into cropland will generate a slight increase in green water use on average per ha of land (e.g. Lathuillière et al. 2018b). The difference in water appropriation between grassland and croplands, however, is relatively small in comparison with the difference between forests and grass- or croplands and will not therefore generate a major change in the hydrological landscape (Lathuillière et al., 2018b). From a long-term perspective, land cover change, such as deforestation, may result in less water vapour being emitted to the atmosphere, potentially resulting in decreased precipitation on a regional scale (e.g. Gordon et al. 2005; Keys et al. 2014; Keys et al. 2012a) and ultimately affecting green and blue water availability. Increased irrigation could counteract such changes by increasing the water vapour return flow to the atmosphere (Rockström et al., 2005; Rost et al., 2008a) via regional precipitation recycling (e.g. Quinteiro et al. 2015). This addition, however, in contrast to non-irrigated agriculture, generally originates from blue water.

An additional complexity of assessing CWU is that green and blue water resources are interlinked. A change in land cover, for example resulting from cropland expansion into forested areas can increase run-off and thus increase downstream blue water availability (Karlberg et al., 2009; Pradinaud et al., 2019). In addition, deep-rooted trees may access blue water reservoirs as well as the green water available as soil moisture. This makes CWU for forestry, divided into blue and green water, particularly difficult to estimate (Quinteiro et al., 2018).

In order to understand the complexity of CWU, and the linkages to ecosystem functions, Chapter two studies the relation between CWU to ecosystem services and Chapter three reviews existing methods of CWU assessments for livestock products. The findings in Chapter two and three highlight two key elements that must to be considered in CWU assessments of livestock systems if they are to be able to inform a more sustainable management of water resources. First, CWU assessments must *acknowledge and distinguish the different types of water available for use* and the different impacts of their use in the landscape. Second, consideration of *green water resources should always be in connection with the land use*, that is, on the land over which the water is consumed, as green water use depends on the specific context of the land use, and on land management decisions. One way to connect green water and land use, as identified in Chapter three, is to include competition over water resources in CWU assessments, which will be further described in Section 7.3 of this general discussion.

To provide the context for Section 7.3 I will further elaborate on the details of how to connect green water and land use, I will first describe the history of assessment methods of CWU for livestock and highlight how they have dealt with above mentioned key elements.

7.3 Different methodological approaches to assess consumptive water use in livestock systems

Historically, CWU estimates of agriculture and livestock systems have largely focused on blue water use (e.g. Schyns et al. 2019). The demand for blue water, primarily for use by households, industry and in irrigation of agriculture, is steadily increasing and scarcity of blue water pose a threat to a sustainable human society. Mekonnen and Hoekstra (2016) estimate that four billion people experience severe freshwater scarcity for at least part of the year. Agriculture is the single largest user of blue water resources, responsible for about 80% of global freshwater withdrawals, primarily for irrigation (Poore and Nemecek, 2018). However, the vast majority of the water used in agricultural and livestock production is green water (Hoekstra and Mekonnen, 2012; Liu and Yang, 2010; Mekonnen and Hoekstra, 2012).

A number of approaches to and methods for assessing CWU in livestock and agricultural production have been developed, as illustrated in Chapter three. Water productivity and livestock water productivity studies largely emerged in the 2000s to highlight the efficiency of water use in agricultural production and contribute 'more crop per drop', that is, to produce more crops per unit of water (e.g. Giordano et al. 2006; Molden et al. 2007b; Peden et al. 2009). They have also proved useful in identifying technologies and interventions that improve livelihoods of smallholder farmers, without resulting negative effects on environmental health (Descheemaeker et al., 2010). Livestock water productivity studies often aggregated green and blue water into one measure which was then related to the monetary or quantitative benefits generated from the production system. They have been a successful way of incorporating the many benefits of livestock production systems, although difficulties emerge when assessing the economic value of some of the livestock benefits such as insurance and draught power (Chapter three). However, they do not clearly distinguish between the different types of water in use for livestock production and the green water use is not related to the land where it is consumed. In addition, even with the inclusion of the multiple benefits of livestock, water productivity studies remain a single-factor assessment that focuses on efficiency of water use per unit of production and other natural resources are not considered.

Generally, water productivity assessments are presented as a single aggregated figure in kg of produce per unit of water[4]. The first estimates attributed all water evapotranspired over pasture areas to livestock production, generating CWU figures of 100,000-200,000 litres/kg of beef (Pimentel et al., 2004; Pimentel et al., 1997), using rough estimates of pasture evapotranspiration. More recent studies have included a range of production systems from extensive to intensive

[4] For comparative reasons, the livestock water productivity estimates have been recalculated to water use in litres per kg of produce, which is the most common format of illustrating agricultural CWU figures.

production, and estimate livestock CWU between 10,000 and 40,000 litres/kg (Deutsch et al., 2010; Van Breugel et al., 2010).

The introduction of the water footprint (WF) concept (Hoekstra and Huynen, 2002), which is the inverse of water productivity, brought attention to the potentially large water use of livestock systems, particularly in beef production (Mekonnen and Hoekstra, 2012). The aim of the WF concept is to highlight water consumption and pollution along entire supply chains, since the organization, characteristics and function of a value chain greatly affects the final WF of a product (Hoekstra et al., 2011). In addition, WFs, as a visualization of the water that is embedded in consumer products, can contribute to a better understanding of global green and blue CWU and the effects of consumption and trade on CWU, ultimately resulting in better management of global freshwater resources (Hoekstra et al., 2011). Water footprints highlight different types of water use, i.e. green, blue and grey, the latter being a proxy for water pollution. Although WFs were originally presented as a single figure, more recent studies have presented individual WFs for green, blue and grey water resources (e.g. Mekonnen and Hoekstra 2010a; Mekonnen and Hoekstra 2012). They are described as volumetric estimates of CWU and water pollution and aim to decrease CWU by reducing local and global WFs of products or services. However, they do not aim to measure the specific local impact of that water use (Hoekstra et al., 2011). WFs for beef are in the proximity of water productivity figures, ranging between 10,000-20,000 l/kg.

The lack of connection between WFAs and local impacts was criticized, which resulted in a revised water footprint method being developed by the life cycle assessment (LCA) community. An LCA-based WF focuses on reducing the local environmental impact of CWU per unit of production and aims to contextualise purely volumetric CWU assessments and make them spatially relevant. LCA-based WF are often weighted against estimated water scarcity indices (e.g. Pfister et al. 2009) to illustrate the contribution of CWU to water stress or scarcity. They mainly consider blue water use and scarcity, as a result of the fact that blue water use can directly contribute to or cause severe environmental problems locally (Quinteiro et al., 2018), as well as the difficulty of relating the majority of green water use directly to causing human or environmental harm (Ridoutt et al., 2009). Thus, LCA-based WFs generally exclude the majority of the water used in livestock production systems, which makes LCA-based WFs significantly lower than traditional WFs, for example in the study by Ridoutt et al. (2012), beef estimates range from 12-217 litres of water equivalents[5] per kg of beef.

In an attempt to include green water in CWU assessments, and at the same time illustrate the related effect of CWU on ecosystem functions, Chapter two identifies the potential effect of green and blue CWU in beef systems on water-related ecosystem services in a defined region. The chapter finds that the potential impacts of CWU on ecosystem services are markedly different for different production systems. For example, improvement of natural grasslands, as occurs in mixed and intensive production systems, will increase the provisioning ecosystem service of primary production as grassland productivity increases. However, conversion of natural grasslands into

[5] 1 litre of water equivalents illustrate the burden on water systems from 1 litre of consumptive freshwater use at the global average water stress index (Pfister et al., 2009, Ridoutt and Pfister, 2010)

croplands induces changes in water cycling and reduces favourable soil formation and habitat formation.

In addition, the results of Chapter two identifies that the impacts on ecosystem services of each beef production system are not relative to the pattern in the volumetric measure of CWU. The most extensive beef production system studied in Chapter two had a significantly larger CWU than the most intensive beef production systems but generated the least negative impact on water-related ecosystem services, for example loss of erosion control, soil quality and water quality. It should also be noted that the extensive system had a higher CWU per kg of beef and lower beef productivity, and thus contributed less to the provisioning ecosystem services (provision of food), which is the very purpose of the production system.

To conclude, different methodological approaches generate different results from livestock CWU assessments. For example**,** if the aim is to reduce the overall CWU, comparing livestock products based on volumetric WFs is appropriate. In that case, pork and chicken are preferable to beef production. However, when compared to pasture-based beef production systems both pork and chicken generally consume more blue water than beef (Mekonnen and Hoekstra 2010a; 2012), which can directly contribute to local water scarcity (Ridoutt and Pfister, 2010). As showed in Chapter two, extensive beef production is also associated with fewer negative impacts on ecosystem services than more intensive beef production systems. In addition, as is illustrated throughout this thesis, there is a greater feed-food competition associated with more intensively produced beef production, which increases competition over green water resources in the landscape and will be further discussed in Section 4 below.

In contrast to comparing volumetric WFs with the aim of reducing the CWU associated with human activities, an LCA study focuses primarily on the blue water fraction of each livestock product, which indicates that beef production should be preferred to pork and chicken (e.g. Mekonnen and Hoekstra 2012; Ridoutt et al. 2012). If the LCA-based WF is weighted against the contribution to water scarcity, the fact that pork has the highest overall blue water use per kg, would not be significant as long as that blue CWU does not occur in a water scarce region. Blue water use, however, can result in trade-offs for other potential production systems that depend on blue water resources, since these can be withdrawn from one place and used elsewhere (Schyns et al., 2019). This indicates that the volumetric size of a blue WF can be of significance, even though the use of water does not directly cause local water stress.

In summary, existing CWU assessments generally aim to calculate either the impact of blue water use in terms of local water scarcity (e.g. Ridoutt and Pfister 2010) or consider CWU in terms of green and blue water volumes (e.g. Hoekstra et al., 2011). As illustrated above, these two different methodological approaches generate markedly different results. Our results highlight that for volumetric CWU assessments to provide useful information about the water use related to an agricultural product, they must be clearly related to a local context. However, that does not mean that the total green and blue CWU for agricultural products cannot provide meaningful information. Preferably, lines of thoughts from both approaches should be integrated to provide measures of CWU that consider the impact of water resources use from both a local and a more regional/global perspective. Before I introduce a framework on how to achieve this, I will first explain how current methods integrate green CWU in the landscape.

7.4 Integrating green consumptive water use with the functions of the landscape

Both the water footprint assessments (WFA) and the LCA-based WF are used to inform decision-making at the scale of production and potentially consumption, that is, the micro scale. For example, a product-focused assessment, such as the WFA of beef, will generate results that are relevant at the micro scale, and primarily aim to inform the decision-making of actors along the value chain, especially producers and consumers on how to minimise their WF (Lathuillière et al., 2018a). The same applies for water productivity assessments, where estimates of water use are compared to other products or a benchmark in order to identify the most effective way to produce agricultural products per unit of water (e.g. Giordano et al. 2017; Molden 2007b).

The water footprint sustainability assessment (WFSA) was introduced in an attempt to compare WFs to available global water resources that can be sustainably allocated to human purposes (Hoekstra et al., 2011). The method aims to contextualise traditional WFAs and connect them to water management and decisions at meso and macro scales (Lathuillière et al., 2018a), thereby putting the WF of a product in the context of water availability at the water basin, regional or even global scale (e.g. Mekonnen and Hoekstra 2016; Schyns et al. 2019). The WFSA has deliberately been defined as a sustainability assessment rather than an impact assessment to ensure the inclusion of issues beyond the local scale, as well as those impacts that are not immediately visible (Hoekstra et al., 2011). In other words, the WFSA stresses the importance of considering the sustainability of water footprints in a broader context not restricted only to areas that are identified as water stressed or water scarce. The argument is that if we can increase the efficiency of water use of a water demanding activity in areas where water is abundant, such an increase could contribute to that water demanding activity not having to take place in a water-scarce area (Hoekstra et al., 2011).

The WFSA method uses a similar approach to LCA-based WF with regard to the contextualisation of blue water resource use in the landscape (Hoekstra, 2016; Hoekstra et al., 2011; Hoekstra et al., 2012). For green water use, however, the methodological approaches to contextualisation between LCA-based WFs and the WF network are different. The LCA-based WF suggest considering green water as "net green water", to illustrate the net change in green water consumption in relation to natural vegetation (Pfister et al., 2017; Quinteiro et al., 2018). This method has been used to identify changes in regional green water flows, via evapotranspiration, that cause long-term changes in blue water availability (Milà i Canals et al., 2009; Núñez et al., 2013a; Ridoutt et al., 2010), changes in green water flows to the atmosphere that have effects on precipitation levels (Lathuillière et al., 2016b) and changes in the land use production system that cause an effect on terrestrial evapotranspiration and surface runoff (Quinteiro et al., 2015).

However, the WFA manual argues that to be a useful indicator of freshwater use WFs must consider total CWU volumes (Hoekstra et al., 2011). Thus, the WFSA instead suggests comparing green WFs with the green water considered to be sustainably available for human purposes. Sustainably available green water for human purposes is calculated as the total evapotranspiration in a defined area minus an estimate of the volume of green water required to maintain key ecosystem functions, such as biodiversity (Hoekstra et al., 2011; Schyns et al., 2019). The green water required to maintain key ecosystem services does not necessarily equal the green water currently in use for upholding such key ecosystem functions.

Recent publications by Schyns et al. (2019) and Lathuillière et al. (2018b), two of the three studies to have applied the WFSA methodology to green water use, identify regions where green CWU is near to or exceeds the amount of green water identified available for human purposes. Thus, these are areas that have limited or no potential to allocate more green water to the production of food, feed, fuel or fibre without exhausting green water resources that are deemed to be needed to maintain biodiversity and ecosystem functions according to predefined goals. However, this methodological approach does not address alternative uses of land and water resources for human purposes and how these might potentially affect the sustainability of green CWU; for example, if green water use in a specific region, allocated for feed production, results in less impact in terms of competition over scarce resources, or if it contributes directly to food production or to maintaining key ecosystem functions in that area.

In the context of sustainable use of water resources, there is an important distinction to be made between green and blue CWU. More blue water can be made available for competing uses by means of one user simply using less blue water. In the case of green water, however, two difficulties arise. First, estimating the required share of available green water that should be allocated to nature conservation in order to ensure a sustainable use of water resources is highly complex (Quinteiro et al., 2018; Schyns et al., 2015; Schyns et al., 2019). Second, the production of food, feed, fuel and fibre cannot use green water in a more effective way to ensure that there will immediately be more water available for nature conservation. In other words, water availability for maintaining biodiversity is directly related to land use (i.e., the land that is allocated for nature conservation) rather than to an efficient use of green water resources. In that sense, it is land that needs to be allocated differently in order to ensure that a larger share of water resources will be reserved for nature conservation. The emphasis of the WFSA remains on minimising WFs and their impact on water availability, rather than on the WFs of different activities, such as feed and food production, and how managing those activities can contribute to a sustainable use of water resources. Thus, in order to discuss the sustainable use of green water resources, the green CWU must be considered in regard to the land area over which the water has been consumed. This is the foundation of the framework that has been developed, which is described and discussed in the following section.

7.5 Framework for addressing the effects of CWU in a landscape

The general aim of a CWU assessment is to inform and to contribute to water resources being managed more sustainably across scales. In order to highlight the scarcity of green water resources, and the effects of green CWU in a landscape, it is important to properly identify the different users and uses of water within a defined area, and the extent to which they compete over the same resources over the same land. To put volumetric CWU assessments in the context of the landscape and enable them to be relevant to decision-making processes at multiple scales, the emphasis should not be on assessing the CWU in relation to sustainably available water resources in a basin for human appropriation, as suggested in the WFSA. Instead, the emphasis should be on considering the CWU associated with agricultural production in the wider context of the socio-ecological landscape where the production takes place, and the multitude of potential users of and uses for the water resources that are currently allocated to agriculture.

To this end, we have developed a methodological framework that compares CWU assessments for agricultural production with competitive uses in the landscape. The developed framework is based on integrating and harmonising previous methods to properly address scarcity of both green and blue water resources, as suggested by Lathuillière et al. (2018a). The methodological approach to contextualising CWU estimates, presented, applied and tested in this thesis, differs from the WFA, WFSA and LCA-based WFs. The point of departure our method takes is that the resources required for a unit of production is directly connected to the land use over which that (green) water is appropriated.

The framework was developed for livestock production systems and tested by comparing the competition over water resources between feed and food production in beef production systems in Uruguay and Brazil (see Chapters four and six). However, the framework can be applied to other agricultural production systems and should also consider other competitive uses in the landscape, such as the production of food, feed, fuel, fibre and the support of ecosystem functions (see Figure 1).

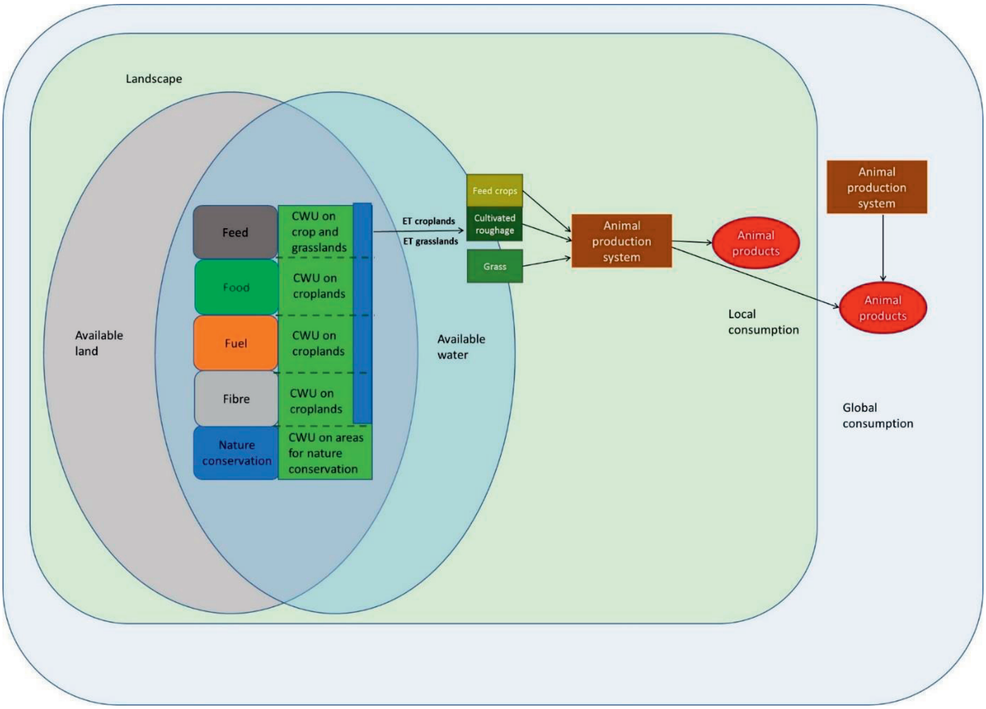

Figure 1: Conceptual framework of how to identify impacts of CWU assessments for feed, food, fuel, fibre and nature conservation by their competition over water and land resources in the landscape. Adapted from Ran et al. (2017).

To contextualize volumetric CWU assessments, the methodological approach illustrated in Figure 1 first quantifies and categorizes CWU into blue and green water. Second, in contrast to other methods, the CWU is categorized according to its land use, that is, crop or grasslands. By doing so, the CWU can be directly related to the associated competition over water resources between feed and food production. Assessing this competition over water resources in the landscape enables CWU assessments to help: i) to identify pathways to limit resource competition where resources are not available in excess; ii) define in what way water resources could be used most sustainably considering a specific goal; and iii) define potential alternative use of water resources that would contribute to a more sustainable use and management of water resources.

It has been proposed within the LCA network that green water use is an indicator of land use rather than water use and should not be included in CWU assessments (Ridoutt and Pfister, 2010). However, as illustrated in Chapter four, it is useful to assess green water as a part of CWU assessments since it can be both land and water that restrict agricultural production, and effective use of both green and blue water can enable large water savings in water restricted areas (Falkenmark and Lannerstad, 2010).

The approach that is presented in this thesis focuses on the competition over water resources in the landscape, which is another element of importance in assessing the sustainability of water use. This approach enables identification of the relative share, and spatial location, of the green and blue CWU of livestock products, for example beef production, that contribute to increased competition over water resources (Figure 1). In order to highlight the scarcity of green water resources, the land where particular green water resources are used must be properly identified and presented, coupled with CWU estimates, and categorized for each function, that is, the production of food, feed, fuel, fibre or whether it is required for ecosystem functions as outlined in Chapter four.

In Chapters four and six, we develop and test the proposed method, illustrating the limitations of water resource availability through a comparison of feed-food competition for water resources. We chose to compare the production of feed and food since the two types of agricultural production are easily comparable as they both contribute to the food system. For example, in Chapter four, we calculated a water use ratio (WUR), defined as the maximum amount of human digestible protein (HDP) derived from food crops from CWU appropriated to produce 1 kg of animal-source food (ASF) over the amount of HDP in that 1 kg of ASF. The study illustrates that it is possible to produce significantly more HDP from natural pasture-based beef production systems that do not require land with the potential to support crop production. More intensive beef production systems, however, which largely rely on crops for animal feed, contribute less HDP than if the same land and water were used to produce crops that are eaten by humans directly.

Chapter six studies CWU per kg of beef produced, and as the total CWU required for the production of beef in four different production systems in the Brazilian Cerrado. The results clearly indicate that total CWU and the efficiency of water use will increase for beef production in more intensified systems, including where feed crops constitute a larger share of the feed composition. This also results in a significant increase in competition over land, as cattle are fed with crops that directly compete for water resources with food production, potentially also competing over water resources with fuel and fibre production. The Cerrado has recently undergone a rapid agricultural transformation, in which native grasslands and forests have been cleared for planted crops and pasture lands (e.g. Arima et al. 2011; Beuchle et al. 2015), to increase productivity in the agriculture and livestock sectors. This example clearly illustrates that in addition to direct competition over land and water resources between food, fuel, feed and fibre, there is also increasing competition with maintaining the biodiversity in the region. Thus, CWU for beef production that takes place on a cropland that has replaced native grassland, or a planted grassland that has replaced tropical forest, can be argued to consume resources that are required for maintaining biodiversity.

7.6 Future development of the framework and harmonising of methods

The methodology presented in Chapter four has been developed and tested for beef production systems in Uruguay and Brazil. However, the method should be extended to integrate the full complexity of an increasingly global livestock sector, and also further extended to cover other functions in the landscape.

The study in Chapter five is complementary to the methodological approach developed for CWU estimates for livestock feed, described in Section 7.4. Chapter five contributes a novel methodological approach to assessing CWU for traded agricultural products, such as livestock feed crops. This approach enables CWU estimates of the production and consumption of traded crops to be spatially connected below the sub-national level, in contrast to existing methods that generally estimate agricultural trade on a national level. This novel methodology should be further integrated with the methodological approach developed for livestock CWU assessments in order to properly integrate livestock feed components that are imported into one region from another and acknowledge that the agricultural sector is becoming increasingly global (Galloway et al., 2007). Indeed, the links between consumers and producers need to be made more transparent to enable the identification of decisions that impact global food and feed supply chains in regard to CWU, as well as other natural resource use and greenhouse gas emissions that regulate the planetary boundaries (Rockström et al., 2009).

The comparison between feed and food production is not easily extended to other types of land uses, such as fuel and fibre production or land required for nature conservation, since these cannot be compared to human-edible foods. In order to capture whether there is local demand for agricultural land and the corresponding water to produce fuel or fibre, the approach outlined in this thesis should be accompanied by a measure for spatial allocation of land resources in a landscape. Land use planning is defined as a systematic assessment of the potential of land, and alternatives for optimal land uses in a social-ecological system. It is based on a participatory process that include a multitude of sectors and stakeholders and is scale dependent. The aim of a land use planning process is to support decisionmakers to sustainably manage land resources with regard to human purposes while safeguarding natural resources and the delivery of ecosystem services (Ziadat et al., 2017).

Land use planning is said to be beneficial for the future use of both land and water resources (Liu et al., 2007), and across scales (Nha, 2017). Integrating land use planning into CWU assessments would enable an illustration of local and regional competition over resources between food, feed, fuel, fibre and ecosystem functions. Successful land use planning should, of course, be extensive and include various stakeholders in a participatory setting. To be successful it must balance all elements of a social-ecological system to build resilience, ensure that land is sustainably allocated in regard to humans and nature, and contribute to this outcome (Brown and Fagerholm, 2015). Modelling and understanding how land use and the hydrological context respond to land use scenarios is important for optimal land use planning, management and policy (Liu et al., 2007), as well as the management of and policies on management of water resources. Land use planning can contribute to sustainable development as it contributes space for development, includes a commitment from a variety of stakeholders, and considers multiple scales of both short- and long-term development (Nha, 2017).

Even with the incorporation of land use planning, however, it is difficult to present results from CWU assessment in such a way that they can inform consumers how their consumption can contribute best to the sustainable use of water resources. Consumers 'best choice' will be highly locally dependent on the hydrological context in the area of production, and the local competition over land and green water resources. Thus, the best option may differ widely between locations. In addition, blue water use and contribution to blue water scarcity should be considered across scales. As CWU assessments of livestock and agricultural products must incorporate such complexity to be properly contextualized in the landscape, it is difficult to provide comprehensive results that can be of real use to consumers. However, the findings in this thesis suggest that if presentations of volumetric CWU assessments are, as a minimum, accompanied by their land use, and with regard to competing uses in a landscape and water scarcity, CWU figures could be made more useful to the informed consumer.

It should also be noted that the methodological approach outlined in this paper focuses on water use, and subsequent land use. By adding land use planning, aspects of nature and biodiversity conservation could also be included in the assessment. However, it would be beneficial to also consider other environmental factors associated with livestock production to ensure that the information and recommendations are well-grounded in a holistic view of the social-ecological system that is agriculture.

Since the mid-2000s, and the publication of *Livestock's Long Shadow* (Steinfeld et al., 2006), livestock systems have received a lot of attention for the environmental impacts associated with them. Beef production systems in particular are often associated with consumption of large amounts of water resources, as is illustrated throughout this thesis.

Despite their consumption of natural resources, it is widely recognized that animal products increase the nutritional value of diets (e.g. Mottet et al. 2017) and may therefore be an essential part of the diet in areas where people suffer from food and nutritional insecurity. However, the per capita intake of livestock products is significantly higher in developed regions than in low-income settings (FAOSTAT, 2018) and the global livestock sector is expected to continue the rapid growth that has been seen in recent decades (Godfray et al., 2018). This will further increase the pressure on the global food sector to produce more animal products.

The impacts of climate change are also expected to greatly affect the agricultural sector and increase insecurity in the global food system. For example, climate change is predicted to cause both more frequent droughts and excessive rain events in Latin America, which will severely affect water availability for agriculture (e.g. ECLAC, 2016; Filho et al., 2018). Schyns et al. (2019) estimate that for parts of the year Brazil already requires more green water than is sustainably available. This indicates the importance of managing water resources sustainably. In addition, Flachsbarth et al. (2015) have identified areas in Brazil, among others, where increased irrigation is already contributing to severe problems of blue water stress.

Thus, there is a demand for more sustainable management of water resources, including of both green and blue water. The traditional approach to increasing resource use efficiency has been to intensify agricultural practices. For livestock value chains this has largely meant a drive towards more crops being fed to cattle. However, as is demonstrated in this thesis, intensification may result in additional competition over resources, as well as a loss of other key ecosystem functions such as biodiversity. However, livestock production systems can also make use of low opportunity cost feed resources and in this way provide an important opportunity to increase global food security (van Zanten et al., 2018).

This thesis has identified that there are multiple pathways to decreasing the CWU of livestock products. The general approach is to increase productivity in livestock production systems, but this often results in increased feed-food competition over water resources, as illustrated in Chapter four and six. Looking at water use from a strictly volumetric perspective, another solution for reducing CWU is simply to produce and consume fewer livestock products in favour of plant-based foods, that generally require less water per kg. However, such an approach would ignore the potential contribution that livestock makes through its ability to produce food for human consumption from non-edible biomass.

Thus, a third potential pathway to reducing the impacts associated with CWU from livestock is to produce and consume an increased amount of livestock products reared using low opportunity cost feed, for example by favouring grass-fed beef production over intensive feedlot beef. This would, however, suggest that the average per capita consumption of livestock products would also have to be reduced. Thus, to conclude, there are sustainable ways to produce livestock products in regard to natural resource use and management. The problem (and the question) is primarily how much livestock products we can sustainably consume.

7.7 Main conclusions

The two objectives of this thesis are to improve understanding of the effects of CWU (i.e. blue and green) in a landscape, and to develop and apply a method to better assess such effects of CWU in livestock systems.

The main conclusions from this thesis are:

- Estimates of water use of livestock and livestock feed should distinguish between the different types of water, i.e. green and blue, and must be analysed in association with the local context to enable understanding of the impact in the landscape in which they are consumed;
- In order to relate green water consumption to the context of the landscape, green CWU should be categorised according to the land area and type of land use over which it is evapotranspired, e.g. over cropland or grassland. This allows identification of alternative uses and can support sustainable use of green water resources;
- From a food systems perspective, livestock fed on grasslands and other types of low opportunity cost feeds, use land and water resources more sustainably than livestock production systems that rely on land and water with high opportunity cost for food crop production;
- The former principles can be illustrated by the fact that different beef production systems result in distinctly different green and blue CWU. More extensive systems tend to have a higher volumetric CWU than intensive systems, while the latter contribute to higher competition of water resources between feed and food production, and require a larger blue water volume on average;
- To increase the policy relevance and better contribute to sustainability along global supply-chains, CWU assessment of traded agricultural products, such as livestock feed, need to be spatially explicit and consider trade flows at a sub-national scale;
- The framework presented in this thesis provides a basis for future studies to estimate CWU of livestock systems while accounting for feed-food competition, and the potential to include other functions and impact factors in the landscape. Such a framework is needed to capture the full complexity of the increasingly global food and livestock sector, and to ensure a sustainable management of water and land resources across scales.

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Appendices

Appendix A

Supporting information for the manuscript

Towards more spatially explicit assessment of virtual water flows: linking local water use and scarcity to global demand of Brazilian farming commodities

Rafaela Flach*, Ylva Ran, Javier Godar, Louise Karlberg, Clement Suavet

* To whom correspondence should be addressed. Email: rafaela.a.flach@gmail.com

A.1. Methodological approach

The analysis described in this paper includes a nation-wide assessment of water embedded in the trade of soy and sugar cane at a national and municipal resolution. A tiered approach is used, in which the role of international demand for water resources is analysed at a municipal scale and critical regions are identified (Table A.1).

A global water footprint accounting model from Mekonnen & Hoekstra (2011) was adapted from the period 1996-2005 to the period 2001-2011 to reflect changes in production and harvested area at the municipal scale SI. Thereafter, the SEI-PCS model was used to link global consumption with production at the municipal scale. Finally, the virtual water trade of soy and sugar cane were estimated by multiplying the estimated water footprint with the amount of soy and sugar respectively, in each municipality.

In order to estimate the impact of virtual water trade at the local scale, we use a set of high resolution data on water stress and scarcity (ANA, 2013). This data is thus used in the analysis to assess potential environmental impacts related to water of the sub-national water footprints.

Table A.1: Summary of the three steps combined for the Water Footprint Assessment carried out in this study.

	Water Footprint Accounting	**Material Flow Estimation**	**Water Stress Assessment**
Traditional analysis	Water footprint accounting for the period 1996-2005 [6]	Country-to-country flows [7]	--
This paper´s approach	Water footprint accounting adapted for the period 2001-2011	Spatially explicit flows [8]	Brazilian Water Agency data for estimating water stress

[6] (Mekonnen and Hoekstra 2011)
[7] (Kastner, 2011)
[8] (Godar et al., 2015)

A.2. Water footprint accounting

This study did not attempt to run one model applying climate, soil and crop data in Brazil for estimating water footprints, but instead it adapted global water footprint results from Mekonnen and Hoekstra (2011) to Brazilian crop footprints beyond the spatial and temporal resolutions of their study. Mekonnen and Hoekstra (2011) quantified the green, blue and grey water footprint of global crop production for the period 1996–2005, estimating the water footprint of 126 crops at a 5 by 5 arc minute grid; this model takes into account the daily soil water balance and climatic conditions for each grid cell. The results from this study are freely available and are widely used by researchers and practitioners worldwide; for example they have been previously applied for estimating Brazilian crop water footprints (Rocha & Studart 2013).

Water footprint flow accounting is sensitive to uncertainties related to precipitation, potential evapotranspiration, temperature, and crop calendar (Zhuo et al., 2014). As the footprints in Mekonnen and Hoekstra (2011) were estimated for the period between 1996 and 2005, not coinciding with the period of analysis chosen for this study, an analysis of the climatic changes between these periods was performed to establish if the climate differences between the two periods are significant, and where these changes are more pronounced. Reanalysis gridded climate data were obtained from CRU TS3.21 - Climatic Research Unit (CRU) Time-Series (TS) Version 3.21 of High Resolution Gridded Data of Month-by-month Variation in Climate (University of East Anglia Climatic Research Unit et al., 2013) – and analysed for the periods between 1995-2006 and 2001-2011.

The raster maps with the information on water footprints was provided in mm/y per grid cell. These values were first regionalized by municipality through a zonal statistic function in QGIS, and multiplied by the cultivated area per municipality, available in (IBGE 2015).

Besides the changes in climate, changes in the distribution of crop production in Brazil, the harvested area and consequently the yield were corrected. Equations (1) to (3) demonstrate how the water footprint of a certain municipality in 2011 can be corrected for changes in yield for soy production.

A

$$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = WF_{1996-2005}^{Soy}\left[\frac{m^3}{yr}\right] * \frac{Yield_{1996-2005}^{Soy}}{Yield_{2011}^{Soy}} \quad (1)$$

$$Yield = \frac{Production}{Harvested\ Area}\left[\frac{ton}{ha}\right] \quad (2)$$

$$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = WF_{1996-2005}^{Soy} * \frac{HA_{2011}}{HA_{1996-2005}} * \frac{Production_{1996-2005}^{Soy}}{Production_{2011}^{Soy}} \quad (3)’$$

Where WF is the water footprint in a municipality for a certain period, and HA is total municipal harvested area.

In this study, both changes in yield and harvested area were corrected from the period of the model simulation (1996-2005) to the study period (2001-2011). Equation (4) demonstrates the general methodology for correcting for changes in yield and harvested area.

$$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = WF_{1996-2005}^{Soy} * c * \left(1 + \frac{\Delta HA}{HA_{1996-2005}}\right)$$

$$c = \frac{HA_{2011}}{HA_{1996-2005}} * \frac{Production_{1996-2005}^{Soy}}{Production_{2011}^{Soy}} \qquad (4)$$

In terms of area, fives typologies of change in harvested area between the two periods can be distinguished (Table A.2). While most of the producing municipalities either increased or decreased the harvested area, some municipalities' production for a certain crop dropped to zero, and in a few municipalities where there was no harvested area for a certain crop between 1996 and 2005.

Table A.2: Calculation method for updating the water footprints, for each type of change in production between 1996-2005 and 2001-2011.

	Equation
Never Produced and Stopped Production	$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = 0$
Reduced Area and Increased Area	$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = WF_{1996-2005}^{Soy} * c * \left(1 + \frac{\Delta HA}{HA_{1996-2005}}\right)\ c = \frac{Yield_{1996-2005}^{Soy}}{Yield_{2011}^{Soy}}$
Started Production	$WF_{2011}^{Soy}\left[\frac{m^3}{yr}\right] = \left[WF_{1996-2005}^{Soy}\left[\frac{m^3}{yr}\right] * Yield_{1996-2005}^{Soy}\right]_{Neighbour} * \frac{1}{Yield_{2011}^{Soy}}$

For the municipalities for which no footprint was calculated in the 1996-2005 period, and fall in the category of the municipalities that started to produce the commodity between the two periods, the footprint was calculated based on a spatial interpolation of the water footprints in the neighbouring municipalities, and corrected for the yield in that municipality in the year of interest.

A.2.1 Uncertainties Due to Climate Variability

As previously mentioned, water footprint accounting is sensitive to uncertainties related to precipitation, potential evapotranspiration, and temperature (Zhuo et al., 2014). Adapting the results from (Mekonnen an Hoekstra, 2010b) required first the analysis of climatic changes between the two periods. Reanalysis gridded climate data for temperature and precipitation were obtained from University of East Anglia Climatic Research Unit, (2013) and analysed for the periods between 1995-2006 and 2001-2011.

Changes in the average precipitation and temperature for the two periods were calculated, and a t-student test with 95% of significance level was applied to verify the significance of these changes.

Figure A.1 shows the average temperature for the two periods (maps on the right) and the difference between the two averages (map on the left); the area with significant changes is highlighted with a dashed line.

Figure A.2 shows the average precipitation for the two periods (maps on the right) and the difference between the two averages (map on the left); the area with significant changes is highlighted with a dashed line.

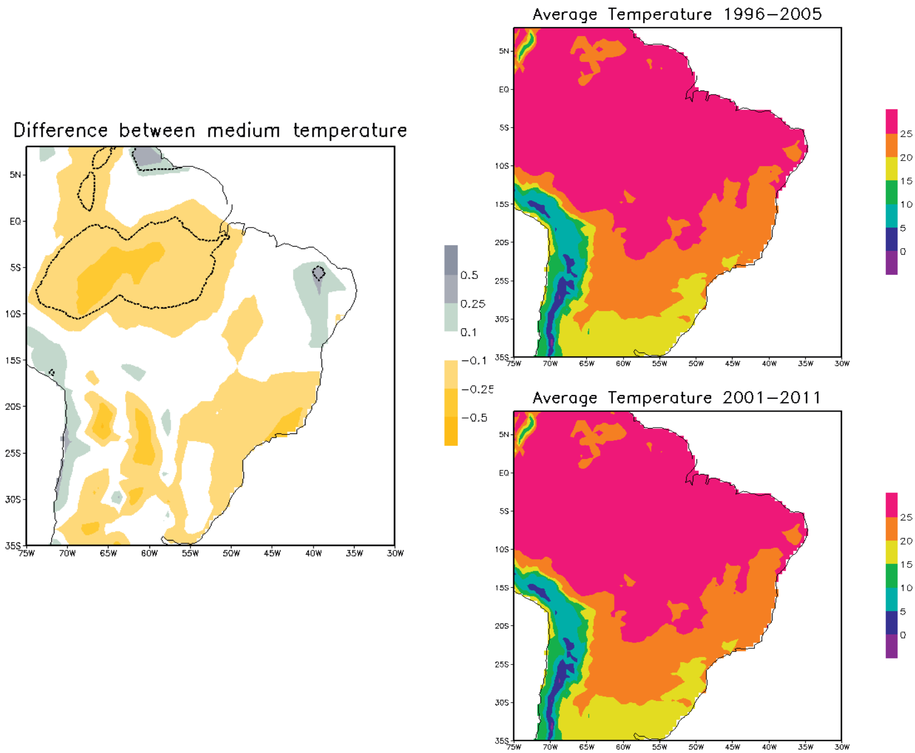

Figure A.1: Difference between the medium temperatures in the two periods (left, %) with significance level of 95% in t-student test (dashed line). Average temperature in the 1996--2005 period (above) and in the 2001-2011 period (below) (mm).

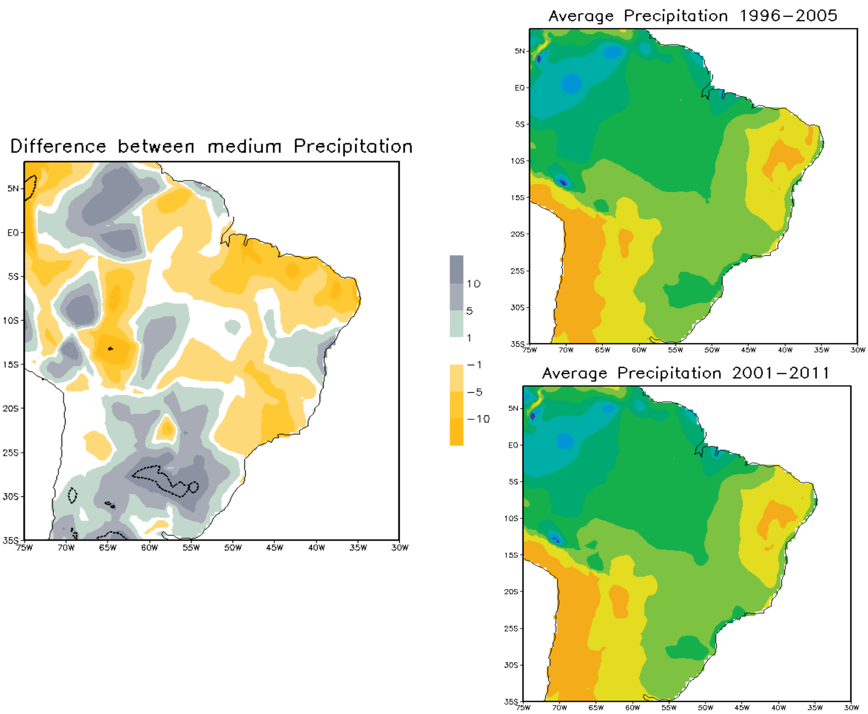

Figure A.2: Difference between the medium precipitations in the two periods (left, %) with significance level of 95% in t-student test (dashed line). Average temperature in the 1996-2005 period (above) and in the 2001-2011 period (below) (mm).

Even though by looking to the maps with the average temperature and precipitation for the two periods it is difficult to visualize the differences between the two periods, the maps with the difference between the averages demonstrate the regions with positive and negative changes throughout the country. In terms of temperature, the area with significant positive changes is located in the Amazon basin; this area is likely to have the footprints slightly underestimated for the period of 2001-2011. The changes in precipitation, on the other side, were not significant in most of the country apart from a small region in the south of the country.

A.3. Material trade flows

The methodology for modelling spatially explicit trade flows is described at length in Godar et al. (2015). Throughout this paper, soy and sugarcane equivalent are used, and include soybeans, soy cake, soy oil and soy sauce for the soybean crop, and sugar from sugarcane and ethanol for the sugarcane. The traded products defined by the Harmonized Commodity Description and Coding System from the World Customs Organization. Table A3 and Table A4 show the aggregated commodities, their FAO and NCM codes, and their respective conversion factors.

Table A.3: Soy NCM trade codes, corresponding FAO codes for traded commodities, calorific content and conversion factor applied to processed soy products to estimate the equivalent tons of soybeans. Obtained from FAO (2001) and FAO (2003).

NCM CODE	FAO CODE	FAO CLASSIFICATION	CONVERSION FACTOR[a]
12010010,12010090, 12011000,12019000	236	Soybean	1
15071000,15079011, 15079019,15121911,15079090	237	Soybean oil	2.639
12081000,23040010,23040090	238	Soybean cake	0.779
21031010,21031090	239	Soy sauce	0.167

[a] Calorific content vs. calorific content of soybean

Table A.4: Sugarcane NCM trade codes, corresponding FAO codes for traded commodities, calorific content and conversion factor applied to processed soy products to estimate the equivalent tons of soybeans. Obtained from FAO (2001) and FAO (2003).

NCM CODE	FAO CODE	FAO CLASSIFICATIO N	CONVERSION FACTOR[a]
17011100|17011400|17019100	156	Sugar Cane	1
17011400|17011100	162	Sugar Raw Centrifugal	7.6077
17019900	164	Sugar refined	7.6077
17011300	167	Sugar nes	7.6077
22071000|22071010|22071090|22072010|22072011|22071019	2207	Ethanol	15.95291

A.4. Water stress assessment

A typology of water criticality was projected based on an indicator of water stress, which made it possible to differentiate water footprints from regions with different degrees of water stress, and identify critical regions. First, the data used to produce these indicators are described, as well as its source and estimation method. Then, the methodology to calculate the three indicators will be described, and the matrix of typologies is demonstrated.

A.4.1 Available Data

The water availability and water demand data were obtained from the Brazilian Water Agency, and the population data was obtained from the National Institute of Geography and Statistics (ANA, 2013; IBGE, 2011). In 2013 the Brazilian Water Agency (ANA) published the Situation Analysis of Water Resources report, which evaluates the country's water resources in terms of availability, quality, multiple user demand, water conflict resolution and governance (ANA, 2013). After the publication of this report, this extensive database of water availability and demand estimated on the micro-basin scale for the entire country was made available. The finer scale data has the spatial resolution of level 12 in the Otto Pfapfstetter catchment coding system (Furnans and Olivera, 2001), which results in 168843 polygons with average and maximum area of 5071 and 371245 hectares, respectively.

The Brazilian Water Agency conceptualizes water demand as:

> *"Corresponds to the withdrawal flow, i.e., the water destined to meet diverse consumptive uses. Part of this claimed water is given back to the environment after use, which is denominated as return flow. (...) The non-return water, the consumptive flow, is calculated as the difference between the water withdraw and the return flow".* (Author's translation, ANA, 2013, p.87)

The water availability, on the other hand, is defined as the $Q_{95\%}$, i.e. the flow in cubic metres per second which was equalled or exceeded for 95% of the flow record, summed to the regularized flow, in case of existence of upstream dams. The water stress indicator estimated by the Brazilian Water Agency is estimated with the same method described by Smakhtin et al. (2004) for estimation of the Water Stress Index (WSI) without consideration of Environmental Water Requirements (EWR).

The indicators of water availability and water demand were obtained in the microbasin level, and were then regionalized to the municipality scale with the use of Geographical Information System analysis. The water stress indicator was calculated both for the municipal and microbasin scale.

For estimation of water stress, a use-to-availability indicator was calculated, by dividing the total water demand by the available water flow in the same area (ANA, 2013).

Table A.5 shows the thresholds for each class of water stress, based on Raskin et al. (1996).

Table A.5: Characterization of water stress use-to-availability ratio (Raskin et al., 1996; adapted from Perveen and James, 2011)

Percent withdrawal	Technical water stress
<10	Low water stress
10–40	Medium water stress
>40	High water stress

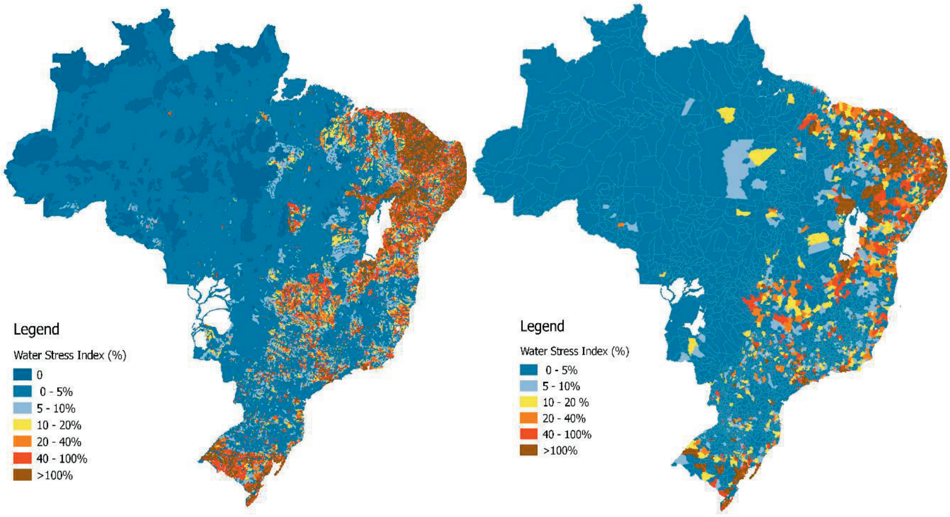

Figure A.3: Map of water stress (%) per microbasin (left) and per municipality (right)

Water stress was calculated throughout the country, at the micro-basin and municipality levels (Figure A.3). It can be seen that, although low levels of water stress are observed throughout most of the country, there is great variability. Although the water stress indicator outlines the relationship between demand and availability, it does not identify the causes of stress, which might be due to low availability, high demand, or both; it also does not identify which is the main use that determines high demand – industrial, urban, agricultural, etc. The Brazilian Water Agency differentiates, however, between three different main causes of stress, that can be identified in this map: low water availability in the north-eastern semi-arid, high irrigation demand for rice fields in the extreme south, and high urban demand in the main metropolitan regions, mainly in the southeast (ANA, 2013).

It can be observed that finer scales provide significantly more relevant information in terms of assessment of water stress, and the use of aggregate national and regional averages can mask local scarcity found in some cities and metropolitan areas. It can also be observed, when comparing basin-level and municipal indicators, that some regions with high water stress when analysed in basin scale are perceived to have less stress on the municipal scale; this happens as a result of the fact that, when regionalizing water availability throughout the municipality area, the flows from one or more water-abundant areas within the municipality are summed to the general municipal water availability. This implies that water can be transported from more abundant to scarce basins within the municipality to other more scarce areas, which might not be the reality.

Appendix B

Supplementary information for the manuscript

Consumptive water use for beef production in the Brazilian Cerrado: past and future trends

Ran, Y.[1,2], De Boer, I.J.M.[1], Lannerstad, M.[3], Van Middelaar, C.E.[1]

B.1. Methodological approach

B.1.1 Beef production systems

This study analyses four beef production systems in the Brazilian Cerrado. These are described in more detail in Tables B1-B2.

Table B1: Minimum and maximum weight of cattle in different categories of the cattle cycle for the four beef production systems: natural pasture (NP), improved pasture with legumes (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and feedlot (FL).

	NP		IP_{leg}		IP_{supp}		FL	
Stage of cattle cycle	Weight of animals (kg)	Number of animal units (LU) of each category	Weight of animals (kg)	Number of animal units (LU) of each category	Weight of animals (kg)	Number of animal units (LU) of each category	Weight of animals (kg)	Number of animal units (LU) of each category
	Min-max		Min-max		Min-max		Min-max	
	kg	LU	kg	LU	kg	LU	kg	LU
Bulls	650.0	23.1	650	23.1	650.0	23.1	650.0	23.1
Cows	430.0	382.2	430	382.2	430.0	382.2	430.0	382.2
Calves (f)	32-155.0	41.3	35.0-170	52.9	35.0-170	52.9	35.0-170	52.9
Calves (m)	32-170.0	45.3	35.0-185	57.6	35.0-185	57.6	35.0-185	57.6
Heifer	155-360.0	91.2	170.0-360	106.4	170.0-360	106.4	170.0-360	106.4
Steer	170-380.0	96.3	185.0-380	112.3	185.0-380	112.3	185.0-380	112.3
Finishing heifer	360-420.0	106.4	360.0-440	130.0	360.0-440	130.0	360.0-451.9	133.6

Finishing steer	380-470.6	119.2	380.0-480	142.1	380.0-480.8	142.1	380.0-490.7	145.0

Source: Adapted from Cardoso et al. (2016).

Table B2: Stocking rates, cropland and grassland requirement and slaughter rate for four beef production systems in the Cerrado: A natural pasture system (NP), an improved pasture with legumes (IP_{leg}), an improved pasture system with supplementary feeding (IP_{supp}) and a feedlot system (FL).

Production system	Stocking rate	Grassland	Cropland	Total area	Slaughter rate
	LU/ha	(ha)			%
NP	1.00	679.5	0.00	679.5	20.2
IP_{leg}	1.70	432.1	5.80	437.8	21.4
IP_{supp}	2.50	293.8	10.7	304.5	21.4
FL	2.75	267.1	27.2	294.3	21.4

Source: Adapted from Cardoso et al. (2016)

B.1.2 Consumptive water use calculations:

The water use estimates for feed crops and silage are calculated using a global pixel-based model that accounts for area-specific crop water requirements, both irrigated and rainfed, vegetation growth and yield per pixel at a resolution of 0.5°. The CWU is calculated as an average over a period of 10 years. CWU for crops is estimated as green and blue water use by accumulating daily evapotranspiration during the entire growing period and relating it to the yield to provide CWU estimates per ton of crop produced. The model uses yield data for each crop type: soybeans, maize and silage as well as additional input data on, for example, temperature, precipitation frequency, days with precipitation, hours of sunshine and soil texture.

Evapotranspiration over pasture was determined for each state using the following equation, Eq B1 (Zhang et al., 2001):

$$ETp = \frac{1 + 0{,}5\frac{1100}{P}}{1 + 0{,}5\frac{1100}{P} + \frac{P}{1100}} P$$

(Eq B1)

Where ET_p is pasture evapotranspiration and P is annual precipitation in mm y^{-1}. Pasture ET was estimated for each state using the FAOs CROPWAT model (FAO, 2010b, 2014) and the CLIMWAT database on precipitation, estimated as an average over a data minimum of 15 years. All the variables, apart from potential evapotranspiration are direct observations or conversions of observations (FAO, 2010a).

Finally, pasture ET per kg of DM is calculated by dividing ETp by an estimate of pasture productivity of 5.3 tons of dry matter per ha (Thiago and Silva, 2006). This estimate was based on a cattle density of 1.5 animal units per ha, which is slightly higher than the NP system. Thus, pasture productivity for the NP system is somewhat overestimated and will therefore generate a small underestimate of the CWU for natural pastures.

B.2 Specific results:

The NP system relies entirely on pasture for animal feed. However, the IP_{leg}, IP_{supp} and FL are also constitutes of maize and soybeans. To illustrate the relative share of the CWU on cropland for these three system, in relation to the total maize and soy production in the Cerrado states, we calculated how much of the total production of maize and soy in each Cerrado state, was required for feed in 2016 (illustrated in Table B3).

Table B3 illustrates that for the two improved pasture systems, the feed crop requirement for maize is generally less than 1% of total production. The requirement for soy exceeds 1% for the IP_{supp} and FL system only in the state of Pará. However, the feedlot system requires more than 1% of total maize production in two of the cerrado states, more than 2% in Mato Grosso do Sul, 6% in Goias and Bahia and between 30-50% inPará, Maranhão and Tocantins. These results highlight that the CWU of the feed crops required for beef production in these states is not insignificant. Table B4 illustrates the proportion of soy from each state that is exported outside of Brazil.

Table B3: Proportion of total production of maize and soy in the cerrado states required for three beef production systems in 2016: improved pasture with legumes (IP_{leg}), improved pasture with supplementary feeding (IP_{supp}) and feedlot system (FL).

Production system	IP_{leg}		IP_{supp}		FL	
Crop (%)	Soy	Maize	Soy	Maize	Soy	Maize
Mato Grosso	<1	<1	<1	<1	<1	1
Mato Grosso do Sul	<1	<1	<1	<1	<1	2
Minas Gerais	<1	<1	<1	<1	<1	<1
Goias	<1	<1	<1	<1	<1	6
Pará	<1	<1	1	1	<1	28
São Paolo	<1	<1	<1	<1	<1	1

Bahia	<1	<1	<1	<1	<1	6
Tocantins	<1	<1	<1	3	<1	47
Maranhão	<1	<1	<1	<1	<1	38
Piauí	<1	<1	<1	<1	<1	8

Table B4 Proportion of soy produced in each Cerrado state exported from Brazil

State	%
Mato Grosso	35
Mato Grosso do Sul	40
Minas Gerais	55
Goiás	53
Pará	<1
São Paulo	29
Bahia	71
Tocantins	96
Maranhão	75
Piauí	96
Distrito Federal	<1

Source: TRASE (2019).

B.3. Limitations of the study

The hydrological model that is used in the study for crop estimates of crop water requirements operates at a 0.5 spatial degree resolution. Ideally, to properly identify spatially explicit differences in grass and crop water requirements, water modelling should be optimized for a lower spatial resolution to deliver more precise national/sub-national results on crop and grass water requirements and spatial variability within the region.

Moreover, the crop and pasture water requirements would both be more accurate if they were based on more recent meteorological and model-input data. However, the results are only presented as relative to each other, and should not be interpreted as absolute measures. We therefore find it reasonable to use averages that have been calculated over time, and which are widely applied in other studies. In addition, climatic changes over time in Brazil have been studied for the period 2001-2011 in Flach et al. (2016) which found very minor changes that proved significant in regard to precipitation and temperature, and none situated in the region under study.

We assumed that all the feed required for production in each system was produced within each state. Data on the actual production location of all feed, and trade flows of feed between states, would enable an improved comparison of local efficiency of CWU. We have primarily identified effects at a regional level in this study and discuss them in terms of relative difference to identify areas of concern.

Acknowledgement

To arrive at the end of the chapter that is a PhD education is not a one woman job. During these past years I have been honoured to meet and work with a large number of people that I now extend my deepest gratitude to.

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Summary

One of the key challenges of today is to produce enough nutritious food for a growing and increasingly affluent global population, while avoiding unsustainable use of natural resources that results in a loss of key functions of our global social-ecological system. Water is one of these key natural resources. The expected rising demand for animal products is likely to amplify environmental impacts related to livestock production, including water use. This thesis primarily focusses on consumptive water use (CWU), water that is withdrawn from a watershed and not discharged in the same watershed, and distinguishes water into green and blue water resources. Green water is rainfall available as soil moisture for plant growth in the unsaturated zone, whereas blue water is water available as ground or surface water. Most studies that addressed water use in livestock production systems, however, do not address the local effects of both green and blue CWU in the landscape; this is essential to ensure a sustainable management of water resources across scales.

To address this knowledge gap, the two objectives of this thesis are to improve our understanding of the effects of CWU (i.e. blue and green) in a landscape, and to develop and apply a method to better assess such effects of CWU for livestock production systems. We define a landscape as an area with a multitude of functions and users that share the same land and water resources, such as production of food, feed, fuel, fibre and maintenance of biodiversity and ecosystem services.

Following the introduction, **Chapter 2** of this thesis addresses the first objective by emphasising that the impact of water use estimates for livestock should go beyond volumetric assessments. This key message is illustrated for three beef production systems in Uruguay; extensive, mixed and intensive. We explored impacts on water-related ecosystem services associated with each system. Results indicate that the most water effective beef production system is the one with the largest potential negative impact on water-related ecosystem services, such as erosion control, soil formation and water quality. Moreover, we identified potential trade-offs between efficiency of food production, water use efficiencies, and other water-related ecosystem services, such as soil formation, water quality and erosion control. These results highlight the importance of not increasing provisioning services at the expense of other key ecosystem services in the landscape, resulting in unwanted long-term side effects.

Chapter 3 presents a review of existing methods for CWU assessments of livestock production systems, and key areas for improvement. Methods are classified into three categories: water productivity assessments, water footprint assessments and life cycle assessments. Results show that the water productivity approach has been used to assess benefits of livestock production systems related to their CWU; the water footprint approach has raised awareness of the large amounts of water required for livestock production; whereas life cycle assessments highlighted the important connection between water resource use and local impacts. Key areas for improvement are: 1) both green and blue water resources should be included in assessments, and presented separately; 2) measures of water quality should not be summarized within quantitative assessments of water resource use; and 3) methods for assessing water use in livestock systems must consider the alternative uses and benefits of resource use in a specific location.

In response to the findings and recommendations of Chapter 3, **Chapter 4** presents a newly developed method to account for the freshwater use competition between food crop and animal

feed production to evaluate the CWU in livestock production systems. The developed water use ratio (WUR) is defined as the maximum amount of human digestible protein (HDP) derived from food crops over the amount of HDP in ASF, using the same CWU. The method assesses feed-food competition by determining the amount of HDP that could have been produced from food crops, using the same CWU currently used to produce ASF. The method enables identification of livestock production systems that contribute to global food supply without competing with food production for water resources. Three beef production systems in Uruguay were used to illustrate this novel method; a natural pasture system (NP-NP), a system where cattle are fed on seeded pastures (SP-SP) and a feedlot system where cattle are first kept on seeded pastures and finished in feedlots (SP-FL). Results show that the NP-NP system uses the largest amount of water per kg of beef output. However, the SP-SP and SP-FL systems can potentially produce more HDP by growing food crops than by producing beef. Based on the traditional measure for water productivity, that is, the quantity of CWU per kilo of beef produced, we would conclude that the NP-NP system is least efficient, whereas based on the WUR the NP-NP system is the only system producing HDP more efficiently than food crops.

Chapter 5 recognizes the importance of globalisation when it comes to CWU assessments and contributes a novel approach to estimate CWU for traded agricultural products. Traditional methods for CWU assessments of traded goods rely on international trade flows at the country to country resolution. However, the water requirement for a crop varies substantially between different regions within a country, and the effects of CWU are highly local. Therefore, to improve estimates of water use associated with trade, Chapter 5 presents a method that connects producers at a sub-national scale to consumers in a global food and livestock sector. We calculated virtual water flows from Brazilian municipalities to countries of consumption, by allocating high-resolution spatially explicit water footprints of sugarcane and soy production to international trade flows. Results show that this approach results in differences of virtual water use estimations of over 20% when compared to approaches that disregard spatial heterogeneity in sourcing patterns. This difference against methods using national resolution in trade flows is due to national heterogeneity in water resources and differential sourcing.

In **Chapter 6,** we further applied and developed the methodological approach outlined in Chapter 4. We explored potential pathways for beef production in the Brazilian Cerrado to use water resources in a sustainable way while accounting for feed-food production. To this end, we analyse trends in water use for beef production in the Brazilian Cerrado for the period 2010-2016, and subsequently estimated maximum potential beef production and associated water uses for four distinct production systems: one natural pasture system (NP), one improved pasture system with legumes (IP_{leg}), one improved pasture system with supplementary feeding (IP_{supp}) and one feedlot system (FL).

Results illustrate that water requirements are relatively similar across all systems. The NP system, however, requires the largest amount of water per kg of beef produced, while the FL system is the most water efficient. Analysing the maximum potential beef production on current pasture area in the Cerrado states shows that the FL system can contribute a significant increase in beef production, but also consumes a significant amount of water over cropland that would be suitable for producing more human edible protein from food crops. In contrast to all other three systems, the NP system does not consume any water over cropland and, thus, does not contribute to

increased competition over land and water resources with food production. Results show that there are multiple pathways to increase beef production without significantly increasing feed-food competition over land and water resources, and that low-opportunity cost feeds, such as pastures can effectively contribute to a sustainable development of the food sector in areas where resources are scarce.

The general discussion in **Chapter 7** further elaborates on the different aspects of CWU assessments of livestock products, and how they can be developed to capture the impacts of CWU for livestock production in a landscape. The discussion identifies the needs to better integrate different methodological approaches in order to properly address the impacts of water use and ensure that results of different CWU assessments are not contradictory, do not target different decision-makers, or result in recommendations at one scale that will impede sustainable use and management of water resources, or result in negative trade-offs, across a multitude of scales and users.

To conclude, this thesis demonstrates that estimates of water use in livestock value chains should distinguish between the different types of water, i.e. green and blue water. In addition, the water use should be considered in a local context in order to identify potential impacts of CWU in the landscape. To address the impacts resulting from green CWU, green water use should always be categorised according to the land area and land use where it is consumed, for example on crop or grasslands. This allows the identification of alternative uses and can contribute to more sustainable use of green water resources.

Samenvatting

De aankomende decennia staan we voor de uitdaging om op een duurzame manier in de stijgende vraag naar voedsel te voorzien, met minimaal gebruik van schaarse grondstoffen en hulpbronnen, zoals land en water. De verwachte toename in de productie van dierlijke producten zal echter gepaard gaan met een toenemende druk op het milieu en een toenemend gebruik van grondstoffen. Dit proefschrift richt zich op blauw en groen waterverbruik in de veehouderij, waarbij verbruik betekent dat het water niet terugkeert naar de bron waaruit het is onttrokken. Groen water refereert hierbij naar regenwater in de bovenlaag van de bodem, beschikbaar voor de groei van planten. Blauw water refereert naar grondwater of oppervlaktewater dat kan worden opgepompt om vervolgens te worden gebruikt als drinkwater of als irrigatiewater. De meeste studies die zich richten op waterverbruik in de veehouderij houden geen rekening met de lokale impact ten gevolge van dit waterverbruik, terwijl dit essentieel is om duurzaam waterverbruik te garanderen.

Om onze kennis op gebied van duurzaam waterverbruik te vergroten, heeft dit proefschrift het doel om duidelijkheid te scheppen wat de lokale gevolgen van groen en blauw waterverbruik in de veehouderij betreft, en om een model te ontwikkelen om dergelijke gevolgen in kaart te brengen. Om de lokale gevolgen te kwantificeren kan worden gekeken naar de consequenties voor alternatieve vormen van gebruik van land- en water, zoals het gebruik voor de productie van voedselgewassen, biobrandstof, kleding, of voor het behoud van natuurlijke ecosystemen (biodiversiteit).

Na de introductie (Hoofdstuk 1) behandelt **Hoofdstuk 2** de eerste doelstelling van dit proefschrift en laat zien dat het kwantificeren van waterverbruik in absolute termen onvoldoende inzicht geeft in de gevolgen van waterverbruik door de veehouderij. Voor drie verschillende rundvleesproductiesystemen in Uruguay, een extensief, gemixt, en intensief systeem, worden de gevolgen van waterverbruik op ecosysteemdiensten in kaart gebracht. De resultaten laten zien dat het systeem dat het minste water verbruikt in absolute termen, de grootste impact heeft op de ecosysteemdiensten die afhankelijk zijn van water, zoals het voorkomen van erosie, het behoud van een gezonde bodem en het behoud van waterkwaliteit. Ook wordt er inzicht gegeven in mogelijke negatieve wisselwerkingen tussen voedselproductie, waterverbruik, en water afhankelijke ecosysteemdiensten. De resultaten benadrukken het belang van het beperken van watergebruik voor voedselproductie wanneer dit ten kosten gaat van andere ecosysteemdiensten, daar dit op langere termijn ongewenste gevolgen kan hebben.

Hoofdstuk 3 geeft een overzicht van bestaande methoden om waterverbruik in de veehouderij te kwantificeren en geeft aanbevelingen voor het verbeteren van de huidige methoden. De methoden zijn ingedeeld in drie categorieën: methoden om waterproductiviteit te bepalen, methoden om de watervoetafdruk te bepalen en levenscyclusanalyse. Waterproductiviteit kwantificeert de opbrengst van de veehouderij per eenheid waterverbruik. De watervoetafdruk heeft bekendheid gegeven aan de hoeveelheid water die gebruikt worden voor de productie van dierlijke producten, terwijl de levenscyclusanalyse de relatie tussen waterverbruik en lokale impact benadrukt. De belangrijkste aanbevelingen zijn: 1) zowel blauw- als groenwaterverbruik dienen apart geanalyseerd en gepresenteerd te worden; 2) indicatoren voor waterkwaliteit en waterkwantiteit dienen gescheiden te blijven; 3) alternatief gebruik van water en potentiele voordelen van deze alternatieve vormen van gebruik dienen gekwantificeerd te worden voor de regio waar het waterverbruik plaatsvindt.

In navolging van Hoofdstuk 3, wordt er in **Hoofdstuk 4** een nieuwe methode gepresenteerd die rekening houdt met het feit dat water gebruikt voor de productie van diervoeders niet gebruikt kan worden voor de productie van voedselgewassen. De zogenoemde waterverbruiksratio (WVR) geeft aan hoeveel verteerbaar eiwit voor humane consumptie (VE_{HU}) er maximaal geproduceerd had kunnen worden uit voedselgewassen per eenheid VE_{HU} geproduceerd in dierlijk product, bij eenzelfde waterverbruik. De methode geeft dus inzicht in de competitie om water tussen voer- en voedselproductie. De methode biedt daarmee een manier om veehouderijsystemen te identificeren die een bijdrage leveren aan de wereldvoedselvoorziening zonder competitie om water met voedselproductie. Ook in dit hoofdstuk worden drie rundvleesproductiesystemen in Uruguay gebruikt om de methode te illustreren: rundvleesproductie op natuurlijk graslanden (NG-NG), rundvleesproductie op ingezaaide graslanden (IG-IG) en een feedlotsysteem waarbij het vee eerst op ingezaaide graslanden wordt gehouden en vervolgens in een feedlotsysteem wordt afgemest (IG-FL). De resultaten laten zien dat het NG-NG systeem het meeste water verbruikt per kg rundvlees, maar dat de twee andere systemen meer VE_{HU} hadden kunnen produceren wanneer het water gebruikt zou zijn voor de productie van voedselgewassen. Op basis van traditionele methoden (absoluut waterverbruik) zou geconcludeerd worden dat het NG-NG systeem het minst efficiënt is, terwijl de WVR laat zien dat het NG-NG systeem als enige meer VE_{HU} produceert dan voedselgewassen.

Hoofdstuk 5 erkent het belang van globalisering voor het bepalen van waterverbruik van agrarische producten en draagt een nieuwe methode aan voor producten die verhandeld worden. Traditionele methoden voor het bepalen van waterverbruik voor verhandelbare producten zijn gebaseerd op internationale handelsstromen tussen landen. Het waterverbruik voor gewasproductie kent echter grote regionale verschillen, ook binnen een land. Hoofdstuk 5 beschrijft een methode om producenten op een sub-nationaal niveau te verbinden met de wereldwijde voedselmarkt, om zo het inschatten van het waterverbruik gerelateerd aan handel te verbeteren. Dit hoofdstuk brengt de virtuele waterstromen gerelateerd aan de productie van suikerriet en soja in verschillende regio's in Brazilië tot aan de plek van consumptie in kaart, door lokale productiedata te combineren met internationale handelsstromen. De resultaten van dit hoofdstuk verschillen tot 20% van de resultaten gebaseerd op traditionele methoden (nationale statistieken), waarbij het verschil verklaard wordt door regionale verschillen in waterverbruik en productiemethoden.

In **Hoofdstuk 6** wordt de methode uit Hoofdstuk 4 verder toegepast. Mogelijke manieren om rundvleesproductie in de Cerrado (Brazilië) te verduurzamen door competitie om water met voedselproductie te voorkomen worden geëxploreerd. Ten eerste geeft het hoofdstuk inzicht in de trends aangaande waterverbruik door de rundvleessector in de betreffende regio voor de periode 2010-2016. Vervolgens geeft het het mogelijke productievolume en daaraan gerelateerde waterverbruik voor vier verschillende systemen: rundvleesproductie op natuurlijke graslanden (NG), rundvleesproductie op verbeterde graslanden (VG), rundvleesproductie op verbeterde graslanden met gebruik van voedingssupplementen (VG_{supp}) en rundvleesproductie in een feedlotsysteem (FL).

De resultaten laten zien dat het totale waterverbruik voor de vier systemen grotendeels gelijk is. Het NG systeem verbruikt echter het meeste water per kg rundvlees, terwijl het FL systeem het efficiëntst is. Met het huidige graslandareaal als limiterende factor heeft het FL systeem de meeste potentie om het productievolume in de Cerrado te vergroten en kan het een belangrijke bijdrage leveren aan de verwachte productiestijging in de regio. Dit systeem verbruikt echter ook grote hoeveelheden groen water op akkerbouwland, water dat gebruikt zou kunnen worden voor productie van voedselgewassen. In tegenstelling tot alle andere systemen, is het NG systeem wederom het enige dat geen water verbruikt op land geschikt voor de productie van voedselgewassen. Er worden verschillende manier getoond om het productievolume in de rundvleessector te vergroten zonder de competitie om water tussen voer- en voedselproductie te vergroten, bijvoorbeeld door het gebruik van marginale graslanden en het gebruik van bijproducten uit de voedselindustrie. Op deze manier kan de veehouderij een belangrijke bijdrage leveren aan voedselproductie in gebieden waar land en water schaars zijn.

De algemene discussie in **Hoofdstuk 7** gaat dieper in op de verschillende aspecten van waterverbruik in de veehouderij en op mogelijkheden om de lokale gevolgen van waterverbruik beter te meten. Dit hoofdstuk benadrukt het belang van een betere integratie van verschillende methoden om ervoor te zorgen, dat waterverbruik juist wordt ingeschat, dat resultaten elkaar niet tegenspreken, dat aanbevelingen op een bepaald niveau of gericht aan een bepaalde doelgroep niet in tegenspraak zijn met die op een ander niveau of gericht aan een andere doelgroep en dat er geen negatieve wisselwerking ontstaat met andere duurzaamheidsaspecten.

Dit proefschrift concludeert dat er bij het inschatten van waterverbruik in de veehouderij onderscheid gemaakt dient te worden tussen groen en blauw water en dat de lokale gevolgen van waterverbruik in ogenschouw genomen dienen te worden. Om de lokale gevolgen van groenwaterverbruik te bepalen, dient groen water te worden ingedeeld naar het type land waarop het verbruikt wordt, bijvoorbeeld op gras- of akkerbouwland. Hiermee kan inzicht worden verkregen in eventuele alternatieven vormen van gebruik en dit zal bijdragen aan een duurzaam gebruik van groen water.

About the author

Ylva Ran was born in Stockholm, Sweden in 1984. She completed her BSc studies in Biology and Environmental Sciences at Gothenburg University in 2010. She earned an MSc in Ecosystems, Resilience and Governance from Stockholm Resilience Centre, Stockholm University. Her master thesis was written as an internship at Stockholm Environment Institute, and explored water use in livestock production systems. After finalising her master studies, she commenced a PhD with the Animal Production Systems group at the Wageningen Institute for Animal Sciences in 2013 to further explore how to account for water use for livestock production and how to relate such water use to environmental impact in the landscape.

In 2012 she was employed as a Research Associate at Stockholm Environment Institute (SEI), an international research institute that aims to bridge policy and science. At SEI she continued carrying out her research on livestock systems and their natural resource use. She also engaged in other research projects focusing primarily on natural resource management in agricultural systems. The employment at SEI has continued where Ylva is currently employed as a Research Fellow. Her research primarily focus on agricultural systems, environmental impact assessment, natural resource management, participatory methods for measuring environmental impact and the design of robust development interventions for a sustainable intensification of smallholders.

Publication list

Peer-reviewed journal publications

Ran, Y., Deutsch, L., Lannerstad, L., Heinke, J. (2013). Rapidly intensified beef production in Uruguay: Impacts on water-related ecosystem services. *Aquatic Procedia 1*: 77-87

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Ran, Y., de Boer, I. J. M., Lannerstad, M., Herrero, M., van Middelaar, C. E. (2019). Consumptive water use for beef production in the Brazilian Cerrado: past and future trends.

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Education certificate

Completed training and supervision plan

The Basic Package (3 ECTS)

- WIAS Introduction Course (2014)
- WIAS Course on philosophy of science and/or ethics (2019)

Scientific Exposure (15 ECTS)

- 3d Noble Laureate Symposium on Global Sustainability (2011)
- World water week Stockholm (2012)
- Water for food, water for life - Global Conference. Nebraska (2013)
- Conference on Agricultural research towards sustainable development goals, Agri4D, Uppsala (2014)
- First international conference on global food security, Nordwijkerhout (2013)
- Second international conference on global food security, Ithaca (2015)
- The SLU Food Security symposium, Uppsala (2013)
- High level expert workshop on environmental assessment of dairy value chains. Stockholm (2013)
- EURO-AGRIWAT conference, Wageningen (2016)
- Stockholm Environment Institutes Science forum, Stockholm (2016)
- Hydrologidagarna, Uppsala (2016)

In-Depth Studies (10 ECTS)

- Forage evaluation in ruminant nutrition (2015)
- Environmental impact assessment of livestock systems (2015)
- Statistical programming in R, Gothenburg University (2013-2014)

Professional Skills Support Courses (5 ECTS)

- Course Techniques for Scientific Writing (2015-2016)

Research Skills Training (10 ECTS)

- Preparing own PhD research proposal (2013)
- GIS course (2012)

Total: 43.0 ECTS

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