Zentrum für Entwicklungsforschung (ZEF)

Doing more with less – Identifying opportunities for sustainable agricultural intensification with Life Cycle Assessment

Dissertation

zur

Erlangung des Grades
Doktorin der Agrarwissenschaften
(Dr. agr.)

der
Agrar-, Ernährungs- und Ingenieurwissenschaftlichen Fakultät

der
Rheinischen Friedrich-Wilhelms-Universität Bonn

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Frankfurt/Main, Deutschland

Bonn 2025

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Tag der mündlichen Prüfung: 24.06.2024

Angefertigt mit Genehmigung der Agrar-, Ernährungs- und Ingenieurwissenschaftlichen Fakultät der Universität Bonn

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Frankfurt, 14. März 2024

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I. ABSTRACT

Sustainable intensification of agriculture is the challenge of the coming decades. For example, the United Nations state that demand for food will rise by 60% until 2020. The challenge is to produce "more crop per drop" while lowering GHG emissions and halt biodiversity loss from cropland expansion and from agrochemical pollution. The overall goal of the dissertation is to contribute to the development of policies and scientific methodologies to support sustainable agricultural intensification. Core methodology for this contribution is the Life Cycle Assessment (LCA) methodology. LCA is a well-established method to assess the environmental impacts of products over their complete life cycle. The first policy that required a LCA for each product batch from the producer was the EU Renewable Energy Directive. Several calculation tools were developed by market players to help farmers and processing industry to compile these LCAs. However, it was not known if a producer would get the same result from each tool when entering the same production data. In chapter 2 we make a systematic comparison between the two main tools on the market. We show that results of both tools differed up to 50% for the same production pathway. This means that producers may be able to improve their GHG balance by choosing a different tool without any actual improvements in the process. These findings show a policy gap, a regulatory gap that needs to be addressed by policy makers in order to guarantee a level playing field on the market and to create an incentive to improve the GHG performance of crop production. The findings of this thesis are of high relevance for national authorities that oversee implementing the EU renewable energy directive and in general for policy makers that would like to make a policy of this kind (LCA-policy connection). Global LCA studies on impacts of agricultural production exist, however they often do not sufficiently take site-specific conditions into account. We look in the third chapter at individual producers in Mexico to assess if and how biofuels can be produced in Mexico without aggravating water scarcity, reducing greenhouse gas (GHG) emissions and avoiding indirect land-use changes. These are well established impact categories in LCA literature. Regarding blue water impacts LCA assessment is rather advanced. What is new is that we also take greenwater use into account to assess how better greenwater management could improve blue water impacts. The thesis shows how a mix of extensification of irrigated farming and intensification of rainfed farming can help to relieve water stress while increasing overall production. The main conclusion is that there are significant green water opportunities for farmers in many regions. Whereas methods exist to find hotspots of blue water use and scarcity there is a lack of methods to identify green water opportunities at a regional level. In chapter 4 we develop a method to estimate the efficiency of green water use in rainfed agriculture at regional level. The step-by-step methodology provided in this chapter is useful for companies, farmers and policy makers that want to decrease their water footprint. It could complement water footprint methods and regionalized life cycle assessments in order to join the two strategies to approach water scarcity – use less irrigation and make green water use more efficient. While some parts of the thesis, like the methodology to develop Greenwater Opportunity Maps is readily applicable to different regions and scales, more research is needed to make LCA usable at farm level. For the time-consuming collection of field and soil data for LCA inventories at field level, the development of remote sensing might bring faster and yet precise enough alternatives. Also, all studies made in this thesis have been assessed with current and historical climate data not considering climate change. A next step would be to replicate the studies based on projected climate conditions. For the use of the method in political decision making and development of strategic water plans, future climate conditions need to be considered to make the recommendations "climate-proof". With such further efforts the coverage of impacts related to water use in LCA would continue to steadily improve and help to detect options for sustainable agricultural intensification. This thesis represents a first step in this direction.

II. ZUSAMMENFASSUNG

MIT WENIGER MEHR ERREICHEN - IDENTIFIZIERUNG VON MÖGLICHKEITEN ZUR NACHHALTIGEN INTENSIVIERUNG DER LANDWIRTSCHAFT MIT HILFE VON ÖKOBILANZEN

Die nachhaltige Intensivierung der Landwirtschaft ist die Herausforderung der nächsten Jahrzehnte. Nach Angaben der Vereinten Nationen wird die Nachfrage nach Lebensmitteln bis 2020 um 60 % steigen. Die Herausforderung besteht darin, "mehr Ernte pro Tropfen" zu produzieren und gleichzeitig die Treibhausgasemissionen zu senken und den Verlust der biologischen Vielfalt durch Ausweitung der Anbauflächen und Verschmutzung durch Agrochemikalien aufzuhalten. Das übergeordnete Ziel der Dissertation ist es, einen Beitrag zur Entwicklung von Strategien und wissenschaftlichen Methoden zur Unterstützung einer nachhaltigen Intensivierung der Landwirtschaft zu leisten. Die wichtigste Methode für diesen Beitrag ist die Ökobilanz (LCA). Die Ökobilanz ist eine bewährte Methode zur Bewertung der Umweltauswirkungen von Produkten während ihres gesamten Lebenszyklus. Das erste Gesetz, das eine Ökobilanz für jede Produktcharge vom Hersteller verlangte, war die EU-Richtlinie zu erneuerbaren Energien. Mehrere Tools wurden von Marktteilnehmern entwickelt, um Landwirten und der verarbeitenden Industrie bei der Erstellung dieser Ökobilanzen zu helfen. Nicht bekannt war, ob ein Erzeuger mit den gleichen Produktionsdaten mit jedem Tool das gleiche Ergebnis erzielt. In Kapitel 2 führen wir einen systematischen Vergleich zwischen den beiden wichtigsten auf dem Markt befindlichen Tools durch. Wir zeigen, dass die Ergebnisse der beiden Tools für denselben Produktionspfad um bis zu 50 % voneinander abweichen. Das bedeutet, dass die Produzenten ihre Treibhausgasbilanz durch die Wahl eines anderen Tools verbessern können, ohne dass es zu tatsächlichen Verbesserungen im Prozess kommt. Diese Ergebnisse zeigen eine Gesetzeslücke, die von politischen Entscheidungsträgern geschlossen werden muss, um gleiche Wettbewerbsbedingungen auf dem Markt zu gewährleisten und einen Anreiz zur Verbesserung der Treibhausgasbilanz der landwirtschaftlichen Produktion zu schaffen. Die Ergebnisse dieser Arbeit sind von großer Bedeutung für die nationalen Behörden, die die Umsetzung der EU-Richtlinie über erneuerbare Energien beaufsichtigen, und generell für politische Entscheidungsträger, die entsprechende Gesetze erlassen möchten (LCA-Policy Connection). Es gibt LCA-Studien auf globaler Ebene zu den Auswirkungen der landwirtschaftlichen Produktion, die jedoch oft nicht ausreichend die standortspezifischen Bedingungen berücksichtigen. Im dritten Kapitel betrachten wir einzelne Produzenten in Mexiko, um herauszufinden, ob und wie Biokraftstoffe in Mexiko produziert werden können, ohne Wasserknappheit zu verschärfen, dabei Treibhausgasemissionen reduzieren und indirekte Landnutzungsänderungen zu vermeiden. Diese Wirkungskategorien sind in der LCA-Literatur gut etabliert. In Bezug auf die Auswirkungen auf das blaue Wasser ist die LCA-Bewertung schon recht weit fortgeschritten. Neu ist, dass wir auch die Nutzung von Grünwasser berücksichtigen, um zu bewerten, wie eine besseres Grünwasser-Management die Auswirkungen auf das blaue Wasser verbessern könnte. Die Arbeit zeigt, wie eine Mischung aus Extensivierung der bewässerten Landwirtschaft und Intensivierung des Regenfeldbaus dazu beitragen kann, den Wasserstress zu verringern und gleichzeitig die Gesamtproduktion zu steigern. Die wichtigste Schlussfolgerung ist, dass es für Landwirte in vielen Regionen beträchtliche Möglichkeiten zur Nutzung von grünem Wasser gibt. Während es Methoden gibt, um Hotspots für die Nutzung von blauem Wasser und für Wasserknappheit zu finden, fehlt es an Methoden, um Opportunitäten für Nutzung von grünem Wasser auf regionaler Ebene zu identifizieren. In Kapitel 4 entwickeln wir eine Methode zur Schätzung der Nutzungseffizienz von grünem Wasser im Regenfeldbau auf regionaler Ebene. Die in diesem Kapitel vorgestellte Schritt-für-Schritt-Methode ist nützlich für Unternehmen, Landwirte und politische Entscheidungsträger, die ihren Wasserfußabdruck

verringern wollen. Sie könnte Wasserfußabdruckmethoden und regionalisierte Ökobilanzen ergänzen, um die beiden Strategien zur Bewältigung von Wasserknappheit - weniger Bewässerung und effizientere Nutzung von grünem Wasser - zu verbinden. Während einige Teile der Arbeit, wie z.B. die Methodik zur Entwicklung von Greenwater Opportunity Maps, ohne weiteres auf verschiedene Regionen und Maßstäbe anwendbar sind, bedarf es weiterer Forschung, um Ökobilanzen auf Feldebene nutzbar zu machen. Für die zeitaufwändige Erhebung von Feld- und Bodendaten für Ökobilanzen auf Feldebene könnte die Entwicklung der Fernerkundung schnellere und dennoch ausreichend genaue Alternativen bieten. Außerdem wurden alle in dieser Arbeit durchgeführten Studien mit aktuellen und historischen Klimadaten ohne Berücksichtigung des Klimawandels durchgeführt. Ein nächster Schritt wäre, die Studien auf der Grundlage der prognostizierten Klimabedingungen zu wiederholen. Für den Einsatz der Methode bei der politischen Entscheidungsfindung und der Entwicklung von Wassernutzungsstrategien müssen zukünftige Klimabedingungen berücksichtigt werden, um die Empfehlungen "climate-proof" zu machen. Mit solchen weiteren Anstrengungen würde sich die Erfassung der Auswirkungen der Wassernutzung in Ökobilanzen stetig verbessern und helfen, Optionen für eine nachhaltige landwirtschaftliche Intensivierung aufzuzeigen. Die vorliegende Arbeit ist ein erster Schritt in diese Richtung.

To Emma and Paulina

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III. LIST OF ACRONYMS AND ABBREVIATIONS

EU RED	European Renewable Energy Directive
GHG	Greenhouse gas
GOP	Greenwater Opportunity
iLUC	Indirect Land Use Change
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
LIIB	Low Indirect Impact Biofuel
LUC	Land Use Change
PME	Biodiesel from oil palm
RME	Biodiesel from rapeseed
RSB	Roundtable on Sustainable Biofuels
Sc-EtOH	Ethanol from sugarcane
W-EtOH	Ethanol from wheat
ZEF	Center for Development Research

Sugarcane farmers in Mexico (above) with documentation of yields (below), 2009

1. INTRODUCTION

1.1 Problem Statement

Global population growth, changing consumption patterns and climate change is projected increase demand for food by 60% until 2050 (van Dijk et al. 2021). One strategy for attaining production gains is cropland expansion, the other one intensification. Because arable land is limited, most of the additional production will have to come from sustainable agricultural intensification (Alexandratos and Bruinsma 2012; Zabel et al. 2019). In addition, expansion of cropland is associated with a higher risk for biodiversity loss (Zabel et al. 2019). Sustainable intensification is the broad term for an approach to agriculture that increases food production from existing farmland without increasing the impact on the environment. Farmers produce more from the same area of land and use fewer inputs while producing greater yields therefore resource utilization and management is improved. The term is also used more broadly to consider social issues like individual empowerment in the intensification decision process and equitable distribution of food (Loos et al. 2014).

Agriculture is both cause and victim of climate change. Agriculture and forestry are responsible for 23% of global Greenhouse gas (GHG) emissions (IPCC 2020) and its dependence on water resources makes it vulnerable to changing climate patterns. Two main challenges to agricultural intensification of existing farmland is nutrient and water management (Mueller et al. 2012a) focussing on closing "yield gaps" in underperforming landscapes. At the same time there are large opportunities to reduce the environmental impact of agriculture by eliminating nutrient overuse (Mueller et al. 2012a). Most of the world's farmland is rainfed but a number of studies indicate an optimization potential in green water use (Rockström 2003; Molden et al. 2007; Rockström et al. 2009; Hoff et al. 2010). The International Water Management Institute estimates that improvements in rain-fed agriculture could contribute 75% of the yield increases needed to fulfil future needs for agricultural products compared to only 25% from expanding or improving the irrigated agriculture (Molden et al. 2007).

The challenges of agricultural intensification can be tackled from different scientific angles and efforts from society, market players and policy makers. The Life Cycle Assessment (LCA) methodology is interesting in that respect because it is widely used to compare options for production and identify the best production pathway with respect to sustainability aspects (International Organisation for Standardization. Geneva). Results of these are commonly used by market actors that want to improve their production pathways, for end-consumer communication by certification and labelling (e.g., ecolabel like "*Blauer Engel*") but also at intergovernmental level (Sonnemann et al. 2018). In Europe, for example, 159 policies and 167 communications that include LCA to promote Sustainable Consumption and Production have been issued between 1992 and 2020 (Sala et al. 2021). Most of them are in sectors like products, vehicles and waste and are linked to Ecolabels and Eco-design. Implementation in the agricultural sector is still at an early stage (Sala et al. 2021). One of the first to include the agricultural sector was the European Renewable Energy Directive (European Parliament and Council 2009a) which focusses on GHG emissions in agriculture and subsequent production processes. It incentivizes the production of biofuels, thus incentivizing additional crop production. To do so, it specifies that biofuels can only be accredited to the biofuel quota if they meet a certain GHG threshold in their production process. Producers had to calculate the GHG reduction of each product batch with an LCA. As LCA is time consuming and requires methodological know-how, several calculation tools were developed by market players to help farmers and processing Industry. These tools were often Excel tools or online tools in which market players entered their production data (i.e., input data like amount of fertilizer or electricity used and output data like amount of crop or plant oil produced) and the tool calculated the Carbon Footprint. By the time the tools were issued, it was not known if a producer would get the same value from each tool when he entered the same production data.

Bioenergy policies are interesting because they allow to introduce more ambitious sustainability requirements than food production. In contrast to crop production for food, crop production for bioenergy would not take place without political support.

Therefore, they provide an opportunity to introduce more strict sustainability requirements. However, lessons can be drawn for potential future agricultural policies which may have a stronger focus on GHG. A meta-study on the evolution of LCA concluded that there are in general still issues to be solved in the interface between science and policy making (Sonnemann et al. 2018). For the EU RED, there is no study yet on the stringency of the LCA requirements and its consequences for their verification and hence its effectiveness in guiding towards sustainable agricultural production.

Changes to management practices to close yield gaps via sustainable intensification vary considerably by region and existing production intensity (Mueller et al. 2012b). With respect to water, global modelling studies show that many countries currently rated as "severely water short" would be able to produce enough food if green water were managed well (Iraldo et al. 2020). For example, in Mexico water shortage is considered severe and many cropping systems are underperforming (Rockström 2003a). LCA/Carbon/water/land use footprint for agricultural products do exist at a generic level and for selected countries and crops (Maciel et al. 2015) but not for specific crops in Mexico. There is a lack site-specific studies.

Water management is crucial for sustainable agricultural intensification (Mueller et al. 2012b). This includes both the management of irrigated cropping systems as well as rainfed systems or rainfed systems with supplemental irrigation. There are several methodologies that assess the impact of water use. The so called "water footprint" distinguishes between blue water (surface and groundwater), green water (rainwater stored as soil moisture), and grey water consumption (volume of freshwater required to dilute pollutants to applicable water quality standards) (Hoekstra et al. 2011).
Volumetric methods as proposed by Hoekstra simply sum up the volume of water used (Hoekstra et al. 2011). In contrast to GHG emissions, impacts of water consumption depend on spatial factors such as freshwater availability and use patterns at the specific location (1L consumed in Mexico has a different impact than 1L consumed in Sweden). Therefore, water-scarcity footprints multiply the water volume with a regional

characterization factor based on local water scarcity. Some methods assess local water scarcity by using consumption to availability ratios for each watershed, on yearly or monthly temporal resolution (Weinzettel and Pfister 2019) or providing regional characterisation factors for water scarcity footprints such as the AWARE method recommended by UNEP SETAC (Rockström et al. 2009a). Other LCA methods model impacts on endpoint indicators like human health and ecosystem quality (Bayart et al. 2010; Kounina et al. 2013; Berger et al. 2014; Pfister and Bayer 2014; Motoshita et al. 2018). Also, LCA frameworks to include potential long-term impacts on freshwater from land use by altering runoff, infiltration and erosion processes have been suggested (Pradinaud et al. 2019). These methods are useful to detect impacts of blue and grey water consumption. While current LCA studies highlight mostly hotspots of blue water use and grey water (through impact categories like toxicity and eutrophication), there is a lack of studies that investigate how greenwater use can be optimized.

Life cycle inventory (LCI) is the methodological step that involves creating an inventory of input and output flows for a product system. Such flows include inputs of water, energy, and raw materials, and releases to air, land, and water. The inventory can be based on literature analysis, field data or on simulations. For agricultural processes, a general challenge in LCA is to capture the diversity in cropping systems (Cucurachi et al. 2019). For example, comparative LCAs from crops from organic and conventional farming often do not differentiate the characteristics of the farming systems (Meier et al. 2015). For building water LCI, the FAO CROPWAT model has been used for building the LCI in most of the water LCA studies (Smith 1992). Up to date no LCA studies have been published based on the more recently published FAO AquaCrop Tool ((AquaCrop publications)(Mejias and Piraux 2017) even though it has been recommended to build water-flow-inventories (Payen et al. 2018) because it allows a more detailed analysis of green water flows than CROPWAT.

1.2 Goals

The overall goal of the dissertation is to contribute to the development of policies and scientific methodologies to support sustainable agricultural intensification. Core methodology for this contribution is the LCA methodology. Three key research goals are formulated based on the research gaps mentioned before.

1) Give recommendations to policy makers and national authorities that are in charge of implementing the EU renewable energy directive. How can the LCA requirement be improved to reach its aim to incentivize sustainable agricultural production? And more general: derive recommendations to policy makers planning to integrate LCA as a requirement in their policies on sustainable agriculture.

2) Explore opportunities for sustainable (in terms of GHG, water and land use) intensification at the level of individual producers using the LCA approach. How can green water management contribute to sustainable intensification?

3) Develop a method to find green water opportunities for sustainable agricultural intensification at a regional level to complement LCA blue water approaches.

1.3 Structure and approach of thesis

To produce more crop per drop, with less GHG emissions and without expanding agricultural land is the challenge of the coming decades. This challenge can be tackled from different scientific angles and efforts from society, market players and policy makers. In this thesis I chose the LCA method to contribute to this question. The thesis consists of two published papers (Hennecke et al. 2013, 2016) and one submitted paper.

In the second chapter we look at a policy that requires an LCA and aims at ensuring sustainable production.

The first policy in the agricultural sector that required a LCA for each product batch from the producer was the Renewable Energy Directive. In chapter 1, we did a systematic comparison of the two public and free available tools for producers to do such an LCA. One was the tool the Roundtable on Sustainable Biofuels and the other one from BioGrace project, an EU funded initiative. We did so by calculating GHG emissions from four production pathways: ethanol from wheat, ethanol from sugarcane, biodiesel from rapeseed and biodiesel from palm oil. In addition, three land use change (LUC) scenarios were calculated: for expansion of the biofuel cultivation area to grassland and forest and for improvement of agricultural practices. Both tools follow the methodology of the European Renewable Energy Directive and the same input data along the production chain was used. Then, we analysed the discrepancies and their cause to detect potential regulatory gaps and derive recommendations to improve the EU RED.

As the Renewable Energy Directive is a bioenergy policy that requires increasing quotas of biofuel blending in the European Fuel mix, it is a policy that will possibly lead to expansion of agricultural production. **In the third chapter** we look at individual producers with the LCA methodology to assess how a sustainable agricultural intensification could be achieved. We conduct a practical test of one of the tools analysed in chapter 1 with individual producers in Mexico. Environmental sustainability has many aspects. Therefore, we widen the scope of the analysis to include not only GHG but also land use and water scarcity. All of them are well established impact categories in LCA literature. Blue water impact assessment is rather advanced. What is new is that we assess how better greenwater management could improve blue water impacts. Also, we established new site-specific carbon/water/land use footprint for agricultural products for maize in Mexico. New was also that we used the detailed FAO AquaCrop Tool in an LCA. While several studies on the water impacts of agricultural products existed, they all used the former FAO tool.

The study in chapter 3 demonstrates the greenwater optimization potential on the level of individual producers. But how can the greenwater optimization potential be detected at a more global level? For policy makers and market players in water scarce regions it is interesting to know these opportunities also at a more regional level. Therefore, in **the fourth** chapter we widen the focus from individual producers to national level to identify regions of opportunities where unused green water is available for increasing agricultural production without negative consequences for other users. A step-by-step methodology is developed to look into blue and green water efficiency in several cultivation systems in Mexico and derives opportunities for using more rainwater and less blue water in water scarce regions for sustainable agricultural intensification. This complements existing maps where hotspots of blue water use are located by showing where water scarcity could be tackled by greenwater management (join the two approaches – use less blue water, use more green water).

2. LCA AS A TOOL IN POLICIES

Biofuel greenhouse gas calculations under the European Renewable Energy Directive – A comparison of the BioGrace tool vs. the tool of the Roundtable on Sustainable Biofuels [1]

2.1 Abstract

The European Renewable Energy Directive (EU RED) requires biofuels to reduce greenhouse gas emissions (GHG) by 35% compared to fossil fuels in order to count towards mandatory biofuel quota or to be eligible for financial support schemes. This reduction target will rise to 50% in 2017. For biofuel producers this implies that they want or need to calculate their emissions. The purpose of this paper is to compare two calculation tools for economic operators that are on their way to the market: the ‘‘BioGrace tool’’ and the ’’Roundtable on Sustainable Biofuels (RSB) GHG tool’’ for GHG calculations under the Renewable Energy Directive (both of which are freely available). Greenhouse gas emissions from four production pathways were calculated: ethanol from wheat, ethanol from sugarcane, biodiesel from rapeseed and biodiesel from palm oil. In addition, three land use change (LUC) scenarios were calculated: for expansion of the biofuel cultivation area to grassland and to forest (10–30% canopy cover) and for improvement of agricultural practices. Both tools follow the methodology of the European Renewable Energy Directive and the same input data along the production chain was used. Despite this, the results were significantly different. GHG emissions of the pathway ethanol from wheat were 21% lower when calculated with the BioGrace tool than with the RSB GHG tool. Differences were most pronounced in the cultivation phase with 20% deviation between the tools for biodiesel from palm oil and 35% deviation for ethanol from wheat and sugarcane. In practice this means that an economic operator can enhance the GHG performance of his biofuel by 20–35% by using a different calculation tool without improving the production process. We identified the use of different standard values in the two tools, in particular for the production of N-

[1] Article published in Applied Energy Volume 102, February 2013, Pages 55-62. Anna M. Hennecke, Mireille Faist, Jürgen Reinhard, Victoria Junquera, John Neeft, Horst Fehrenbach

fertilisers, for chemicals and electricity and one methodological choice regarding the calculation of field N2O emissions as source of these differences. This methodological point is not specified in the Renewable Energy Directive, giving economic operators and tool developers free choice. GHG emissions from land use changes varied by 14% to 49% due to differences in carbon stock data, methodological differences in allocation and a lack of precise land use type definitions.

We conclude from the results that there is a need for a deep harmonisation in the calculation process that goes beyond the methodological framework set up in current legislation. These findings are relevant because they show a policy gap, a regulatory gap that needs to be addressed by policy makers to guarantee a level playing field on the market and to create an incentive to improve the GHG performance of biofuel production.

2.2 Introduction

In 2009 the European Union set sustainability criteria for biofuels with the legislation of the Renewable Energy Directive (EU RED) and the Fuel Quality Directive (FQD). Greenhouse gas (GHG) emission savings from biofuels must be at least 35% compared to fossil fuels; this figure rises to at least 50% by 2017, and 60% by 2018 for biofuels produced in installations in which production started in 2017 or later (European Parliament and Council 2009a, b).

The directives give default values for GHG emission savings of 22 biofuel production pathways (Annex V of the RED or Annex IV of the FQD). For economic operators who want or need to do the calculations themselves a formula is given; the formula states that total GHG emissions are the sum of emissions from cultivation, processing and transportation of the biofuels.

Yet the directives specify neither the “standard conversion values” nor a number of details in the methodology. “Standard conversion values” are emission factors, lower heating values and other background data that are required to convert “input numbers” (e.g. the amount of fertilisers applied in cultivation) into GHG emissions. Standard values are for instance the emissions of carbon dioxide equivalent per kilogramme of nitrogen fertiliser or per megajoule of natural gas. As neither the RED nor the FQD fix these standard conversion values, certification schemes can freely choose standard conversion values in their tools. This will lead to differences in results among economic operators depending on the tool they use, even when calculations have been made for the same shipment of biofuel. Economic operators will choose the tool that gives the most favourable result.

A number of calculation tools are currently being developed for economic operators. This paper compares two of them – the BioGrace tool and the tool of the Roundtable on Sustainable Biofuels (RSB) for calculations under the EU RED which is referred to as "RSB GHG Tool" in the following.

The aim of this paper is to quantify the differences in the GHG results between tools that are on their way to the market and increasingly being used by economic operators, and to evaluate the source of discrepancies. To do so four typical biofuel production pathways were calculated in parallel with each tool.

The paper does not aim at evaluating and comparing different biofuel pathways. The GHG figures presented in this paper should not be used for this purpose as they exclusively cover those emissions that are covered by the RSB and BioGrace tools and therefore do not cover other emissions that might contribute to the overall GHG result, like emissions from indirect land use changes.

The BioGrace Project

The project BioGrace (BioGrace 2012) is funded by the Intelligent Energy Europe Program, an EU funding program. The consortium is composed of eight European project partners (section 8) that are close to the national government and the implementation process of the Renewable Energy Directive and experts in GHG calculations. The project aim is to harmonise calculations of biofuel GHG emissions and thus to support the implementation of the two directives into national laws. To this end, three sub-objectives were pursued:

1. Make biofuel GHG calculations transparent, by providing a BioGrace Excel GHG calculation tool reproducing the calculation of the GHG emission default values of the 22 biofuel production pathways listed in the RED Annex V and enabling economic operators to do actual calculations in a transparent manner according to RED methodology.

2. Publish a list of standard values. The list of standard values contains the conversion factors and heating values that were used for calculating the default values in the RED Annex V and FQD Annex IV.

3. Seek harmonisation of GHG calculations. Firstly, this is done by spreading the previous results to stakeholders and policy makers through a website, meetings, and a series of workshops, enabling them a) to see how calculation were made or b) to use this harmonised tool for their actual (own) calculations. Secondly, this is done by harmonizing biofuel GHG calculators. Currently user-friendly GHG calculators are being developed in Germany, the Netherlands, Spain, and the United Kingdom – in close co-operation with the project BioGrace. Once these calculators are finalised, BioGrace aims to harmonise these calculators to use the same standard values and produce the same results when the same input data is used.

Calculation tools are also being developed by economic operators (e.g. Abengoa) and multi-stakeholder initiatives like the Roundtable on Sustainable Biofuels that are not directly connected to BioGrace. The BioGrace project is in close contact with the Roundtable of Sustainable Biofuels and this paper is the result of this collaboration.

<u>The Roundtable on sustainable biofuels</u>

The Roundtable on Sustainable Biofuels is an international initiative coordinated by the Energy Center at the École Polytechnique Fédérale de Lausanne (EPFL) in Lausanne that brings together farmers, industry, non-governmental organizations, experts, governments, and inter-governmental agencies concerned with ensuring the sustainability of biofuels production and processing (RSB). Participation in the RSB is open to any organization working in a field relevant to biofuels sustainability.

The RSB has developed a third-party certification system for biofuels sustainability standards, encompassing environmental, social, and economic Principles & Criteria through an open, transparent, and multi-stakeholder process. The RSB GHG Tool allows the user to calculate lifecycle GHG emissions of biofuels using various methodologies, including the RSB methodology, the EU RED methodology, and the Swiss Ordinance on Proof of the Positive Aggregate Environmental Impact of Fuels from Renewable Feedstocks. The Tool also includes default pathway values under the EU RED, the U.S. Renewable Fuel Standard (RFS), and the California Low Carbon Fuel Standard (LCFS).

The RSB has also developed an RSB EU RED Standard in compliance with the EU RED. The RSB EU RED Standard received the recognition of the European Commission on July 19, 2011. The calculation of actual GHG values according to the methodology as laid down in the EU RED and the RSB EU RED Standard can be performed with the RSB GHG Tool, which was used in this paper.

2.3 Materials and Methods

Methodology for GHG emission calculations

Both tools calculate GHG emissions according to the methodological framework set out in Annex V of the EU RED and Annex IV of the FQD (European Parliament and Council 2009b, a).

System boundaries include five production steps: direct land use change (el), crop production (eec), transports (etd), industrial processing (ep) and end use (eu). In addition, four optional production steps that reduce emissions can be included: emission saving from soil carbon accumulation via improved agricultural management (esca), emission saving from carbon capture and geological storage (eccs); emission saving from carbon capture and replacement (eccr); and emission saving from excess electricity from cogeneration (eee) (EU RED Annex V point C.1).

$$E = eec + el + ep + etd + eu - esca - eccs - eccr - eee$$

In contrast to the simplified formula of Annex V, the production pathway, in practice, does not consist of four production blocks that follow linearly but each block is split up into many sub-processes that are scattered over the production chain. For example: ep includes the sub-processes oil mill (ep1), refinery (ep2) and biodiesel plant (ep3). In between, sub-steps of etd take place: Before ep1, a crop transport takes place from the field to the oil mill (etd1), before ep2 the oil is transported to the refinery etc.

Both tools implement this structure in the same way with a modular structure where each sub-process is calculated separately (Figure 1.1). In the RSB GHG Tool, the economic operator calculates the emissions of his operation by adding several modules, e.g. el + eec = Cultivation module; etd1 + ep1 = Processing module (Figure 1.1).

By-product allocation, definition of functional unit and definition of wastes and residues is conducted in both tools as specified in EU RED Annex V.

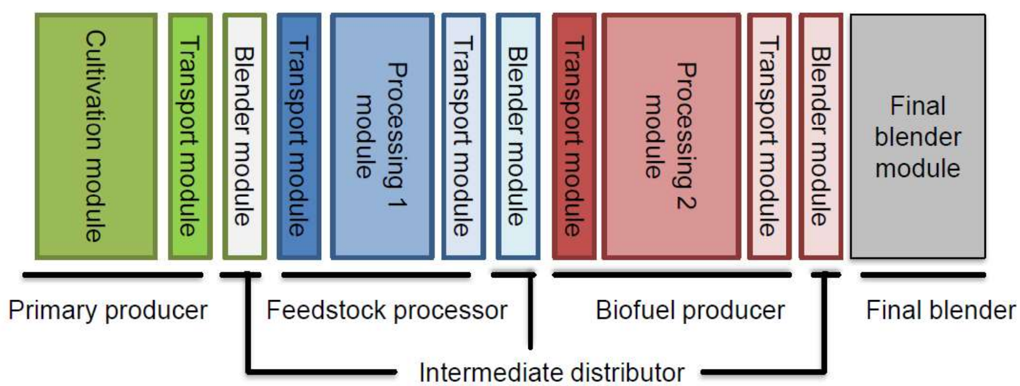

Figure 1.1. Steps of biofuel production that are considered in calculations in the RSB GHG Tool.

Production pathways and input data

Four typical biofuel production pathways were selected for comparison in the RSB-RED and BioGrace Tool:

1. Ethanol from sugarcane (Sc-EtOH)
2. Ethanol from wheat (W-EtOH) with Natural Gas boiler in the processing step
3. Biodiesel from rapeseed (RME)
4. Biodiesel from oil palm (PME)

These pathways were selected because they constitute a major share of biofuels currently available on the market and because they are on the list of production pathways in Annex V (European Parliament and Council 2009a).

Input data was taken from BioGrace tool public version 3 (BioGrace 2013). This includes all input data like energy and fertiliser inputs in wheat, rapeseed and oil palm cultivation, transport distances and vehicle types as well as inputs and outputs in the industrial phase. These input data originally stem from the JRC (Joint Research Centre of the European Commission) database and had been used to calculate the Annex V default values for the Commission.

Units of the input data were adjusted for the RSB GHG tool using background data from BioGrace tool version 3.

Land use change calculations

Direct Land Use Changes (dLUC)

When biofuel cultivation expands to new growing areas the carbon stock in vegetation and soil changes. Carbon is stored in the vegetation in form of leaves and branches, in form of dead wood on the ground and in roots and humus in soils. Where the carbon stock is larger before the establishment of the plantation than thereafter, the difference is released as CO_2 by burning or microbial decomposition of above and below ground carbon. This has a negative influence on the GHG balance, i.e. GHG emissions increase.

GHG emissions from land use changes (el) are calculated in both tools according to a general formula in Annex V from the difference of the carbon stock of the reference land use (CSR) to the carbon stock of the actual land use (CSA) annualised over 20 years.

$$el = (CSR - CSA) \times 3{,}664 \times 1/20 \times 1/P$$

P is the productivity of the crop (in MJ biofuel per ha per year) (EU RED Annex V Point C.7).

The carbon stock of an area consists of two components: soil organic carbon and carbon stock in the above and below ground vegetation. The European Commission published tables with values for these carbon pools for different climate regions, continents, ecological zones, management and input regimes, soil types and land use categories (e.g. cropland, perennial crops, grassland, forest land) (Official Journal of the European Union 2009). Both tools draw values for calculation of emissions from land use changes from this official document.

For the comparison of the two tools, two reference land uses were selected:

1. Grassland
2. Forest (canopy cover 10-30%).

For each crop a climate region, soil type, management and input regime was chosen that is common for the respective crop. Then emissions were calculated that arise from converting these reference land uses to a cropping area.

Indirect Land Use Changes (iLUC)

If existing cropping land, such as a soy plantation, is converted to cultivate oil palm for biofuels, this can lead to displacement effects. If demand for soy remains constant, then additional soy must be produced on a different site. This site could possibly be developed by clearing forests. The oil palm plantation would thus contribute indirectly to an increase in greenhouse gas emissions.

These emissions from iLUC can considerably decrease GHG savings of the biofuel or even cause that biofuels emit more GHG than their fossil counterparts as a number of studies have shown (Searchinger et al. 2008; Fonseca M 2010; Laborde 2011).

The importance of iLUC is acknowledged by the European Commission in their "Report on indirect land use change related to biofuels and bioliquids" (European Commission 2010). Due to several deficiencies and uncertainties associated with the modeling of iLUC there is no consensus yet on how to include these effects in legislation (European Commission 2010).

As a consequence, iLUC emissions are currently not considered by tools and certification schemes for biofuels. In this paper we cover only those aspects that are covered by the tools. Therefore, iLUC scenarios are not calculated in this paper.

Emission savings from improved agricultural management

The EU Commission also published guidance and default values on how to calculate GHG emission savings from improved agricultural management. These basically consist of factors for annual and perennial cropland which are multiplied with the standard soil organic carbon. Factors are available for three tillage (full-, reduced- and no-tillage) and four input regimes (low-, medium input, high input with manure, high input without manure). For the comparison of the tools GHG emissions from a change in agricultural management of wheat was calculated: Wheat (full-tillage, input high without manure) to Wheat (no-tillage, input high without manure).

Calculation tools

Both tools are publicly available and free of charge (BioGrace 2013; HTW Berlin 2012).

BioGrace tool

The BioGrace tool (BioGrace 2013) is an Excel based tool with 22 preset biofuel production chains. It reproduces the calculation of the GHG emission default values of 22 biofuel production pathways listed in the RED Annex V part A. This was made according to the methodology laid out in the same Annex part C. The calculation tool aims to enable economic operators and other users to calculate actual values of biofuel GHG emissions following the same methodology. The BioGrace calculation tool allows to

- use individual input numbers
- define own standard values
- add process steps to an existing biofuel production chain (e.g. add drying step, or extra transport step)
- set up completely new biofuel production chains.

Using the tool requires some knowledge of Excel and GHG calculations from the user which differentiates it from the RSB GHG tool.

RSB GHG tool

The RSB GHG Tool (HTW Berlin) is a web-based tool allowing the calculation of the GHG value of a biofuel product according to the RSB GHG methodology. For products meant to enter the EU RED market, the operator can choose to calculate the GHG emissions according to the methodology of the EU RED using the tool.

The RSB GHG Tool allows calculating with individual input numbers from the producer and setting up a completely new biofuel production chain. However, it does not allow the user to define own standard values. At this moment, the standard conversion values are based on the Ecoinvent database and represent the GHG results of Ecoinvent datasets without infrastructure and for the GHG CO_2, N2O and CH4.

2.4 Results

Total GHG Results

Results of our GHG emission calculations are expressed in CO_2e emission units per energy unit (g CO2e/MJbiofuel). Figure 1.2 and 1.3 include both the BioGrace results and the RSB-EU RED results to allow direct comparison of the tools. Figure 1.2 also shows the emissions of the reference fossil fuel to ease the assessment of the mitigation potential and it shows the threshold of 54.47 g CO_2e/MJ which biofuels need to meet in order to fulfil the 35% mitigation potential.

GHG emissions vary most between the tools for the pathway ethanol from wheat: Emissions in the BioGrace tool are 21% lower than in the RSB-EU RED calculation (Table

I). That means that from exactly the same data along the whole production chain, an economic operator would calculate significantly different GHG emissions depending on which tool was used.

Table I. Deviation of GHG results of RSB-EU RED tool from BioGrace tool.

	BioGrace Tool	RSB Tool	Deviation
W-EtOH			
Cultivation	23,3	31,4	-35%
Processing	29,4	32,8	-12%
Transport	1,9	1,9	0%
Total	54,6	66,1	-21%
RME			
Cultivation	28,7	34,6	-20%
Processing	21,6	19,7	9%
Transport	1,4	1,8	-29%
Total	51,7	56,1	-8%
PME			
Cultivation	14,2	10,7	24%
Processing	46,8	44,8	4%
Transport	5,0	5,7	-14%
Total	66,0	61,2	7%
Sc-EtOH			
Cultivation	14,1	19,1	-35%
Processing	0,9	0,9	-2%
Transport	9,0	3,8	58%
Total	24,0	23,7	1%

GHG emissions of biodiesel from rapeseed are 8% lower in the BioGrace tool compared to the RSB GHG Tool. Even though the deviation is moderate, it is critical, because GHG emission reductions are just above the 35% threshold with the BioGrace tool and just below the threshold with the RSB-EU RED calculation (Figure 1.2). Hence, an economic operator would fulfil the sustainability criteria of the EU RED and accordingly would be able to use the biofuel to count towards obligations and receive financial support using one tool but not using the other.

Total GHG emissions of ethanol from sugarcane match between the tools.

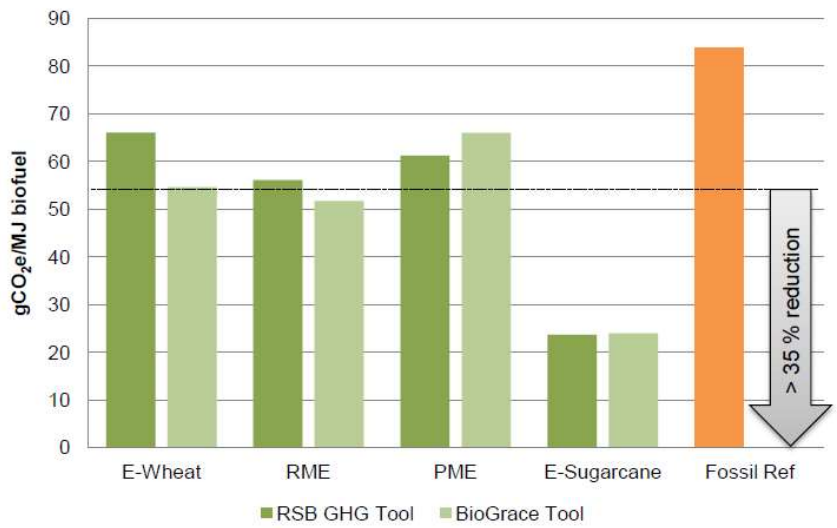

Figure 1.2. Total GHG emissions of ethanol from wheat and sugarcane and biodiesel from rapeseed and palm oil. Biofuels causing lower emissions than indicated by the dotted line fulfil the 35% reduction criterion of EU RED.

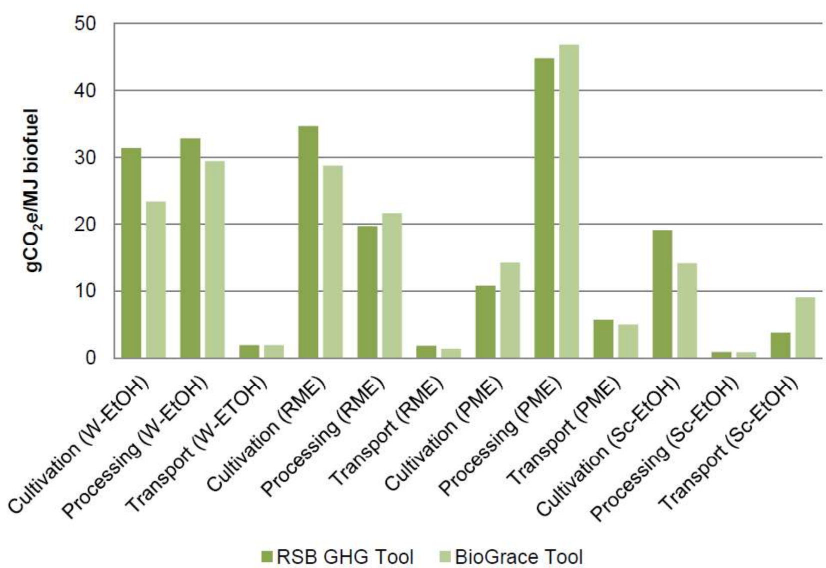

Figure 1.3. GHG emissions disaggregated in biofuel production steps.

Disaggregated GHG results

In the cultivation step, discrepancy of both tools is largest. Results diverge between 20% for rapeseed cultivation and 35% for wheat and sugarcane cultivation (Table 1.1 and Figure 1.3).

In the processing step, deviation is moderate between -2% for ethanol from sugarcane and 12% for ethanol from wheat.

In the transport step GHG results perfectly match in the pathway ethanol from wheat but deviate in the other pathways. However, as the contribution of the transport to overall emissions is low (Figure 1.3), these variations are of little relevance.

GHG results from land use changes and improved management

When biofuel cultivation is expanded to new growing areas, the resulting GHG emissions or emission savings dominate the GHG result (Figure 1.4). The same applies when the management of an existing biofuel cultivation area is changed e.g. improved management of wheat (Figure 1.4).

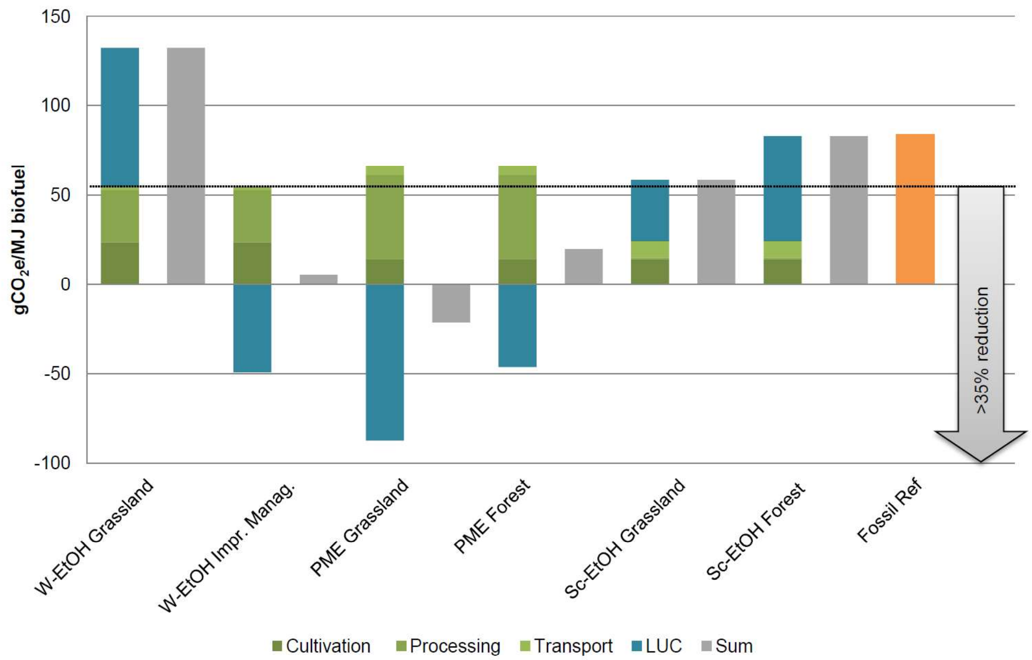

Figure 1.4. GHG emissions/emission savings from land use changes (blue), biofuel production (green) and sum of both (grey) calculated by the BioGrace tool. Biofuels causing less emissions than indicated by the dotted line fulfil the 35% reduction criterion of EU RED.

GHG emissions vary between the tools in half of the pathways we calculated (Figure 1.5). For land use changes from grassland the tools perfectly match in all three cases. For land use changes from forest (with 10-30% canopy cover) the tools differ between -14% and 49% (Table 1.2). For improved management of wheat the RSB GHG Tool calculates 40% higher emission savings than the BioGrace tool.

Table 1.2. Deviation of GHG results of RSB-EU RED tool from BioGrace tool.

Land Use Change	BioGrace gCO_2eq/MJ	RSB gCO_2eq/MJ	Deviation gCO_2eq/MJ
Grassland			
to wheat	77.8	77.9	0%
to oil palm	-87.6	-87.6	0%
to sugarcane	34.4	34.4	0%
Forest (10-30% canopy cover)			
to oil palm	-46.3	-38.3	17%
to sugarcane (a)	58.9	67.2	-14%
to sugarcane (b)	58.9	30.0	49%
Improved management of wheat	-49.1	-29.3	40%

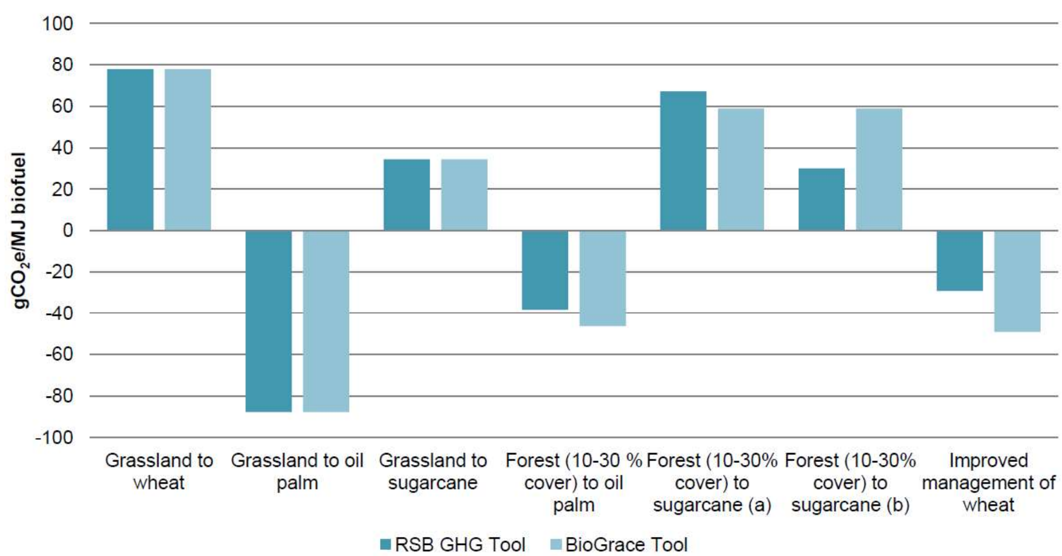

Figure 1.5 GHG emissions and emission savings from land use changes.

2.5 Discussion

Analysing discrepancies – Cultivation

Emissions from cultivation are most important in the pathways Sc-EtOH, W-EtOH and RME where they account for 80% of total emissions. In the pathway PME, cultivation accounts for about 20% of total emissions. In this case, the processing phase is dominant. Hence for sugarcane, wheat and rapeseed, deviations between tools in the cultivation phase are of special relevance.

The cultivation step is also the phase where deviation between the tools is largest (Table 1.1, section 5.2). We identified the major sources of deviations. The main contribution to the difference is the methodology for calculating N2O-emissions from N-fertiliser use. It explains about 60% of the differences of GHG emissions in wheat cultivation.

In the BioGrace tool, soil N2O emissions of wheat cultivation were calculated by the JRC with the DNDC model (European Commission Joint Research Center 2007).

In the RSB GHG Tool, soil N2O emissions were calculated according to the method proposed by the Intergovernmental Panel on Climate Change (IPCC) in 2006 (IPCC 2006). This calculation uses as parameters the N-inputs through fertilisers, the indirect emissions due to leaching (nitrate) and volatilising (ammonia) and the yield. For crop residues, the tool uses standard values.

The second greatest contributor to the observed differences was the standard value for N-fertiliser. This value corresponds to the amount of GHG emissions that arose during fertiliser manufacture. It accounted for about 40% of the observed differences.

Input data are entered on a more detailed level in the RSB GHG Tool, so that another set of input data, e.g. in the fertilizer mix, could lead to other results.In the BioGrace tool, the standard conversion value applied for N-fertiliser was 5880.6 gCO2e/kg N which corresponds to an average of different N-fertilisers used. In the RSB GHG tool, the value varies between 2000 and 15300 g CO2e/kg N, depending on the type of fertiliser, which the user can define according to his specific consumption.

Here the RSB used the proportion of different fertilisers as in the Ecoinvent database. The large difference between the fertiliser factors show that there are large improvement potentials in cultivation resulting from the different GHG intensity of the fertilisers. For example, if the user chooses e.g. for wheat cultivation the N-fertilizer with the lowest GHG emission intensity, the cultivation results of the RSB GHG Tool are 5% lower than with the BioGrace tool. In the contrary, RSB GHG Tool results for cultivation are 84% higher if one applies the N-fertilizer with the highest GHG emission factor in the same pathway.

Overall, these findings match well with a large number of existing studies on biofuel GHG emissions that discuss the relatively large contribution of fertiliser derived emissions in crop cultivation (Macedo et al. 2008; García et al. 2011). As various different values for N-fertiliser production and methodologies for N2O emissions can be found in literature, and tool developers and economic operators are free to choose their sources, large gaps between the results of different tools are not surprising to GHG calculation experts. The findings are relevant because they show a policy gap that needs to be addressed in future improvements of the RED and FQD.

Analysing discrepancies – Processing

GHG emissions from the processing step accounted for 35-54% in the pathways RME and W-EtOH respectively and about 70% in the pathway PME.

For the processing step deviations between the tools were moderate (Table 1.1, section 5.2). The main reasons for differences were different numbers in the standard values for chemicals and for electricity. For example, the BioGrace tool calculated with electricity grid emissions of 129.19 g CO_2e/MJ electricity whereas the same factor in the RSB GHG Tool is 144 g CO_2e/MJelectricity. Moreover, the tool uses at this point the electricity mix for the specific country. Readers who would like more detail regarding the numbers of the standard values (emission factors, Lower Heating Values etc.) used by the tools may obtain the complete lists from the RSB GHG calculation methodology and the BioGrace "List of standard values", which are both publicly available (RSB 2011; BioGrace 2012).

Analysing discrepancies – Land use changes and improved management

When a land use change was assumed, the emissions resulting from carbon stock changes dominated the GHG result. Figure 1.4 shows the GHG emissions and emission savings from land use changes in most cases to be higher than the combined GHG emissions of cultivation, processing and transport.

The main source of differences in land use change calculations from forest (10-30% canopy cover) was that the RSB GHG Tool calculated dead organic matter in the forest whereas the BioGrace tool did not. This led to higher emissions in the RSB GHG Tool when a forest was converted. A minor source of differences was that the RSB GHG Tool calculated N_2O and CH_4 emissions from land clearing by burning whereas the BioGrace tool did not consider these emissions.

In the land use change calculations from forest (10-30% canopy cover) to sugarcane the RSB GHG Tool deviated from the BioGrace tool by -14% in one example and by 49% in the other example (see sugarcane (a) and (b), Table 1.2).

The reason for this was that in the RSB GHG Tool, sugarcane was declared as land use category "cropland" in one case and as "perennial tree/crop" in the other case. The choice of the land use category is left open to the user. As different land use categories have different carbon stock values the choice affects the GHG emissions.

This clearly shows that unambiguous definitions of the land use types are crucial to ensure that all users categorize a certain type of land in the same category. Such clear definitions of land use types are lacking in the EU RED and the specification documents.

The GHG savings from improved management of wheat cultivation deviated by 40% between the tools. The reason for this deviation was that in the RSB GHG Tool emission savings were allocated between main product and by-product whereas this was not the case in the BioGrace tool. Emission savings before allocation matched exactly in both tools. The EU RED does not include a clear specification on whether to apply allocation to the four emission saving terms (emission saving from soil carbon accumulation via improved agricultural management; emission saving from carbon capture and geological storage; emission saving from carbon capture and replacement; and emission saving from excess electricity from cogeneration). A specification of this issue in the EU RED is much needed, as the use or not of allocation can lead to great deviations in the calculation results.

For land use changes from grassland both tools perfectly matched in three cases, which is expected, as both tools incorporate the carbon stock data from the EU COM (Official Journal of the European Union 2009) without changes.

Overall, both tools principally draw the carbon stock values from the same official document to calculate GHG emissions from land use changes. However, differences in the results arose from tool-specific modification of forest values (addition of dead organic matter by RSB GHG Tool), missing definitions of land use categories and differences in the allocation rules.

Land use changes and improved agricultural management – critical points of the EU methodology

Expansion of biofuel cultivation can either lead to large additional GHG emissions or to emission savings – depending on which type of land is converted (Fargione et al. 2008). In any case, as soon as LUC is involved the impact of LUC on GHG emissions is decisive for reaching the 35% reduction target. Biofuels from all crops miss the 35% reduction target when the carbon stock decreases. For example, GHG emissions per MJ biofuel triple when grassland is converted to cultivate sugarcane and double when it is converted to cultivate wheat (Figure 1.4).

In contrast, when the land carbon stock increases during the land conversion, these GHG savings lead to an improvement of the GHG performance pushing the GHG emissions of the biofuels below the 35% reduction target, i.e. they meet the target. For example, without land use change, GHG emissions of biodiesel from palm oil were 66 g CO_2e/MJ PME (Table 1.1). In the scenario with land use change from forest to oil palm GHG emissions dropped by two-thirds to 20 g CO_2e/MJ PME. When grassland was converted, they dropped to -22 g CO_2e/MJ PME.

Hence, in the case of oil palm, the values for carbon stocks from the EU Commission set a clear sign pro land use change, a clear sign for producers to convert grassland and forests (10-30% canopy cover) into oil palm plantations.

A closer look at the carbon stock data from the EU Commission (Official Journal of the European Union 2009) that was used in both tools reveals that this incentive might be misleading, because:

1) the GHG values used for the calculation do not take into account the impact of land use change on the surroundings like e.g. fragmentation and destruction of forest due to road building, as well as other impacts of land clearing like run-off or erosion, which have both effects on GHG emissions and biodiversity or soil quality (Germer and Sauerborn 2008).

2) the carbon stock values used for calculation are based on Tier 1 default values from the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006). These values are explicitly designed to assist in estimating GHG emissions at national level - not at project level (IPCC 2006) because they are prone to significant uncertainties. For example, the carbon stock values for grasslands have an error of 75% in the IPCC Guidelines (IPCC 2006). Another example is that the calculation of land use change from forest (canopy cover 10-30%) to oil palm is dominated by the assumption that above-ground biomass in forest with a canopy cover from 10-30% only amounts to about 20% of a forest with a canopy cover over 30% (IPCC 2006).

Thus, whereas IPCC Tier 1 defaults are valuable to get a rough idea on the magnitude of GHG emissions on a national/global level, we do not consider the Tier 1 values appropriate for calculation of LUC emissions at project level. They may set an incentive for land conversion where in reality no benefit in terms of carbon sequestration occurs.

The same conclusion is valid for the calculation of the emission savings from improved agricultural management. In the IPCC Guidelines, an error of 9%-61% is given for the carbon stock values (IPCC 2006). In addition, the categorization of management practices in just 2 categories (tillage intensity and input level) does not give justice to the many agricultural practices that can mitigate CO2 emissions from soils (Smith et al. 2008; Kim et al. 2009; Lal 2019).

In addition, according to the EU RED, emission savings from improved agricultural management can be calculated to improve the GHG performance. However, there is no obligation to include additional GHG emissions that may arise from deterioration of management practices, such as intensification of tilling, losses of crop rotations, which in turn may lead to soil carbon losses. These changes in carbon stock due to deterioration of management practices can contribute more to overall emissions than cultivation, transport and processing combined (Figure 1.4, third bar from left gives an impression of the magnitude of emissions due to a change in wheat management). Even though emissions may be high, the current EU RED legislation has a blind spot on that regard. This can lead to a situation where biofuels are supported that cause more GHG emissions than fossil fuels. We strongly suggest to take these emissions into account in future legislations, e.g. by obliging farmers to monitor soil carbon or implement a soil protection plan.

Apart from direct land use changes, there is growing scientific evidence on the impact of indirect land use changes on the overall GHG balance. These iLUC effects arise, when biofuels are sourced from existing cropland. This aspect is not covered at all in existing tools. According to the studies conducted by JRC and IFPRI it will considerably impact the overall GHG emissions of biofuels (Fonseca M 2010; Laborde 2011). Therefore, the inclusion of iLUC effects in legislation and tools for certification will be an important task for future work.

2.6 Conclusions

From the comparison of the tools in this study we conclude that

- despite the effort to harmonize EU RED calculations, the directive still allows choices resulting in relevant discrepancies for the calculation results of the same pathway
- therefore there is a need for more thorough and precise specification of the calculation methodology in the EU RED
- the guideline of the EU Commissions for calculating GHG emissions from land use changes needs to be revised to make sure no misleading incentives for conversion of grasslands and forests to oil palm plantations are given.
- depending on his behaviour and choices, the operator can greatly improve or worsen his GHG intensity. The EU RED methodology should therefore take into account not only improvements in management practices, but also deterioration in management practices.

3. LCA CASE STUDIES ON SUSTAINABLE AGRICULTURAL INTENSIFICATION

Optimizing the water, carbon, and land-use footprint of bioenergy production in Mexico - Six case studies and the nationwide implications[2]

3.1 Abstract

This study aims to answer the question if and how biofuels can be produced in Mexico without aggravating water scarcity, reducing greenhouse gas (GHG) emissions and avoiding indirect land-use changes. We analyzed environmental impacts of six potential maize-bioethanol production systems in Mexico on water resources, land use, and GHG emissions by using a life cycle assessment approach. Three irrigated high-input maize systems and three rain-fed low-yield systems were analyzed. Inventory data was acquired by soil sampling and interviews with farmers. For the water footprint, field water balances were modeled using the Food and Agriculture Organization (FAO) AquaCrop Model. For the carbon footprint, the BioGrace tool was used. Based on the results of status-quo analysis, scenarios with improved agricultural management were defined to identify optimization potential. Additionally, the producible amount of biofuels was estimated on a national level. The analyses showed that improving management in rain-fed agriculture offers the best opportunities for biofuel production without compromising regional water availability and without unwanted indirect effects on food prices and GHG emissions. Around 3.4% of Mexican gas consumption could be produced from maize bioethanol in Mexico without the above mentioned unwanted effects. By optimizing green water use in rain-fed maize production, around 3 billion m3 of non-productive soil evaporation would be put into productive use. This is around 10% of the total water extracted from aquifers in Mexico. From this we conclude that unproductive soil evaporation is an underestimated water resource which should be considered in water management.

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[2] Article published in BioFPR Volume 10, Issue 3, May/June 2016, Pages 222-239.
Anna M. Hennecke, Maria Mueller-Lindenlauf, Carlos A. García, Alfredo Fuentes, Enrique Riegelhaupt, Stefanie Hellweg

3.2 Introduction

In Mexico interest in biofuels has emerged with the aim to diversify energy sources, mitigate Greenhouse Gas (GHG) emissions in the transport sector and to achieve increased employment in the agricultural sector. At the legislative level a Biofuels Promotion and Development Law (Diario Oficial de la Federación 2008) was passed in 2008 and a program for biofuels introduction was drafted under the framework of the Intersectorial Bioenergy Strategy (SENER (Secretaría de Energía) 2008).

An increase in bioenergy production may pose risks on environmental and social sustainability. Land clearing for expansion of agriculture may lead to loss of biodiversity. Sourcing feedstocks for biofuels from existing agricultural land can cause indirect effects. Most cited are indirect land-use changes which can lead to additional GHG emissions and increases in food/feed commodity prices which can effect food security (Eide 2008; Fargione et al. 2008; Searchinger et al. 2008). Impacts on water resources can be either effects on water quality through eutrophication or impacts on water scarcity because production of biofuels is more water intensive than fossil fuels or other alternative energy sources (Gerbens-Leenes et al. 2009). Water scarcity is a growing issue in Mexico (Comision Nacional del Agua 2014). The number of overexploited aquifers in Mexico has increased from 32 in 1975 to 106 in 2013 and agriculture accounts already today for 77% of total water extraction (Comision Nacional del Agua 2014). 35 mio Mexicans are in a situation of little water availability in terms of quantity and quality (Comision Nacional del Agua 2014).

The question is if biofuels can be produced in Mexico without negative effects on water and land resources and climate change. Note that biofuels are just an example here, this question arises for any increases in agricultural production, be it for food, feed or biofuels.

Life cycle assessment (LCA) and carbon-, water-, and land use footprints are increasingly used to answer these questions and to guide decisions by political decision makers and industry. For example, the carbon footprint of biofuels is a criteria in the European Renewable Energy Directive which decides if the biofuel are counted for the legally binding bioenergy quotas (European Parliament and Council 2009a). The land use footprint is used to compare the use of the scarce resource land for different products. The Water Footprint can be used to inform actors in industry (e.g. retailers) on where the hotspots on water impacts are in their production chains and to improve the supply chain with respect to impacts on water resources (Stoessel et al. 2012).

A number of biofuel LCA s and studies have been issued that look at the above mentioned effects. However, to date no studies on biofuel production in Mexico have been published except for one study on GHG emissions of ethanol production from sugarcane (García et al. 2011).

A regional assessment is of particular importance for the water footprint. In contrast to greenhouse gas emissions impacts of water use depend on spatial factors such as freshwater availability and use patterns at the specific location (1L consumed in Mexico has a different impact than 1L consumed in Sweden). Therefore weighted water footprint methods have been proposed. According to these methods the water volume is multiplied with a regional characterisation factor based on a withdrawal to availability ratio of the local aquifer (Pfister et al. 2009). Volumetric methods as proposed by Hoekstra just sum up the volume of water used and discuss it in the context of the water scarcity of the region (Hoekstra et al. 2011). These methods concentrate on assessing blue and grey water use. A number of studies exist that indicate an optimization potential in green water use as well (Rockström 2003; Molden et al. 2007; Rockström et al. 2009; Hoff et al. 2010). E.g. the International Water Management Institute estimates that improvements in rainfed agriculture could contribute 75% of the yield increases needed to fulfill future needs for agricultural products compared to only 25% from expanding or improving the irrigated agriculture (Molden et al. 2007). There is a lack of studies addressing the yield optimization potential of agriculture in Mexico and quantifying the resulting environmental impacts.

This study has three aims:

The first aim is to inform policy debate by answering the question if and how biofuels can be produced in Mexico without aggravating quantitative water scarcity, reducing GHG emissions and avoiding indirect effects on land use.

The second aim was to find out in how far an optimization of rainwater water management can contribute to this goal.

The third aim was to reveal yield optimization potentials through an increase in rainwater efficiency, complementing the classical assessment of environmental impacts of water footprinting tools and LCA.

Note that the results of this study are applicable to any increase in maize production in Mexico, be it for biofuels, food or feed.

In short, the study aims to answer the questions:

1) Can biofuels be produced in Mexico without aggravating water scarcity, reducing GHG emissions and avoiding indirect effects? How?
2) In how far can improvements of rainwater management contribute?
3) Are current water footprint methods sufficient to detect this potential?

This question was approached with six case studies followed by an estimation of nationwide implications.

3.3 Materials and Methods

Study Cases

A total of 12 study cases were visited in two states in Mexico (Michoacán and Guanujuato) located in the central highlands. A study case is a maize field between 0.5 and 25 hectare. Studies were only selected if the interview with farmer and soil sampling was fully completed and all necessary data could be acquired. Six cases were selected that met these criteria: three rainfed low yield cases in Michoacán and three irrigated high yield cases in Guanajuato. Another six cases were excluded from this study because the farmer could not provide all necessary information or the permission for soil sampling was not granted. The cases differed in climate (annual rainfall for Michoacán cases is 960 mm, for Guanajuato cases it is 600 mm (Mexican Meterological Service), in regional water scarcity (aquifer extraction rate for Michoacán cases is 80% (annual water withdrawal/renewable water), for Guanajuato cases 168% (CONAGUA (Mexican National Water Commission) 2008), in management (irrigated, rainfed, soil protection measures, fertilization, plant density, see Table 2.1) and in cropping system (monoculture, mixed cropping, crop rotations, see Figure 2.1).

Table 2.1: Water, soil and fertility management of status quo and scenarios of Michoacán case studies and Guanajuato case studies.

	Irrigation	Tillage	Mulch cover	Fertility and plant density
Michoacán (Case 1-3)				
Status quo	0	0	0	+
Scenario A	0	0	+	++
Scenario B	0	+	+	++
Guanajuato (Case 4-6)				
Status quo	++	0	0	+++
Scenario A	0	0	0	+++
Scenario B	+	0	0	+++

Irrigation: 0 (rainfed), + (rainwater harvest), ++(irrigated)

Tillage: 0 (full tillage), + (no tillage)

Mulch: 0 (no mulch cover), + (80% mulch cover before seeding)

Fertility: + (low), ++ (medium), +++ (high)

In case 1 maize and bean were cultivated at the same time (mix crop), case 3 was a maize monoculture. The other cases were double crop systems where a second crop (vetch, barley, maize) was cultivated after maize in the same year (Figure 2.1).

For each valid study case land-, carbon- and water footprint of 1 MJ bioethanol from maize was calculated. After assessing the status quo, scenarios with improved management were built (Figure 2.1).

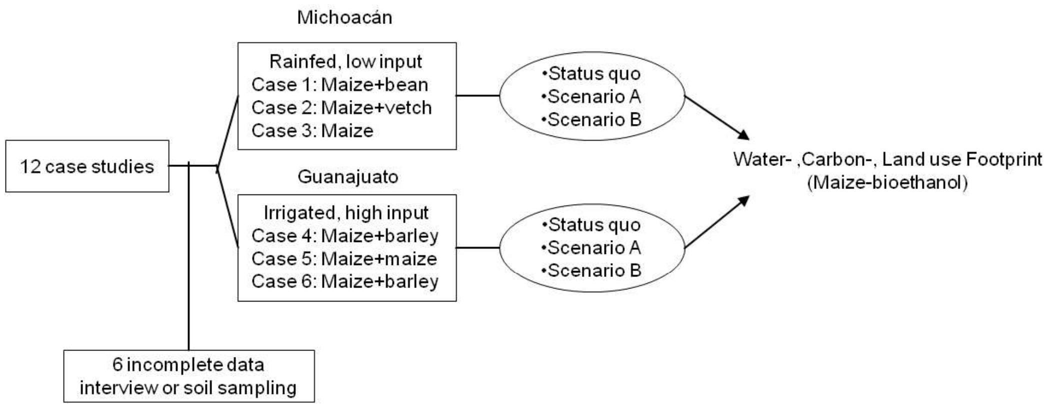


Figure 2.1: Study scheme: Three rainfed and three irrigated case studies were selected. For status quo and improvement scenarios water-, carbon- and land use footprints were calculated

<u>LCA methodology</u>

GHG emissions (gCO_2eq), water consumption (m3) and land use (ha) were calculated for the production of one MJ of biofuel (functional unit). Allocation of GHG emissions, water and land use to by-products was done by energy content (lower heating value).
System boundaries for the carbon footprint included cultivation, bioethanol plant, transport of intermediate products, filling station and ethanol burning. Direct land use change did not occur because maize for biofuel production was grown on existing agricultural land. On-field emissions like burning of diesel and N2O emissions of fertilizers were included as well as upstream emissions that arise in the production of inputs such as diesel and fertilizers (suppl. section 2.1). For the calculation of water- and land use footprint we focused on the agricultural step since this is the most water- and land intensive part of the production. Water and land use in bioethanol plant and filling station as well as upstream process of input production were disregarded.

Blue water consumption was defined as irrigation water that is evapotranspired by the plant during the cropping period. Green water consumption is defined as precipitation evapotranspired by plant during cropping period. In this study we also defined supplemental irrigation from rainwater harvest as blue water. A volumetric water footprint according to Hoekstra (Hoekstra et al. 2011) and a water stress weighted water footprint according to Pfister (Bayart et al. 2010) were calculated over the cropping period (temporal boundary). The carbon footprints were calculated in line with the methodology given in the Renewable Energy directive (European Parliament and Council 2009a). Land use was accounted here as m2*years, without additional impact assessment.

<u>Input data for water footprint (AquaCrop parametrization)</u>

Since blue and green water flows are difficult to measure directly, AquaCrop v.4 (FAO 2012) was used to model the field water balance. AquaCrop modules on climate, crop, irrigation, field management, soil profile, simulation period, initial conditions and off-season conditions were parametrized as described in the following.

Climate data for 1999-2009 on daily minimum and maximum temperature, precipitation and potential evapotranspiration were taken from Mexican Meterological Service (Mexican Meterological Service) and CLIMWAT (FAO 2006) (section 1.1 supp).

Crop parameters were acquired by field interview with farmers (planting and harvest dates, planting density, maximum canopy cover, rooting depth) and partly based on AquaCrop default and literature data (section 1.2 supp).

A local calibration of the fertility module is recommended but could not be done in this study because of time and cost constraints (agricultural field trial plots would need to be established). Therefore, in the AquaCrop field management module soil fertility level was adjusted so that simulated yields matched yields reported by farmers. The approach resulted in a ranking of cases by fertility level which matched roughly the ranking by nitrogen fertilizer inputs reported by the farmers (section 1.3 supp). This gives an indication that this rough approach for including limitations of fertility in yield calculations is sufficiently accurate for the purpose of the study (to show yield tendencies, not absolute numbers).

Field surface practices to prevent runoff were reported by farmers. In case 4 and 6 (Figure 2.1, Table 2.1) soil bunds existed with a height of 0.2 m and for the second crop of case 5 furrows prevented runoff from rain and irrigation events up to 40mm/day. The AquaCrop field management module was set accordingly.

Soil profile characteristics for the AquaCrop soil module were aquired by soil sampling as described in supplementary materials (section 1.4 supp).

Irrigation was supplied from a water well, rates and dates were acquired by interview and the AquaCrop irrigation module was set accordingly (section 1.5 supp).

The soil water content before the beginning of the rainy season was sampled in the field for each case beginning of May 2013 (section 1.4 supp). These are the initial soil moisture conditions at the beginning of the AquaCrop simulation run. Sensitivity analysis have shown that intial soil moisture is relevant for the result, this is discussed in suppl section 1.4. The simulation period was set from May 1st to harvest, for double crop systems to harvest of second crop.

For irrigated systems, it is not possible to determine exactly which part of the evapotranspirated water is from rain, from irrigation water or from soil moisture. To estimate the share of irrigation water consumptively used, we chose a percentage approach:

$$Consumptive\ blue\ water\ use = (T + E) * (Irr/(Irr + R)$$

T=Transpiration, E= Evaporation, Irr= Irrigation, R= Rain in (mm) over cropping period between may 1st to harvest.

Input data for carbon footprint

The carbon footprint was calculated using the BioGrace GHG tool v.4c (BioGrace 2013). For the cultivation step input data for calculation of GHG emissions of bioethanol production were acquired by field interview on-site (fertiliser inputs, straw removal, pumping depth and irrigation amounts for estimation of electricity) and by literature data (e.g. diesel use for high input and low input maize agriculture (Riegelhaupt E 2010), pesticides (BioGrace 2013) (section 2.1 supp.).

Emissions of transports, ethanol plant and refinery were calculated with BioGrace default data on Ethanol from corn (natural gas CHP) (BioGrace 2013). Currently there is no ethanol production from maize in Mexico, therefore field data could not be acquired.

Input data for scenarios

In addition to the status quo assessment of the case studies, scenarios with a modified management were chosen with the aim to improve yields and to mitigate overuse of aquifers. Table 2.1 shows the management of the status quo and scenarios. The following paragraphs and suppl. section 1.6 describe how parameters were adjusted in the AquaCrop simulations and BioGrace modeling for these scenarios.

Yield improvement scenarios: The case studies in Michoacan were characterized by low yields. In scenario A it was assumed that the farmer increases soil fertility by adequate measures (e.g. green manure, more synthetic fertilizers) and increases plant density to intensify production. Soil fertility was increased to achieve a yield of 6-8 T/ha instead of the regional maximum because higher negative impacts on water quality would be expected with a higher increase of yields. The underlying assumption is that a moderate increase might be achieved without impacts on water quality from fertilizer and pesticide leaching. Fertility measures cannot be modeled directly in AquaCrop. To include the effect of increased soil fertility, fertility levels were increased in the AquaCrop soil fertility module from 23% (C1), 25% (C2) and 50% (C3) in status quo to 60% in scenario A. Planting density was increased in the AquaCrop crop module from 51.000 plants/ha with 70% cover (C1), 76.000 plants/ha with 90% cover (C2) and 63.000 plants/ha with 80% cover (C3) in status quo to 76.000 plants/ha and to 90% maximum canopy cover in scenario A. This scenario was chosen because the main reason for low yields in the status quo was a lack of soil fertility.

In the B scenarios, the same measures as in scenario A were adopted and additionally two management measures were simulated that increased the soil water content at the beginning of the cropping period. These measures were applied to avoid yield failures due to water stress at the beginning of the cropping period. An 80% mulch cover was applied before seeding to improve conservation of soil moisture from rainfall in May and June. The AquaCrop Off-season module was set accordingly. For case 3 this measure was sufficient to decrease years with yield failures and yield decreases (>10%) to zero (Table 2.2). For Cases 1 and 2 this measure only partly mitigated yield failures and decrease. Therefore, for C1 and C2 a change in management from conventional tillage to zero tillage with residue retention was included in the scenario. Field trials in that region by Verhulst have shown that this change in management increases initial soil moisture in the topsoil (Verhulst et al. 2011a). Since tillage operations cannot be simulated directly in AquaCrop, initial soil moisture in topsoil (top 60cm) was increased manually in the AquaCrop Initial Conditions module by 40 mm (C2) and 60 mm (C1). This is the minimum increase in initial soil water content needed to avoid the yield failure in the critical years of the 10 year simulations, 2005 and 2008. Please note that this approach has uncertainties (suppl. section 1.6) because Verhulst measured soil moisture increases only in three years which does not give enough statistical evidence that a soil moisture increase of 40-60mm could be reached in average and because initial soil moisture in simulations was based on measurements in our case studies of just one year (2013) (see suppl. section 1.6 for discussion of uncertainties).

For the carbon footprint of the Michoacán scenarios, increased N-fertiliser input was estimated by setting up a regional yield-nutrient curve based on the yield-fertilizer relationship of the six case studies (supp. section 2.2). These were fitted in Matlab to a logistic function which was used to calculate the amount of N-fertiliser needed to achieve yields of Michoacán scenario A and B (supp. Section 2.2. Figure Fit). Additional diesel use for intensified production was taken from literature (Riegelhaupt E 2010).

Table 2.2: Yields and years with yield reductions for status quo from AquaCrop simulations for 1999-2009 for status quo of case studies.

Case	Region	Crop	Reported yield range (T/ha)	Simulated yield (T/ha)	Years with yield reduction >10%	Magnitude of yield reduction (%)
1	Michoacán	maize	1-2	1.7	3	15, 100, 100
2	Michoacán	maize	2-3	3.0	2	15, 100
3	Michoacán	maize	5	4.8	1	100
4 [a]	Guanajuato	maize	10 (normal), 14 (good), 5 (bad)	12.1	0	0
4 [b]	Guanajuato	barley	6-8	5.6	1	26
5 [a]	Guanajuato	maize	10	10.1	2	13, 100
5 [b]	Guanajuato	maize	10	10.7	2	12, 16
6 [a]	Guanajuato	maize	12	12.1	0	0
6 [b]	Guanajuato	barley	7	5.4	0	0

[a] First crop; [b] Second crop

Water-use mitigation scenarios: The case studies in Guanajuato were characterized by high levels of irrigation which was sourced from an overused aquifer. In scenario A and B it was assumed that the farmer does not apply any irrigation i.e. switches to rainfed farming.

In scenario A, AquaCrop simulations were modified in four ways compared to the status quo: First, seeding date for maize was shifted from May to June 30th after the onset of the rainy season) because seeds do not germinate in dry soil. Second, the second crop (barley or maize) was omitted because it completely relied on irrigation water. Third, the initial soil water content was modified in the AquaCrop “initial conditions” module because without irrigation in the dry season the soil water content at the beginning of the cropping season would be lower. The average soil water content of the rainfed cases 2 and 1 was chosen as an estimate (see supplementary materials for more details on the parameter estimation).

In scenario B, supplemental irrigation from rainwater harvest was applied to bridge dry spells. Dry spells are short periods of 2-4 weeks with little rain that lead to yield decreases. There are a number of management options to bridge dry spells. For example measures that increase soil water content before and during cropping period like mulching and no till and in some cases also shift of the seeding date. However, sensitivity analysis has shown that these measures alone were not enough to bridge all dry spells in the simulated years. Therefore, supplemental irrigation from rainwater harvest was chosen as single management measure in the B scenario. Any additional measure would decrease the amount of supplemental irrigation needed.

In scenario B a switch to rainfed farming was assumed and in addition supplemental irrigation from rainwater harvest was applied to bridge dry spells. Supplemental irrigation was applied in those years where yield decreases >10% occurred in the simulations. It was applied when stomatal closure occurred for more than 3 days and it was applied on the first day of stomatal closure. The minimal amount of water needed to avoid stomatal closure and maximize yields was used (section 1.6 supp.) trying step by step which amount of water was sufficient in the simulation.

3.4 Results

<u>Status quo: AquaCrop simulation results</u>

In the rainfed case studies, simulated yields were low with 1.7 - 4.8 T maize per ha (Table 2.2). Yield failures and yield reductions above 10% due to water shortage occurred in 1-3 out of 10 years. Yield failures/reductions were always due to water stress in the canopy expansion phase and occurred in those years where rainfall was below average in May and June. Water stress in the yield formation phase or during flowering was not observed in 10 simulated years. This means that the beginning of the cropping period is critical but there is generally no severe water stress in that region. Low yields were apparently mainly due to the low soil fertility level. This is plausible since reported N-inputs for case 2, 1, 3 were significantly below the amount that is theoretically required for maximum yields.

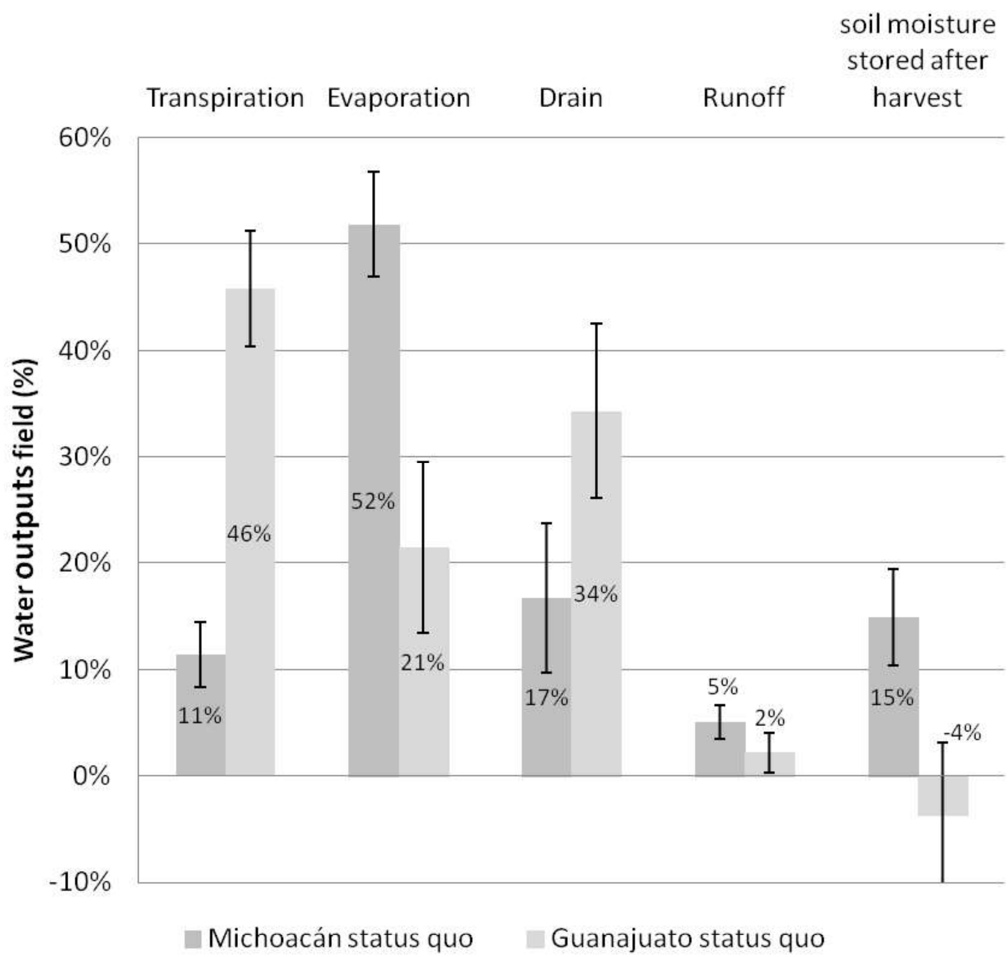
The irrigated case studies produce high and stable yields. Years with yield decrease >10% do not occur (Table 2.2) with one exception: In case 5 the first of the double crop shows 2 out of 11 years with yield decrease >10%. This is not totally consistent because a farmer would probably not invest in fertilizer with that risk. However, the farmer reported that the first of the double crop is grown rainfed but the field is equipped with an irrigation system from a water well. It is unclear if he would make use of it in case water stress would occur. Since he was cropping maize for the first year on this plot there was no operating experience.

Status quo: Greenwater use efficiency

A remarkable characteristic of the low input case study group is the low greenwater productivity. Only around 11 % of the rainfall within the cropping period is productively used by the plant (average over 30 simulated years) (Figure 2.2). At the same time soil evaporation is very high, more than half of the rainwater evaporates unproductively.

Productive plant transpiration is significantly higher in the high yield systems, around 52% of rainfall are productively used by the plant (Figure 2.2). Unproductive soil evaporation is significantly less than in low yield rainfed systems with 21%. This makes sense because a denser canopy develops which is shielding the soil.

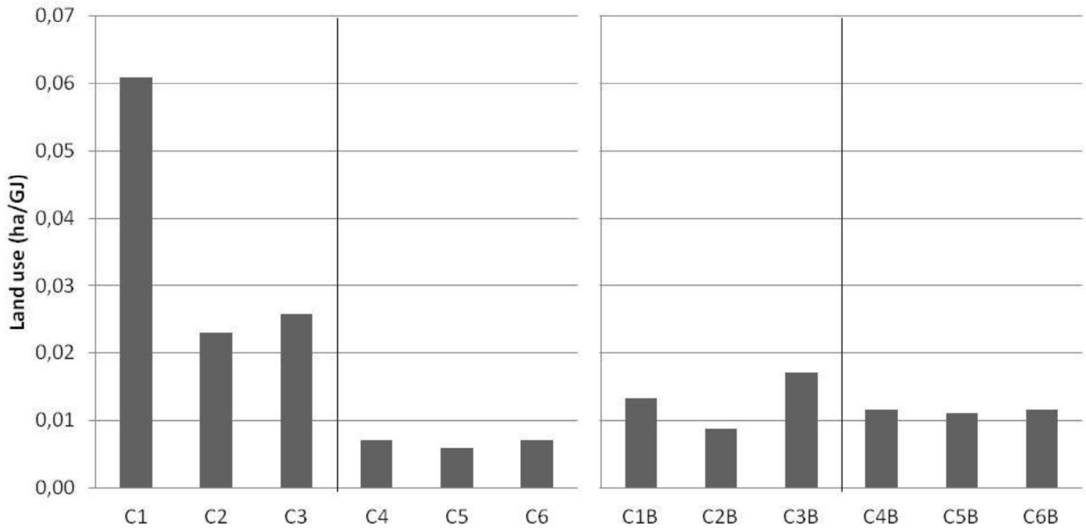
Figure 1.2: Field water balance in cropping period (May-Nov for rainfed cases and May-Apr for irrigated double crop cases), averaged over AquaCrop simulations from 1999-2009.



<u>Status quo: Water- carbon and land use footprint</u>

In the irrigated systems, the land use footprint was low with less than 0.1 ha per GJ (Figure 2.3) because yields were high and two crops were cultivated per year. One crop is cultivated in the dry season and almost completely relied on irrigation water. Irrigation water was taken from the Aquifer Cienega Prieta-Moroleon which is overexploited with an extraction rate of 168% (annual water withdrawal/renewable water) and a high WSI of 0.9997 (CONAGUA (Mexican National Water Commission) 2008; Pfister et al. 2009). Blue water demand was extremely high with around 900 mm irrigation per ha.

Figure 2.2: Land use footprint of rainfed and irrigated case studies in status quo (left) and scenario B (right).



In comparison average precipitation is around 600 mm (climate station No. 11071 (Mexican Meterological Service)). This reflects in the water footprint where the share of blue water was 62% (C4), 39% (C5) and 59% (C6) (Figure 2.4). The WSI-weighted water footprints were 39mm/GJ (C4), 36mm/GJ (C6) and 22 mm/GJ (C5) (Figure 2.5). Thus, the trade-off between water and land use is high. To produce 1 GJ of bioethanol, less than 0.01 ha are required but blue water use is high in an aquifer that is already overexploited.

Figure 2.3: Volumetric water footprint per ha in status quo (left) and scenario B (right). Please note that greenwater consumption of mix-crop and second crop of Case 1 and Case 2 were not included.

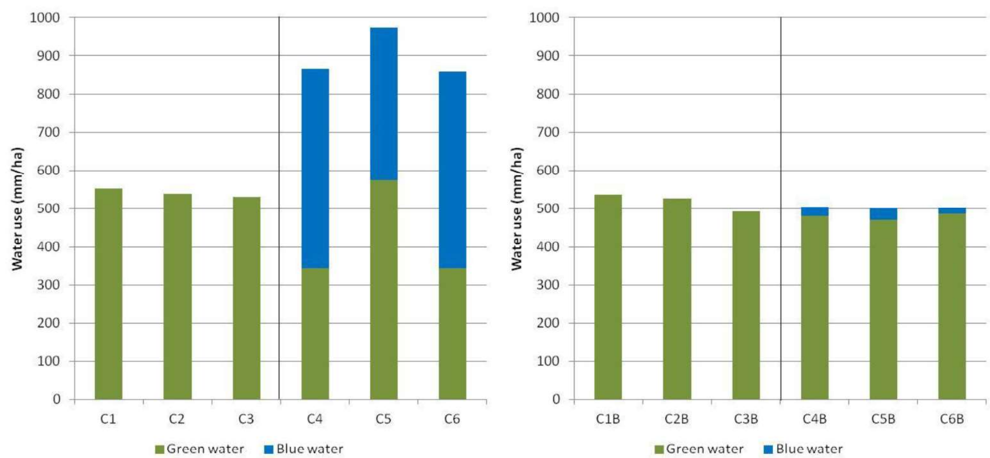
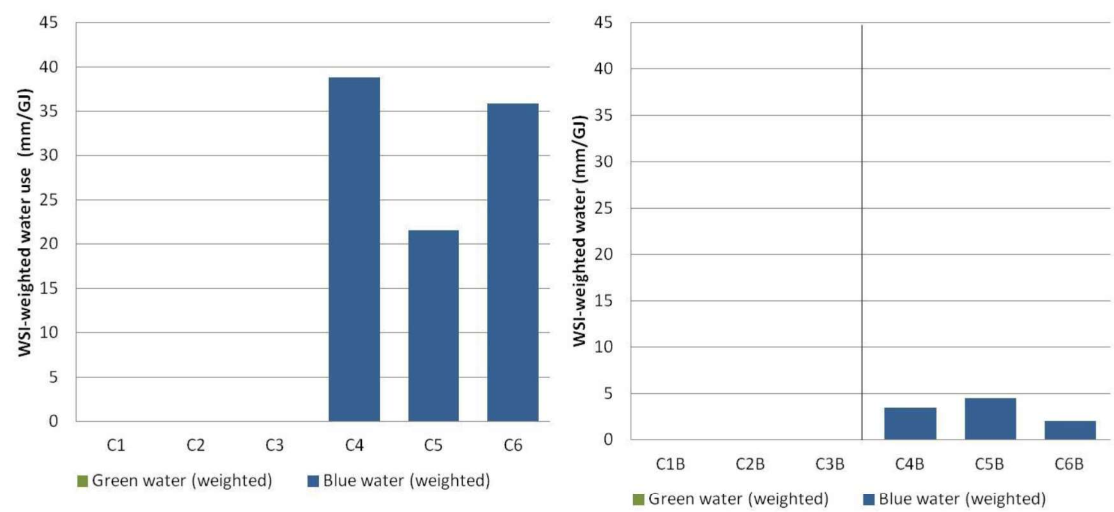


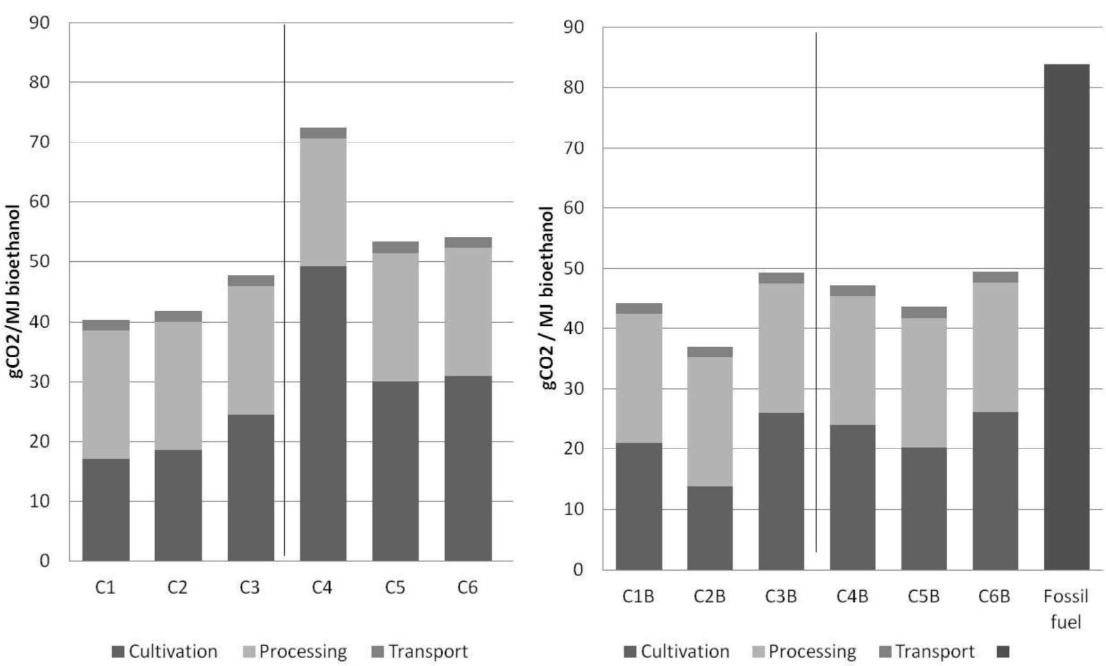
Figure 2.4: WSI-weighted water footprint of status quo (left) and scenario B (right).



The water footprint of the Michoacán case study group was zero (Figure 2.5) but the land use footprint was up to 10 times higher than in irrigated systems (Figure 2.3).

Greenhouse Gas emissions from cultivation were on average about one third lower in the rainfed case study group than in the irrigated case study group with 17-24g CO_2/MJ compared to 30-49 g CO_2/MJ (Figure 2.6). Case 1 and 2 had lowest emissions in the cultivation phase with only 17g CO_2/MJ. For C2 this is plausible because 45% of emissions are allocated to the by-crop. For C1 however, this represents a certain underestimation of emissions because the result is due to low N-inputs that would not sustain the system in the long term because nutrient mining occurs: No synthetic fertilizer were applied, an estimated 20kg N input per ha per year is brought in the system from the leguminose mix crop (N-fixation by fava beans 5kg N/100kg yield (German Federal Government 2012a). N-removal is around 32-45 kg N/ha (N-removal corn 1.15 kg N/100kg yield (German Federal Government 2012a). Despite this possible underestimation of emissions in C1, results show that the carbon footprint of all low input cases was generally lower than for irrigated cases.

Figure 2.5: Carbon footprint of status quo and scenario B.



High input irrigated systems perform worse in the cultivation step because of 1) Energy intensive irrigation: Irrigation water is pumped from 200m depth for C4 and C5 and from 15m depth for C6. Electricity for pumping accounts for 50%, 35 % and 7% of total CO_2-eq emissions from cultivation. 2) The second crop barley negatively influences the GHG balance. The same amount of inputs (fertilizer, diesel) are applied as for maize, but yields per ha are only about half. It is grown in the dry season and requires therefore more irrigation (C4, C6). 3) Allocation of the leguminose-by-crops improves the carbon footprint of the low input systems significantly (C2, C1). For case 2 and 1 45% and 16% of emissions from cultivation are allocated to the leguminose by-crop. Please note that allocation per energy content can be seen critical as these food commodities are not used in the energy sector. However, this method was chosen in order to be in line with the European Renewable Energy Directive.

Sensitivity analysis showed that if neither irrigation nor the cultivation of barley were considered, high input case studies would perform slightly better than low input systems with 17-19 g CO_2/MJ compared to 12-22g CO_2/MJ in cultivation of low input case studies.

A review of five iLUC studies for corn ethanol found that additional GHG emissions from iLUC ranged between 27-103 g CO_2/MJ (ECOFYS, IFPRI, CARB). Even though these studies do not specifically address corn ethanol produced in Mexico they give an indication of the order of magnitude of the iLUC emissions that may be caused. If these additional emissions would be added to the carbon footprint of the study cases, none of them would reduce GHG emissions compared to fossil fuels.

<u>Improvement scenarios: AquaCrop simulation results</u>

In the improvement scenarios we explored if and how the tradeoffs between water, carbon and land use footprint could be reduced by management measures.

In Michoacán scenario A (Table 2.1) yields increased from 1.7- 4.8 T/ha to 6.3 - 6.9 T/ha (Table 2.3). The frequency of yield failures and yield decreases above 10% from water scarcity was identical to the status quo scenarios. This is interesting because it means that yield increases did not intensify water scarcity, there was enough rainfall to support higher production levels in that region. After application of the soil moisture conserving measures in **Michoacán scenarios B** years with yield failures and yield reductions were completely avoided (Table 2.3).

In Guanajuato scenarios A (4A, 5A,6A, Table 2.1) substantial yield failures and yield decreases above 10% occur due to water shortages. In C5A yield decreases occur in 7 out of 11 years with 4 total yield failures (Table 2.3). In C4A yield decreases and failures occur in 6 out of 11 years with 2 total yield failures and in C6A in 4 out of 11 years with 3 total yield failures.

Reason for these yield decreases were dry spells in the middle or end of the cropping period. For example, in 1999 the rainy season ended early with precipitation below average in September and October (Figure 2.7a). This led to a stomatal closure in the yield formation phase and 16% yield reduction (Figure 2.7b). In 2002 there was a dry spell in August (48mm instead 113mm average) (Figure 2.8a). The maize germinated, but on day 34 after planting the stomatal closure started which lead to a yield failure mid of august (Figure 2.8b). In the weeks after the dry spell there was sufficient rain in the rest of the cropping period and in general this year was not drier than the average.

The results show that even though rainfall in the region Santa Maria is high enough to sustain water demand of a high yielding maize crop (around 600mm average precipitation compared to 450mm evapotranspirative demand of a high-yielding maize crop in this region), frequently occurring dry spells within the cropping period lead to substantial yield reductions. Rainfed cropping in this region is hence associated with a high risk of substantial yield decreases due to rainfall variability.

Figure 2.7a/b: Simulation data for C4A on a) precipitation in 1999 (blue) and average precipitation (red) and b) AquaCrop screenshot of simulated soil water content (blue) and threshold for stomatal closure (red line).

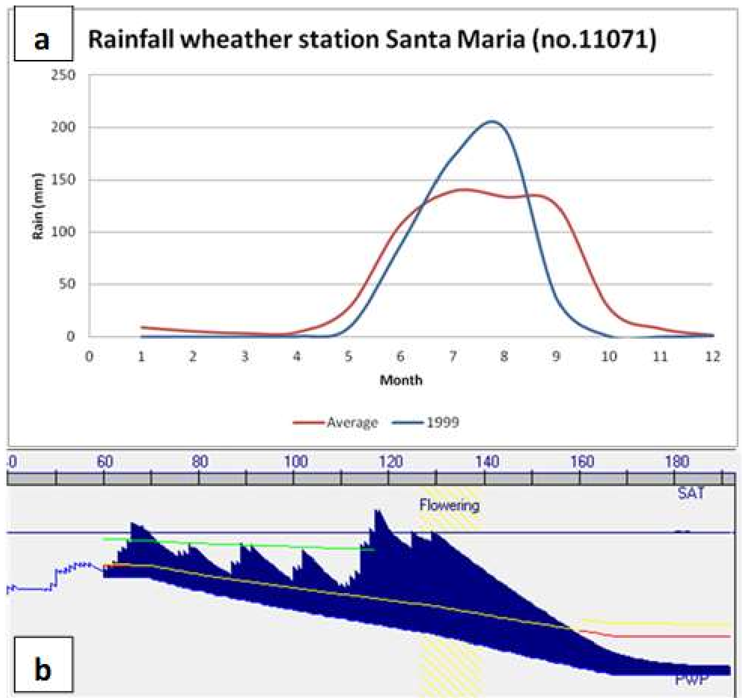
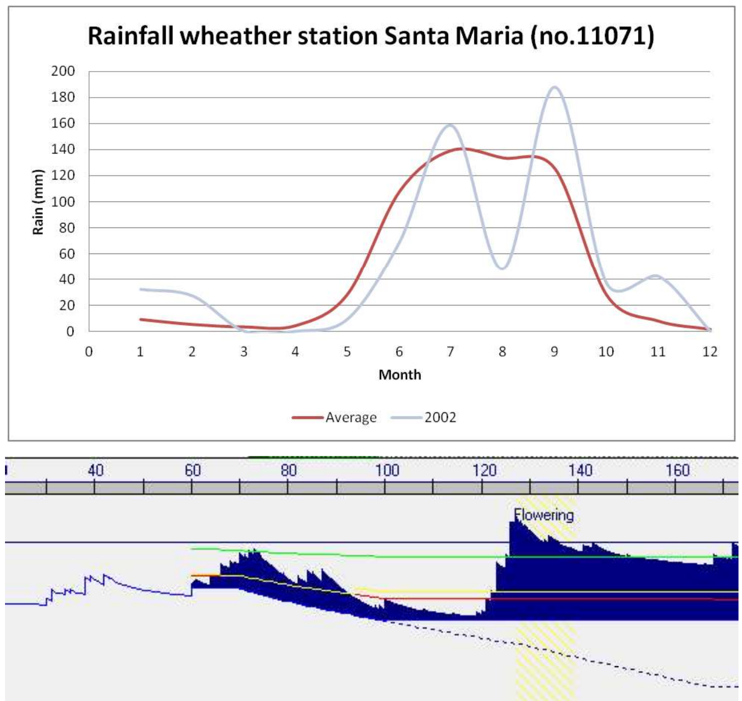


Figure 2.8a/b: Simulation data for C4A on a) precipitation in 2002 (blue) and average precipitation (red) and b) AquaCrop screenshot of simulated soil water content (blue) and threshold for stomatal closure (red line).

a

b



In the Guanajuato scenario B (Table 2.1) this risk was addressed by supplemental irrigation from rainwater harvest. Simulations showed that yield failures and yield reductions were almost completely mitigated with this measure. Years with yield reductions and yield failures were reduced from 4-7 years to only 1 year in the simulated time frame 1999-2009. Average yield was increased from 5.6 - 9.0 T/ha to 10.1 - 10.6 T/ha (Table 2.3).

Table 2.3: Yields and years with yield reductions in scenarios from AquaCrop simulations for 1999-2009.

Case Scenario	Simulated yield (T/ha)	Years with yield reduction >10%	Magnitude of yield reduction (%)[a]
1A	6.3	3	12, 100, 100
2A	6.9	2	16, 100
3A	6.7	1	100
1B	7.8	0	0
2B	7.8	0	0
3B	7.2	0	0
4A	9.0	6	10, 16, 20, 32, 100, 100
5A	5.6	7	18, 29, 100, 100, 100, 100, 100
6A	8.4	4	11, 100, 100, 100
4B	10.6	1 [a]	100 [b]
5B	10.1	1 [a]	100 [b]
6B	10.6	1 [a]	100 [b]

[a] A yield reduction of 100% corresponds a total yield failure.

[b] This is a drought year in 2000 with only 322 mm rainfall between may and November (average is 590mm) which cannot be mitigated by rainwater harvest because it cannot supply the amount of water needed (280mm).

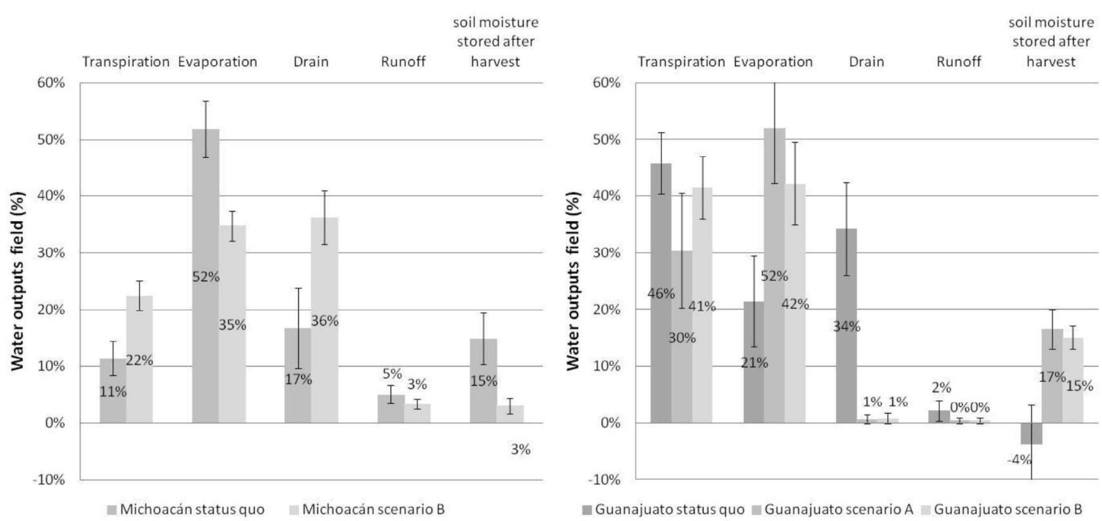
Improvement scenarios: Greenwater use efficiency

In the Michoacán scenarios the total amount of greenwater use per ha remained approximately the same as in the status quo scenario (Figure 2.4). However, a closer look at the field water balance revealed a changed composition of the greenwater footprint (Figure 2.9). The amount of rainfall productively used by the plant increased from 11% in the status quo to 22% in the B scenario. At the same time non-productive soil evaporation was reduced from 52% to 35%. Runoff and drainage did not decrease, the shift to productive plant transpiration came from non-productive soil evaporation. This is interesting because it means that there are no trade-offs with downstream users.

Soil water evaporation could be seen here as an untapped water reservoir that was put into productive use without increasing scarcity for other water users.

In the irrigated Guanajuato case studies unproductive soil evaporation is very low, only 21% of water inputs (rain+irrigation) evaporate unproductively (Figure 2.9). Hence there is no optimization potential. But in scenario A (no-irrigation) unproductive soil evaporation augmented to 52%, comparable to the rainfed systems in Michoacán. This is due to the frequent dry spells which hinder full canopy development. With the management switch from scenario A to scenario B productive plant transpiration increased from 30% to 41% and non-productive plant transpiration decreased from 52% to 42% (Figure 2.9). As in the rainfed scenarios in Michoacán non-productive soil evaporation was shifted to productive plant transpiration.

Figure 2.9: Field water balance in cropping period i.e may to november for rainfed cases (Michoacán and Guanajuato scenarios) and may to april for irrigated cases (Guanajuato status quo). Averaged over AquaCrop simulations from 1999 - 2009.



Improvement scenarios: Water, carbon and land use footprint

The land use footprints of Michoacán case studies were significantly improved by the two management measures. Land use footprints were lowered by 460% (C1), 260% (C2) and 150% (C3) (Figure 2.3). This means that through a change in fertility and soil water management 1.5 – 4.6 times the amount of bioethanol can be produced on the same area as in status quo.

Yield increases are one of the measures identified by WWF, Ecofys and EFPL in their "Low indirect impact biofuel" (LIIB) methodology (Vande Staaij J 2012). Indirect impacts are mainly a result of the displacement of food and feed commodities caused by biofuel production. One of the solutions to reduce these displacement effects are yield increases. Using the methodology of van de Staaij et al (Vande Staaij J 2012) we calculated the amount of LIIB-biofuels that would be produced with an upgrade of rainfed agriculture on the case study fields. This methodology basically calculates the amount of biofuels produced from yield increases above the regional baseline of yield increases (see section 3 suppl.). In scenario 1B around 44 GJ LIIB-biofuels would be produced per ha, 39 GJ/ha in scenario 2B and 20 GJ/ha in scenairo 3B (Table 2.4). The amount of LIIB-biofuels decreases every year because the rising baseline production is substracted.

Table 2.4: Amount of LIIB-maize and LIIB-biofuels produced by an upgrade of rainfed agriculture in scenarios one year and ten years after management shift.

Case	Year	Yield maize (T/ha)	LIIB [a]-maize (T/ha)	LIIB[a]-biofuels (GJ/ha)	Study plot size (ha)	LIIB[a]-biofuels per study plot (MJ)
1B	1	7.8	5.5	44.3	0.4	17.3
1B	10	7.8	5.1	41.4	0.4	16.2
2B	1	7.8	4.9	39.3	0.5	21.2
2B	10	7.8	4.4	35.7	0.5	19.2
3B	1	7.2	2.5	20.0	1.0	20.5
3B	10	7.2	1.7	14.2	1.0	14.5

[a] LIIB=Low indirect impact biofuels. Biofuels with low risk of causing indirect effects.

The land use footprints of the Guanajuato study cases increased in the no-irrigation scenarios (Figure 2.3). Land use footprints almost doubled for C5 (i.e. they increased by 88%) and by around 60% for C4 and 6. Indirect effects can be expected because of the cancelled production.

GHG emissions of the Michoacán B scenarios remained approximately the same per MJ ethanol (Figure 2.6). This is because emissions due to additional inputs applied in the B scenarios are compensated by a higher productivity. The slight decrease for C2B from 40 to 37 g CO_2/MJ and increase for C1B from 40 to 44g CO_2/MJ is due to the fact that nutrient inputs in status quo were slightly above respective below the yield nutrient curve (section 2.2 supp.).

GHG emissions in the Guanajuato B scenarios were significantly reduced (Figure 2.6). Emissions decreased in C4B from 73 to 47 g CO_2/MJ and in C5B from 53 to 44 g CO_2/MJ mainly because energy intensive irrigation from the deep well (200 meter) was ommited. In C6B emissions also decreased, from 54 to 49 g CO_2/MJ. This decrease is less pronounced because irrigation is supplied from a shallow well (15 meter). However, additional emissions from iLUC are possible because the decreased production might lead to an expansion of agricultural land elsewhere. This was not quantified here since this would have required macro-economic modeling.

The WSI-weighted water footprint in the Michoacán B scenario remained zero since green water has a characterization factor of zero (Figure 2.5). The total amount of greenwater use per ha remained approximately the same as in the status quo scenario (Figure 2.4).

The blue water footprint in Guanajuato was significantly improved by a management change from irrigated agriculture to rainfed agriculture plus supplemental irrigation. The WSI-water footprint improved from 25-30 mm/GJ to 2-4 mm/GJ bioethanol (Figure 2.5).

Upscaling

In this section we explore what it would mean at a national level if the measures applied in the scenarios were to be applied to all comparable rainfed and irrigated agriculture in Mexico.

Upscaling Michoacán case studies: In Mexico, the total production volume of rainfed maize was 12.7 mio T in 2012 (Mexican Ministry of Environment (SAGARPA) 2012a). Rainfed maize was cultivated on an area of 6.1 mio ha with an average yield of 1.9 T/ha. Of this area 32% is farmed without any synthetic fertilizer input (SAGARPA-SIAP 2011) and is thus comparable to case 1.

If management changes as in scenario 1B would be implemented on this area total production volume would increase to 24 mio T maize corresponding to 192 GJ bioethanol. Of this 87 GJ would be bioethanol with a low risk of causing indirect impacts (LIIB-bioethanol) which corresponds to around 6% of total Mexican gasoline consumption (International Energy Agency 2011). Note that this amount could be produced without competition with the food sector and without overuse of water resources.

By replacing fossil gasoline with LIIB-bioethanol, around 3.4 mio T CO_2 would be avoided annually in the transport sector (calculated with the average carbon footprint of all B scenarios 45 g CO_2/MJ bioethanol (Figure 2.6).

Through the nationwide upgrade of rainfed agriculture as in C1B a shift of non-productive evaporation to plant transpiration would occur. Around 2900 hm3 of previously unused soil evaporation would be put into productive use. This is around 12% of total water extraction from aquifers in Mexico (Comision Nacional del Agua 2014). This is remarkable given that currently 23% of aquifer water withdrawal is from overexploited aquifers (Comision Nacional del Agua 2014).

Upscaling Guanajuato case studies: An extensification of irrigated agriculture was needed in order to reach sustainable water extraction levels. Around 9.3 mio T of maize are produced from irrigated agriculture in Mexico (30). Of this 4.6 mio T are produced in the dry season in winter (SAGARPA-SIAP 2011). If this second crop production was cancelled as in the Guanajuato B scenarios 4.6 mio T maize less would be produced corresponding to 37 mio GJ less bioethanol.

Combining increases and decreases in production from the upscaling of Michoacán and Guanajuato case studies around 49 mio GJ could be produced. This is around 3.4% of Mexican gasoline consumption. By replacing fossil gasoline with this amount of bioethanol, around 1.9 mio T CO_2 would be avoided annually in the transport sector.

Implementation of the improvement measures as suggested for these two regions would decrease CO_2 emissions per tonne of maize and relieve pressure on Mexican aquifers no matter if the maize would be used in the food, feed or fuel sector.

Note that this is a very rough upscaling. Limitations for the upscaling of Michoacán case studies are that rainfall may not in all areas be enough to support the increased production and soils may not in all areas be adequate for fertiliser input. Only rainfed areas without any fertiliser input were considered in the upscaling (corresponding C1). Areas with low fertiliser inputs might also be adequate for improvement measures (corresponding to C2 and C3). Constraints for the upscaling of Guanajuato case studies were that rainfall may not be enough in all areas to support high-yield rainfed agriculture with rainwater harvest. Aquifers may not be overused in all regions, in these cases extensification would not be necessary from a water resource perspective. Being aware of these limitations, aim of this upscaling is to give an idea of how relevant the studied improvements could be on a national level.

3.5 Discussions

Currently, the trade-offs between water and land use are high in the two studied regions. In the status quo, biofuel production in Michoacán performed better in terms of water and carbon footprint. Based on these two criteria, the recommendation would be to produce biofuels in low yield rainfed systems in the region of Michoacán instead of high yield irrigated systems in Guanajuato. However, the trade off with land use is high. The rainfed case study group required up to 10 times more land. Guanajuato case studies performed well in land use footprint but carbon- and water footprints were high in an aquifer that is already heavily overused. These trade-offs could be reduced in the scenarios by changes in agricultural management.

In the region of Michoacán the land use footprint was improved by 150-460% through an upgrade of rainfed agriculture (mulching + fertility increase), Carbon footprint and water footprint approximately remained at the low values of the status quo. 58-63 MJ/ha biofuels could be produced by an upgrade of rainfed agriculture. Of this, 20-44 MJ/ha were LIIB-biofuels which would not induce indirect land use changes (i.e. no indirect GHG emissions or loss of biodiversity) nor a redirection of crops from the food sector to the energy sector (i.e. no effect on food prizes) because they were from additional biomass. Note that the amount of LIIB-biofuels would decrease every year because the baseline food demand rises. It was interesting that this additional production would not increase water scarcity in terms of water quantity for downstream users because additional water demand of the yield increases was covered by what was previously non-productive soil evaporation. In terms of water quality and eutrophication, effects on downstream users will still need to be evaluated.

In the region of Guanajuato an extensification was necessary to reach sustainable water extraction levels. The trade-off between water and land use could not be avoided completely in the scenarios. By a switch from irrigated double crop agriculture to rainfed agriculture with rainwater harvest the blue water footprint was improved by 90%, the carbon footprint by 9-35%. But this was at the expense of the land use footprint that worsened by 60-88%. Around 55-87 GJ/ha less biofuels would be produced if production was to remain within the boundaries of sustainability i.e. without overuse of the local aquifer.

Interestingly, a rough upscaling indicated that if nationwide irrigated maize production would be extensified and rainfed agriculture was upgraded at the same time, still a certain quantity of biofuels could be produced on top of the current production. These amounts could be produced without causing indirect land use changes, without having negative effects on downstream users in terms of water quantity and including an extensification of highly irrigated maize systems.

The region-specific recommendation for Guanajuato would be to target rainfed systems instead of expanding irrigated high input agriculture. In the study region (valley of Moroleón) three main cropping systems exist depending on their location in the valley: irrigation with water from dam in the surrounding of the dam; irrigation with water from wells where dam water is not available – both high yield systems; and thirdly low yield rainfed maize. In the low yield rainfed system considerably less inputs are used because of the high risk of yield failure (reported by farmer of case 7, one of the cases that had to be omitted because not sufficient data could be obtained from field interview).

In Guanajuato these rainfed low yield systems would offer optimization possibilities without trade-offs between water and land. Rainfall is in general high enough to support a high yielding maize crop but the results show that frequently occurring dry spells need to be mitigated in order to reduce the risk of water related yield losses. These relatively small amounts of water could be supplied by rainwater harvesting and supplemental irrigation without increasing water stress in the aquifer.

The results show that there is more optimization potential in extensive rainfed agriculture than in high input irrigated agriculture. If an increase in production - for the energy or the food sector - was envisaged, the recommendation would be to focus on an upgrade of the low yield rainfed systems and not to expand double crop high irrigation systems.

These results show that increasing yields do not necessarily aggravate water scarcity – even in regions with physical water scarcity - provided that non-productive evaporation can be redirected to productive plant transpiration by management interventions. This opportunity exists primarily for rainfed systems in regions where non-productive soil evaporation is high and where the absolute amount of rainfall is generally high enough to support the crop, but variability of rainfall is the problem.

Interestingly, the rough upscaling indicated that by an upgrade of rainfed maize production around 2900 hm3 of non-productive soil evaporation would be put into productive use. This is around 12% of total water extraction from aquifers in Mexico. From this we conclude that unproductive soil evaporation is an underestimated and under considered water resource which should be considered in water management and in decision support tools like the water footprint

Neither the volumetric water footprint nor the WSI-weighted water footprint gives an indication of this optimization potential (Figure 2.4 and Figure 2.5). In the volumetric water footprint green water is per definition the sum of non-productive evaporation and transpiration, hence a shift from one to another is not notable. The WSI-weighted water footprints assess the environmental impact and typically assign a characterisation factor of zero to green water use. We conclude here that greenwater productivity and non-productive soil evaporation should additionally be considered to reveal potentials for efficiency increases with low environmental impacts. Further research is required developing methodologies to supplement water footprint tools and in mapping those optimization possibilities. To implement these, research is also required in institutional and agronomic implementation barriers in the field like e.g. institutional setup to achieve capacity building for farmers, knowledge on locally adapted agricultural management measures and rainwater harvest technologies (Molden et al. 2007; Garnett et al. 2013).

In summary, we conclude that it is possible to increase maize production in Mexico – for biofuels or to produce more food – without aggravating water scarcity, without indirect land use changes and reducing GHG emissions. This can be achieved by optimizing rainfed low input systems and not by expanding irrigated high input agriculture. This conclusion generally applies to regions where water resources are already overused and where overall rainfall is high enough to support crop growth but variability is the problem. The key to optimize rainfed systems is to increase rainwater productivity by locally adapted management measures. Non-productive soil evaporation is an indicator for this optimization potential. We conclude that this should be considered in the development of tools like the water footprint that are used by policy makers and industry to guide decisions.

4. GREEN WATER OPPORTUNITIES AND LCA

Green water opportunity maps – A complement to water footprint assessment[3]

4.1 Abstract

Irrigation in agriculture is the world´s largest water user, contributing to water scarcity in many regions of the world. At the same time, many agricultural systems use rainwater – called green water – unproductively. Whereas methods exist to find hotspots of blue water use and scarcity there is a lack of methods to identify green water opportunities. The purpose of this work is to develop a method to estimate the efficiency of green water use in rainfed agriculture at regional level. The method is illustrated for the case of maize grown in Mexico. Three maps are produced to show green water opportunities at municipal level. The first map shows unused green water, which is available to increase crop production without increasing water stress for other users. The second map highlights regions with a high share of rainfed systems and with high improvement potential. In these regions, around 37% more rainfed maize could be produced by adequate agricultural measures, which amounts to ~4 million Tons of maize in Mexico. The third map identifies highly water stressed regions where a "green water reservoir" exists that could be used to decrease irrigation in maize production. The step-by-step methodology provided in this paper is useful for companies and policy makers that want to decrease their water footprint. It could complement water footprint methods and regionalized life cycle assessments in order to join the two strategies to approach water scarcity – use less irrigation and make green water use more efficient.

Anna M. Hennecke*, Stephan Pfister, Maria Müller-Lindenlauf, Stefanie Hellweg *(Article in preparation)*

4.2 Introduction

To produce more food with less water is a major challenge of the coming decades. Agriculture accounts for 70% of global water withdrawals and with a growing world population the demand for water and food is expected to rise even further (UNESCO 2012). Because of these withdrawals, water scarcity is already an issue for many regions in the world. For example, in Mexico around 100 of 600 aquifers are overused in a sense that more than 100% of the annual renewable water are withdrawn (Comision Nacional del Agua 2014). The key question is how to use water resources more efficiently, in order to increase yields while delimiting additional stress on water resources.

One solution to this is to improve the efficiency of irrigation to reduce water demand. A number of research activities concentrate on finding hotspots of blue water use and scarcity to inform policy makers and companies where such interventions would be beneficial (Pfister et al 2019.; Stoessel et al. 2012). Another solution might be to improve efficiency of rainwater (green water) (Rockström 2003). Rainwater efficiency is very low in many water scarce countries. In many agricultural systems as little as 5% of the rain is used productively even though around 40-80 % of the rain remains in the soil as soil moisture and could be available for crop growing6. The rest of the soil moisture is lost by unproductive soil evaporation (Molden et al. 2007; Hennecke et al. 2016). There is a need for green water efficiency maps that could complement blue water hotspot maps (Pfister et al. 2011) in order to join the two strategies to approach water scarcity.

The main reasons for low green water productivity in rainfed systems are dry spells - short periods of 3-6 weeks with sub-average rainfall - and a lack of soil fertility. In many regions in Mexico, absolute amount of rainfall is not the limiting factor but the challenge is the variability of the rainfall. A study by Hennecke et al shows for example that in Guanajuato in central Mexico the annual average rainfall of 600 mm is higher than the crop requirement of 450 mm for a high yielding maize crop (Hennecke et al. 2016). However, dry spells occur in 2 out of 3 years in semiarid and dry sub-humid tropical environments (Molden et al. 2007). Effect of these dry spells is that water stress in crucial phases of crop growth decrease yields e.g. during yield formation in the maize grain filling stage or during the vulnerable germination phase. Measures to address dry spells include supplemental irrigation to bridge dry spells. Case studies in central Mexico show that relatively small amounts of water from rainwater harvest (around 10% of the water used for a fully irrigated system) were sufficient to avoid water related yield reductions (Hennecke et al. 2016). The Mexican water development plan mentions supplemental irrigation in its lines of action. However, this topic plays a marginal role in practice, where the primary focus lies on the development of fully irrigated systems (Comision Nacional del Agua 2014). Another measure to address dry spells is to improve soil structure e.g. by returning residues to field, no-till. This increases soil water holding capacity and infiltration rates so that the soil holds more green water for plant growth. Another reason for low green water productivity is a lack of soil fertility i.e. nutrient depletion and loss of organic matter. This leads to high water losses due to soil evaporation because plant density and soil cover is low and cannot retain the water to the soil. In addition, water uptake is poor due to weak roots (Molden et al. 2007). Measures that optimize green water efficiency address these aspects. Improved soil fertility management addresses a lack of soil fertility. This includes adding adequate inputs of organic and synthetic fertilizers and optimizing cropping systems. Among others, mixed cropping systems like the traditional maize and pumpkin mix-crop decrease soil evaporation because pumpkin leaves cover the soil and co-existing roots utilize more green water. These measures improve plant water uptake and decreases soil evaporation. Case studies showed that the amount of rainfall productively used by the plants doubled while soil evaporation decreased accordingly (Hennecke et al. 2016). While measures to improve green water efficiencies are available, there is a need for detailed crop specific green water efficiency indicator that allows mapping regions of opportunity to direct crop specific interventions for improving green water management.

The aim of this study is to propose a method to identify regions of opportunities, where unused green water is available for increasing agricultural production without negative consequences for downstream users and ecosystems, and to illustrate this method with the case of Mexico.

Specifically, in this paper we:

- Propose a method for an indicator that measures the efficiency of the resource green water - a “green water opportunity (GOP)” indicator.
- Make a Green water Opportunity map to answer the question: In which regions green water is used inefficiently in Mexico´s maize fields, i.e. where exists a significant „green water reservoir“?
- Illustrate with the example of maize production in Mexico how such a GOP map can be constructed and used to boost agriculture without additional stress of water resources.

4.3 Materials and Methods

Concept of GOP indicator

As an indicator of how efficient the resource green water is used, we propose the GEFF (Green water EFFiciency) and GOP (Green water Opportunity) indicator:

$$Greenwater\ Efficiency\ (GEFF) = T/P"eff"$$

with

P_{eff} = Effective precipitation in mm. The portion of rain that is agriculturally usable after subtracting drain and runoff, as used in CROPWAT to calculate irrigation demand.

T = Transpiration of green water in mm. This is the portion of rain transpired by the plants.

The GEFF indicator indicates on a scale of 0 to 1, which part of the plant available rainfall the plant used productively. Values close to 0 indicate that very little of the soil moisture is used by the plant. In these cases, unproductive soil evaporation is much higher than productive plant transpiration.

As it is more intuitive that high values show high opportunities, the GEFF indicator is slightly adapted to the GOP indicator:

$$Greenwater\ Opportunity\ (GOP) = 1 - \ GEFF$$

Making a green water opportunity map for Mexico

To illustrate the method and usefulness of a green water opportunity map, we demonstrate the application to a regional case study. We chose Mexico and the crop maize due to (1) the high importance of maize in Mexico, (2) the high potential for green water efficiency gains and (3) data availability from previous research. The map shows the status quo of green water efficiency in existing maize fields in Mexico. Time frame for the calculations was June to October which covers the rainy season where most of the precipitation falls and which is the cropping period of rainfed maize in Mexico. This is a conservative approach. It neglects potential soil moisture from dry season rainfall. The map was generated using the software ArcGIS.

The green water opportunity map was constructed in three steps:

Step 1 P_{eff}: To determine P_{eff} [mm], long-term averaged effective precipitation of the months June to October were summed up (Raster Data Set of Pfister et al. 2011, using two approaches described in CROPWAT). Supporting Information Fig. 1 shows the effective precipitation map for Mexico.

Step 2 T: To determine T [mm], global or country level maps that partition the evaporation and transpiration part of evapotranspiration of maize cultivation are required. These were not available in literature. Therefore, T was estimated as follows:

Step 2.1: Estimation of transpiration coefficient (C): The transpiration coefficient is a measure of the water efficiency of the plant. It refers to how many liters of water the leaf surface transpires to produce 1 kg of yield (dry mass). The transpiration coefficient is crop specific and varies according to climate parameters such as wind speed, air temperature and humidity (Rickman and Sourrel 2014). The potential evapotranspiration (*ETo*) is a commonly used measure for these climate parameters. ETo is the evapotranspiration rate from a grass reference surface, not short of water. It is thus an index for the evaporating power of the atmosphere.

To determine the local values of the transpiration coefficients for maize throughout Mexico we scaled a literature value for the transpiration coefficient of maize (C_{ref} and ETo_{ref}) with the local Mexican climate values (ETo).

$$C_{Maize} = C_{ref} * ETo / ETo_{ref}$$

with C_{ref}= 373 L/kg grain and ETo_{ref} = 235 mm.

C_{ref} was taken from a field study of Al-Kaisi et al (Al-Kaisi et al. 1989). Al-Kaisi et al measured transpiration coefficients of maize in a two-year field experiment in North Dakota to be 373 L/kg grain ETo_{ref} of the growing period May to September (May 23rd to September 18th) at the field site in North Dakota (Latitude: 46.8771900 Longitude: -96.789800) were 235 mm (calculated using monthly long-term average evapotranspiration maps for the period 1961-199010. Supporting Information, Fig. 2 shows the calculated transpiration coefficients for maize for Mexico. Hereby, we assume a linear relationship between Transpiration and ETo. This assumption follows the well-established CROPWAT approach to calculate evapotranspiration (Smith 1992).

Step 2.2: Calculating Transpiration (T_{Maize}) of rainfed maize. Transpiration was calculated by multiplication of the region-specific transpiration coefficient (C_{Maize}) with rainfed yields (Y_{Maize}).

$$T_{Maize} = C_{Maize} * Y_{Maize}$$

Rainfed yields were taken from Mexican Agriculture Information System (SIAP) at municipal level for 2352 municipalities (Mexican Ministry of Environment (SAGARPA) 2012a), transferred to a spatially explicit file format and averaged over a 10- year period from 2005-2014. Supporting Information Fig. 3 shows the calculated Transpiration of rainfed maize cultivation in Mexico. Please note that this approach is a simplified calculation of Transpiration. Transpiration depends not only on climate parameters but also on cropping system, plant density and maize variety (see chapter Discussion).

Step 3: The green water opportunity map was then calculated with

$$Greenwater\ Opportunity = 1 - GEFF"Maize"$$

The resulting map is described in the results chapter.

Identifying hotspots for interventions

GEFF values were classified in four categories: low, medium, high and very high optimization potential (Table 3.1). Comparison with Green water Efficiency Values from literature shows that the classes in this paper are in a similar range but a bit lower (Oweis and Hachum 2001; Rockström 2003). Reason for this might be that GEFF are not crop specific in Oweis et al and Rockström et al and determined by a different approach than in this paper. Please note also that GEFF was calculated as transpiration per total precipitation in the cited literature, while in this paper it is calculated as transpiration per effective precipitation.

Table 3.1. Green water efficiency classes in literature.

Low green water efficiency	Medium green water efficiency	High and very high green water efficiency	Unit	Source
< 0.05	0.5 - 0.10	0.10-1.0	T / P_{eff}	GEFF classification in this paper
0.11	0.22	0.41	T / P_{total}	Hennecke et al
0.1	Not available	0.45-0.55	T / P_{total}	Oweis et al 2001
0.05	0.15 - 0.3		T / P_{total}	Rockström et al 2003

Then, the GOP map was overlaid with other maps to identify regions where interventions would be particularly beneficial.

First, a map showing locations where a lot of rainfed maize (>5% share of area) is grown that has a high and very high GOP improvement potential (<10 % productive use of plant available green water). The resulting map shows target regions to improve yields in existing rainfed maize cultivation by adequate agricultural measures. To do so, from the

GOP map those regions were cut away with less than 5% of rainfed maize (share of area) and less than 0.9 GOP improvement potential. The map of rainfed maize was taken from SIAP and averaged over the years 2004-2015.

Second, regions were identified where irrigated maize was planted (1-10% share of area) and where aquifers were overused (>100% of renewable water taken from aquifers). The map of irrigated maize planted at community level was taken from SIAP (Mexican Ministry of Environment (SAGARPA) 2012a) and averaged 2004-2015. The map shows water of stressed aquifers at state level from Mexican National Water Commission CONAGUA (Mexican national water comission (CONAGUA) 2008). From the GOP map, those regions were cut away where overuse of aquifer is less than 100% and which have less than 0.9 % improvement potential. The resulting map shows target regions where a switch from irrigated to improved rainfed maize cultivation would potentially be feasible.

Potential additional maize production from optimizing green water use

The potential yield increases within the limits given by the availability of green water were assessed. We calculated how much additional maize could be produced if green water efficiency would rise to 15% in all current rainfed maize fields with

$$pot.\,yield\ increase\ [t/ha] = (P"Eff"\ * 0.15 - T"Maize"\ )/C"Maize"$$

Then we calculated total additional maize production with the share of area planted with rainfed maize at level of municipalities and total area of municipalities (Data from CONABIO (Gobierno Federal de Mexico 2011)). For this purpose, potential yield increase of each grid cell was averaged for each municipality.

4.4 Results

<u>Regions of opportunity for improving green water use (GOP map)</u>

The green water opportunity map (Fig. 1) shows the share of green water not productively used by the maize plant in a range from 0% to 100% (0 to 1). Red-orange

regions are „regions of opportunity" in which less than 10% of plant available water were used productively (i.e. more than 90% of the available water in the field unused). Green water productivity is low when a high share of water evaporates unproductively from the soil and when the soils can store a small share of rainwater due to suboptimal soil properties. This can be improved by site-specific agricultural measures like improving soil cover, soil structure and soil fertility and bridging dry spells by water harvesting and supplemental irrigation. Using this green water resource, yields in rainfed agriculture could be increased without a switch to fully irrigated agriculture.
In the yellow-green regions, more than 10% of rainfall was used productively. Here the potential for improving green water use is lower.

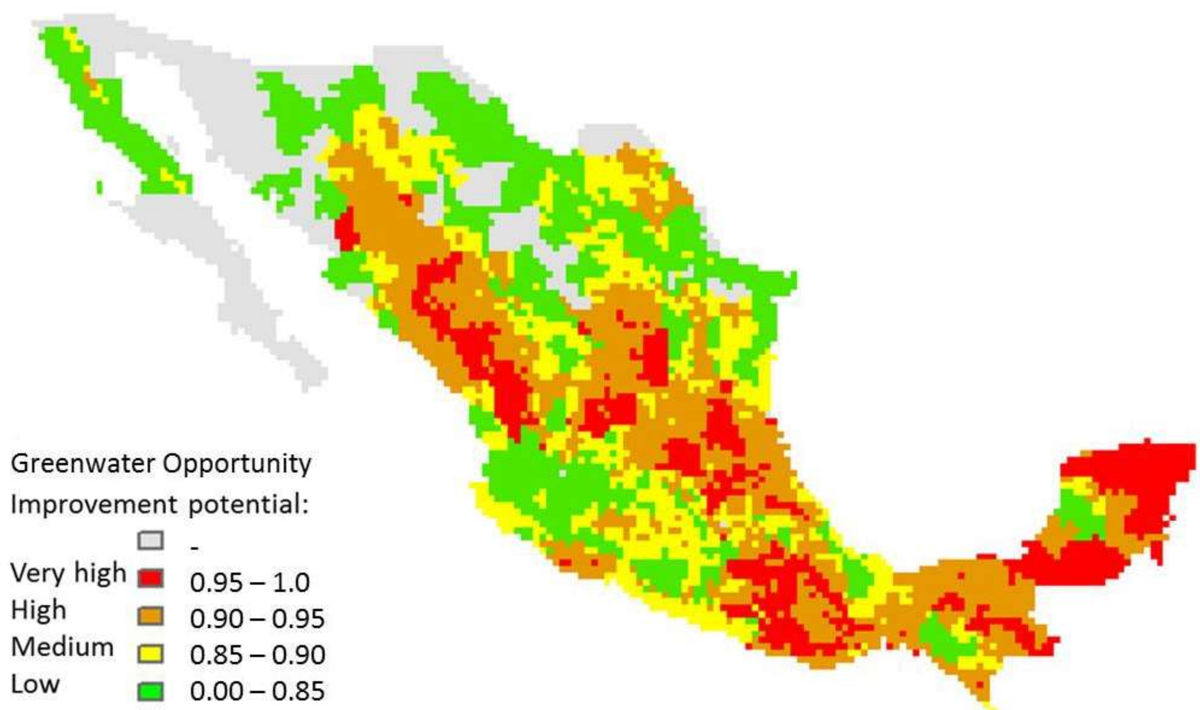


Figure 3.1. Green water Opportunity (GOP) map of rainfed maize cultivation in Mexico.

The GOP map contains only information on the specific yields in a region, but does not contain information on total maize production. If little maize is cultivated in a red-orange "region of opportunity", there is little optimization potential from an overall production perspective. Expanding maize cultivation might of course be an option. However, the scope of this paper is to identify optimization potential without expanding cultivation on additional land. In the northern desert of Mexico hardly any rainfed maize production takes place while in central and south Mexico, a considerable share (in many communities more than 10 % of total area) is planted with rainfed maize (Supporting Information, Fig. 4). Hotspots for improving rainfed maize production include those regions where maize plants cover a considerable share of the area and use only a small share of green water according to the GOP map. The question is: In which regions is a considerable area planted with rainfed maize and only a small share of green water used productively? The overlay of the GOP map with a map of the share of land cultivated with rainfed maize is answering this question (Figure 3.2).

<u>Regions of opportunity for decreasing blue water withdrawals</u>

From the GOP maps above, recommendations can be derived where to improve existing rainfed agriculture. In addition, it can be used to analyze where green water use could be optimized to reduce irrigation in regions with overused aquifers.

Figure 3.3 shows hotspot regions where a) irrigated maize is grown, b) aquifers are overused and c) GOP improvement potential is high. In these regions it would be beneficial (in terms of mitigating aquifer overuse) to reduce irrigation and switch to improved rainfed systems instead (maintaining the production level of one crop per season while reducing irrigation). In these regions a “green water reservoir” exists that could be used for production instead of tapping the “blue water reservoir”, i.e. the aquifer, which leads to water stress impacts.

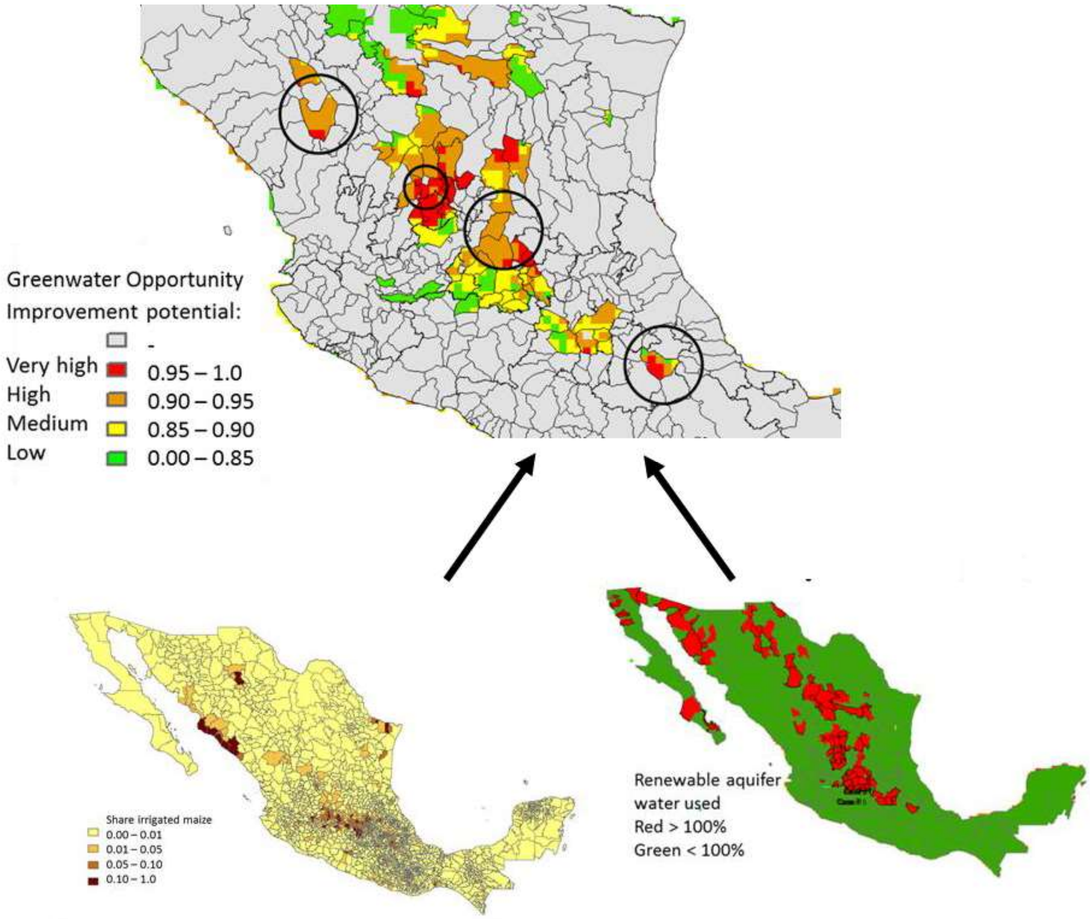


Figure 3.3. Hotspot regions for improving green water efficiency in rainfed maize production for reducing irrigation in maize production in water stressed areas. Circles in large map show regions where aquifers are overused, irrigated maize covers 1-10% of the area and rainfed improvement potential is high to very high. Small maps: Irrigated maize production and overused aquifers.

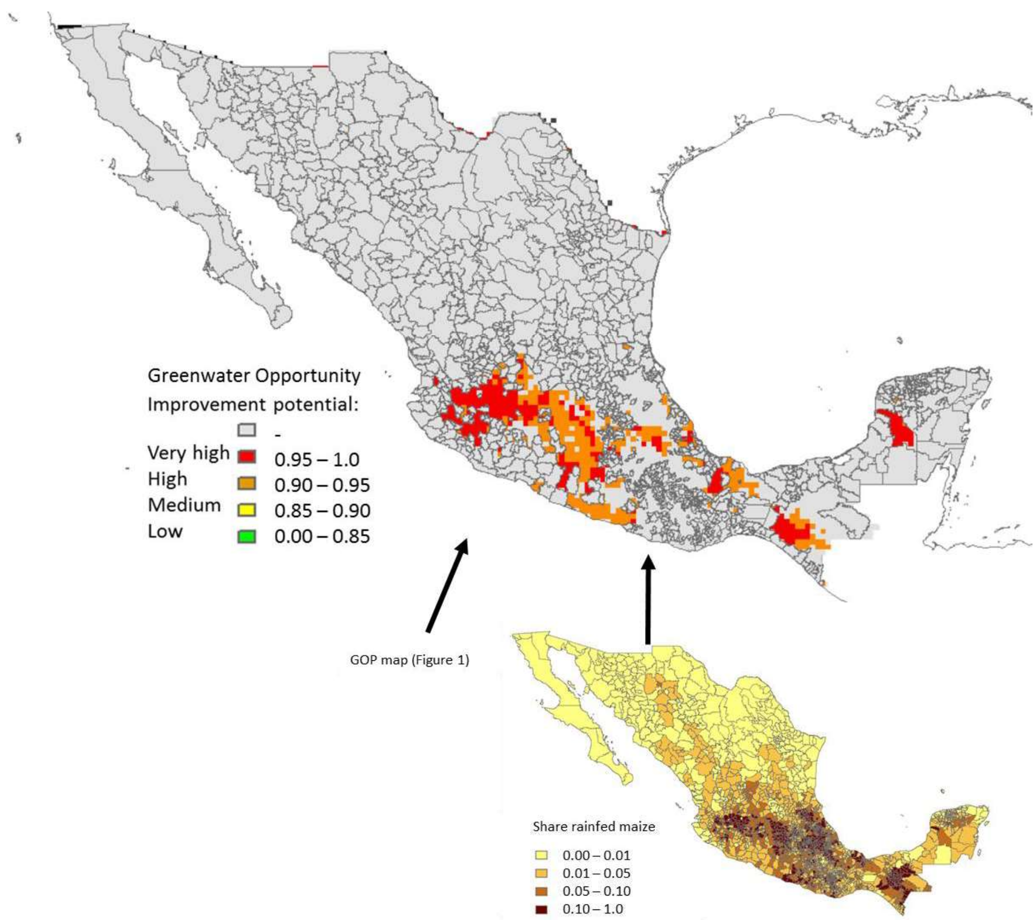


Figure 3.2. Hotspot regions for improving rainfed maize production where a considerable share of land (>5%) is planted with rainfed maize and green water is not used efficiently (large map). Small map: Rainfed maize production as share of area.

If green water use would be increased to 15% (GEFF = 0.15) by adequate agricultural measures, a total of around 4 million Tons of maize could be produced additionally in current rainfed maize production – without irrigation and without expanding cultivated area. This is around 37% of total production in rainfed areas. Figure 3.6 in Supporting Information shows the potential yield increases in t per ha.

4.5 Discussions

<u>Plausibility and constraints</u>

The GOP indicator is applicable for regions where rainfall is scarce or just sufficient to support plant growth. In regions with abundant precipitation, GOP values might show optimization potentials where none exist. A high value for Peff in denominator of the GOP equation leads to low efficiency values even if transpiration in the nominator is at its physiological maximum. This restricts the application of the GOP indicator to regions where rainfall is limited in terms of just sufficient to support plant growth. However, as the goal of the GOP indicator to find optimization potentials in regions where water scarcity is an issue, this limitation is not important for our case study of maize in Mexico.

The calculation of the transpiration was checked for plausibility by comparing selected values with two field case study values from Hennecke et al (Hennecke et al. 2016). For low yields, the GOP approach underestimates the transpiration coefficient by around 25%. For high yields, it overestimates the transpiration coefficient by around 25% (Supporting Information, Fig. B.5). In a certain yield range, the transpiration coefficient is identical (Supporting Information, Fig. B.5). Reason for the deviation between the methods is that the transpiration coefficient does not only depend on climate parameters, but also on plant density. Plants on a sparely populated maize field transpire more water per grain yield than a densely populated high yielding maize field. The simplified approach hence includes also the potential arising from improving the transpiration coefficient and is therefore adequate to detect optimization potentials.

Application of GOP indicator

The GOP map approach identified several regions in central Mexico where green water is used inefficiently in Maize production. These are regions of opportunity where a significant “green water reservoir” exists, which could be used to increase production and to relief overused aquifers, rivers and lakes elsewhere. If Greenwater Efficiency would be increased to a high efficiency, current maize production in Mexico could be increased by 4 million Tons which is 37% of total domestic rainfed production. While our analysis assumed a high efficiency, optimal efficiency would result in even higher production. This is in line with literature results, e.g. Mueller et al. find that only 20-40% of attainable cereal yields are achieved in many regions in Mexico and that maize has most potential for yield increases in Latin America compared to other cereals (Mueller et al. 2012b). Hennecke et al. find higher potential increases of total maize production of around 90%. However they do not take into account water restrictions (Hennecke et al. 2016). Thus, our estimate is rather conservative and even more output could be achieved from green water efficiency gains.

The GOP map is useful to give recommendations to policy makers that aim to increase domestic food, fiber and bioenergy production. For example, Mexico is not self-sufficient in Maize production. Around 8.2 million tons of maize crop were imported (compared to 21 million tons domestic production) (FAO 2017), mostly from the US. Discussions in 2017 on an overhaul of the NAFTA trade agreement fueled plans of the Mexican government to increase domestic production (New York Times 2017). The GOP map shows where maize production could be increased without increasing pressure on limited water resources, which is already high from a global perspective (Boulay et al. 2018). In addition, management of local green water resources has been proposed as a good option to increase resilience of agriculture to climate change, because it reduces risks for dry spells and agricultural droughts which many regions increasingly face (Rockström et al. 2009).

Does an increased green water use affect runoff and groundwater recharge and thus have an impact on downstream users and environment? In the LCA context, there are few studies on environmental impacts of green water consumption. Nunez et al.

compute on a global scale total evapotranspiration (green water) losses from natural vegetation (Núñez et al. 2013b). An assessment of potential environmental impacts was not conducted, since other studies assume that there is generally no effect on blue water availability expected (e.g. (Ridoutt and Pfister 2010)) and thus it is more a problem related to land use in case soil moisture is consumed (in the case of rainwater harvesting). Additionally, the seasonal green water availability should be accounted for, since soil moisture can be stored and thus depleted over a cropping period. Crop rotations might be looked at for this purpose as done by Nunez et al (Núñez et al. 2013a). Case studies for the crop maize show that the shift caused by increasing green water use is mainly from unproductive soil evaporation – this is interesting because this means that water availability for downstream users will not decrease (Hennecke et al. 2016) and significantly increase use of stored soil moisture. However, the detailed effects should be further investigated in the local context.

Outlook

Global blue water maps and regionalized water footprints show hotspots for blue water use. The proposed GOP map is a complement to these maps and water footprints because it shows where green water reservoirs exist that could be used for increasing production. The next step would be to upscale the GOP to the global level and to further crops. This can be done by following the steps described in the Methods section. Global and crop-specific data to calculate GOP are in general available, in particular data on effective precipitation exist at global level and crop specific transpiration coefficients have been determined for many crops. One constraint could be data on rainfed yields with sufficient temporal and spatial resolution. Mexico has excellent agricultural statistics based on survey data. The availability of these statistics at global or country level scale would need to be assessed. A global assessment would also have to take into account that the GOP indicator detects green water optimization potentials particularly in regions with highly variable rainfall, where not absolute amount of rainfall is the challenge but its variability.

Soil profile of maize cropping system in Mexico (left). Fully irrigated maize cropping system in Mexico (right, top). Determination of soil properties for Model parametrization (right, below).

5. CONCLUSIONS AND OUTLOOK

5.1 Discussion and Conclusions

To produce more crop per drop, with less GHG emissions and without expanding agricultural land is the challenge of the coming decades. This dissertation aims to contribute to the development of policies and scientific methodologies to support this. The LCA methodology is the central theme that runs through the dissertation: First, an agricultural policy is assessed that includes a practical application of LCA. Second, the LCA is applied to a case study on sustainable agricultural intensification in Mexico. And third, a complement to LCA methodology is developed in order to enhance its usefulness in identifying opportunities for sustainable agricultural intensification.

<u>LCA in policies</u>

There are many policy instruments to govern and guide a nation´s agriculture. This thesis looked at one policy that integrated Life Cycle Assessment in a policy, namely the European Renewable Energy Directive.

The thesis shows that it is feasible to use LCA in a policy, however there are shortcomings in the current version of the EU Renewable Energy Directive **(Chapter 2)**. The findings make clear that there is a regulatory gap in defining standard tools to quantify GHG balances. This regulatory gap needs to be closed. Otherwise, producers may be able to improve their GHG balance by choosing a different tool and not by actual improvements in the process. A solution to this would be to make a certain tool mandatory or include a set of additional rules as e.g. proposed by the BioGrace project. The findings of this thesis are of high relevance for national authorities that are in charge of implementing the EU renewable energy directive and in general for policy makers that would like to make a policy of this kind (LCA-policy connection).

A more general conclusion is that even though LCA is a well-established and reliable method for scientific studies, it is tricky as a policy requirement, because producers need to consider and follow many methodological details in order to get harmonized results.

This requires a high level of detail and effort by producers and by national authorities that are in charge of checking the calculations. To make sure, that this effort will not be a barrier for small producers and efforts in checking the calculations are unreasonable, I recommend considering alternatives to requiring a LCA in a policy. For example, LCA could be used prior to policy making to identify key leverage points of the GHG balance – e.g. biomass residue burning in the sugarcane ethanol pathway – and calculate a standard value for these pathways. Then the policy could require producers to give proof of which production practice they apply and then use the corresponding standard value. The current version of the policy does include standard values, but they only differentiate crops and not production modes (European Parliament and Council 2009a).

From the analysis made in this dissertation I further conclude that the scope of the EU RED policy is very narrow. Land use and water are not considered. These are two important aspects of sustainable agricultural intensification. In order to reach the goal of sustainably produced biofuels I suggest including these aspects. In order to avoid displacement effects from additional agricultural production, it could be required that production comes from agricultural intensification. This has also been proposed by Ecofy with their iLUC approach (Jasper van de Staaij 2012). The thesis shows how improvement potentials can be detected with LCA at case study level **(Chapter 3)**

<u>LCA Case Study on sustainable agricultural intensification in Mexican Maize Production</u>

The dissertation presents the first site-specific LCA results for the greenhouse gas, water and land use impact for agricultural production (**Chapter 3**) in Mexico and models scenarios for sustainable agricultural intensification in 12 case studies and nationwide. While environmental footprints had been available at generic level (Stoessel et al. 2012), environmental footprints for agricultural products are very site-specific and existed only for Mexican maize production for the GHG impact (García et al. 2011). This thesis is the first that provides a site-specific LCA for individual maize case studies in Mexico covering a broad range of impact categories (water scarcity, land use, GHG).

The findings are relevant for farmers in water scarce regions that rely on rainfed cropping and would like to increase their production as well as farmers that rely on irrigation in water scarce areas. The thesis shows how sustainable agricultural intensification can be achieved with particular focus on water use **(Chapter 3).** The thesis shows where agriculture in a region could be transformed to a mix of extensified irrigated farming and intensified rainfed farming to help relieve water stress while increasing overall production. The main conclusion is that there are significant green water opportunities for farmers in many regions. Farmers could increase their production by using green water more efficiently. This "green water reservoir" can be tapped by adequate site-specific measures like improving soil cover, soil structure and soil fertility and bridging dry spells by water harvesting and supplemental irrigation **(Chapter 3)**.

The findings are also relevant for policy makers making national strategies for sustainable agriculture and water management. The main conclusion is that at national level sustainable agricultural intensification can be achieved by a mix of extensifying high input irrigated agriculture and improved management of low yielding rainfed agriculture. To identify regions for targeted interventions, policy makers can use the step-by-step methodology to create GOP maps (developed in chapter 4).

The current strategic water vision for Mexico only includes measures concerning blue water use but hardly mentions greenwater efficiency (Comision Nacional del Agua 2014). The findings of this thesis suggest that both approaches should be combined. It quantifies with the example of case studies and an upscaling to national level to what magnitude maize production could be increased in Mexico while decreasing blue water use at the same time (**Chapter 3**).

<u>Pilot application of FAO´s AquaCrop Tool</u>

This thesis provides one of the first studies that used the FAO AquaCrop tool in an LCA (AquaCrop publications) (**Chapter 4**). The FAO tool allows very detailed calculations of the field water balance. Finding blue water hotspots is also possible with the older FAO tool CROPWAT (Smith 1992). However, the new FAO AquaCrop tool provides also the opportunity to find hotspots of low rainwater efficiency that is caused by non-productive soil evaporation as well as simulating agricultural measures to improve rainwater efficiency. This thesis shows how the tool can be used to identify green water opportunities. For this thesis soil sampling and field interviews were conducted providing a high-quality parametrization of the tool. Actual data like e.g. the initial soil water content of the soil are crucial for meaningful results. This effort needed to obtain this data may limit the use of the AQUACROP tool for standard LCA studies.

Rainwater efficiency indicator

One main scientific contribution of this thesis is that I developed a Greenwater Efficiency Indicator, which is a complement to current LCA impact assessments on water use. From the current state of water impact assessment LCA can detect weak points: in which regions of an assessed agricultural system are hotspots of blue water use? Where does it “hurt”? They do not provide an answer to the question: what can we do? In this thesis I developed a method to assess green water opportunities at level of individual producers and at regional level (**Chapter 3+4**). This greenwater opportunity assessment shows where green water could be used more efficiently by adequate agricultural measures, to increase production while using less scarce blue water. This is interesting for farmers, policy makers or companies that produce in water scarce regions to know at a regional level where rainwater efficiency can be increased to increase agricultural production. The thesis provides an operational method and step-by-step guidance to find greenwater opportunities at national or global level and for different crops (**Chapter 4**).

5.2 Limitations and Outlook

One important limitation of this thesis is its focus on water scarcity in terms of water quantity. However, water pollution is an important aspect that can contribute to water scarcity as well. Agricultural intensification may be associated with an increased used of fertilizers and pesticides which is likely to have an impact on water quality. A consideration of the grey water component would be a next step.

Greenwater opportunities are only assessed for current and historical climate conditions. Rainfall patterns and temperatures will change due to climate change. A next step would be to elaborate Greenwater Opportunity Maps based on climate change scenarios. For the use of the GOP method in political decision making and development of strategic water plans, future climate conditions need to be considered in order to make the recommendations “climate-proof”.

There are multiple dimensions of the yield gap problem. Green water is only one input to cropping and there are a wide range of social and economic aspects to taking action to close yield gaps. This includes nutrient and fertilizer management, building capacities in farmers for sustainable agricultural management, balancing the economics of intensification and extensification, creating political incentives and framework conditions. These aspects are out of scope of this thesis.

While some parts of the thesis, like the methodology to develop Greenwater Opportunity Maps are readily applicable to different regions and scales, more research is needed to do so at farm level. For the time-consuming collection of field and soil data for LCA inventories at field level, the development of remote sensing might bring faster and yet precise enough alternatives.

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ACKNOWLEDGEMENT

First and foremost, I have to thank my supervisor, Prof. Stefanie Hellweg. Without her invaluable guidance, positive attitude and patience this thesis would have never been accomplished. I would like to thank you very much for your support and understanding over the past years.
I would also like to thank Stefan Pfister for his invaluable support with modelling and careful review of the paper manuscripts, and to Prof. Maria Müller-Lindenlauf. I discussed early versions of the thesis and papers with her and her incredible knowledge on the agricultural sector as well as her structured thinking helped me a lot. It was a pleasure having you as co-authors.

Getting through my dissertation required more than academic support, and I thank my friend Dr. Alexandra Koethe for helpful discussions and wonderful coffee breaks and my fellow ZEF students for enjoyable classes and activities.
During the PhD, I went to the National Autonomous University of Mexico (UNAM) for several months to gather field data. Many thanks to my Mexican crew, Carlos Garcia, Alfredo Fuentes and Enrique Riegelhaupt for our field trips and data gathering. Working with you was an extraordinary experience. I must also thank Manuela Prehn Junquera for logistic support in Mexico and for Spanish classes. As well as to the Mexican farmers from our case studies for providing data, conducting interviews and letting us dig holes in their fields.

I would also like to show gratitude to IFEU (Institute for Energy and Environmental Research in Heidelberg) for funding the PhD thesis. In particular to Jürgen Giegrich and Horst Fehrenbach for finding out-of-the-box solutions for funding, combining job and travel for the PhD and paper ideas. Thanks to John Neeft and all members of the EU BioGrace Project that I could be part of this interesting initiative between science and policy.

I would also like to thank Philipp Schukat from the GIZ department on social and ecological standards. His enthusiasm for the topic made a strong impression on me and he got me interested in the topic of sustainable agriculture in developing countries which set the direction for my further career path.

Finishing the thesis was a challenge. Combining my full-time job at IFEU and later at GIZ and my family with two children has often let the thesis taken a back seat. I´m grateful to Prof. Lisa Freudenberger to accompany the last steps with her encouragement and valuable comments and to Prof. Borgemeister for his help in finding the right department and co-supervisor to get the PhD over the final mile.

Most importantly, none of this could have happened without my friends and family. I cannot express my gratitude and appreciation for their friendship and creating a loving and caring environment which has been the basis for my work. This work is dedicated to my children Emma and Paulina.

APPENDICES

APPENDIX 1: SUPPLEMENTARY INFORMATION FOR CHAPTER 2

1. AQUACROP PARAMETRIZATION

1.1 Climate data

Climate data on daily minimum and maximum temperature Tmin (°C), Tmax (°C) and total rain (mm) were taken from the Mexican Meterological Service (Mexican Meterological Service). Data was entered in the climate module in a daily resolution for the years 1999-2009. Data was taken from the climate station located closest to the case study field. For case 4, 5 and 6 this was climate station No. 11071 (Santa Maria). For case 1, 2 and 3 this was climate station No. 16087 (Patzcuaro). Data on potential evapotranspiration Eto (mm/day) was obtained from CLIMWAT (FAO 2006) as long term monthly averages because Eto data is not available from Mexican Meterological Service.

1.2 Crop parameters

Crop parameters for maize were partly acquired by field interviews with farmers, from literature data or other considerations (Table A.1). For all other crop parameters AquaCrop *Default Maize (Davis)* was applied.

Table A.1: AquaCrop parameters in crop module and field module.

Case	Crop	Planting date (m/d)	Fert. (%)	CC_o (plants/ha)	CC_x (%)	M_{off} (%)	Z_x (m)	Z_{exp} (cm/day)	RO	SB (m)
1	Maize	jun 10th[a]	23[b]	51,000[a]	70[a]	0[a]	0.9	0.7	yes	0
2	Maize	jun 10th[a]	25[b]	76,000[a]	90[a]	0[a]	0.9	0.7	yes	0
3	Maize	jun 22nd[a]	50[b]	63,000[a]	80[a]	0[a]	0.9	0.7	yes	0
4	Maize	may 5th[a]	85[b]	100,000[e]	96[a]	0[a]	0.9	2.0	no	0.2
4	Barley	dec 1st[a]	100[b]	2,000,000[a]	95[f]	0[a]	0.9	1.4	no	0.2
5	Maize	jun 30th[a]	80[b]	103,000[a]	96[a]	0[a]	0.9	0.7	yes	0
5	Maize	feb 5th[a]	80[b]	103,000[d]	96[a]	0[a]	0.9	0.7	no	0
6	Maize	may 12th[a]	85[b]	98,000[a]	96[a]	0[a]	0.9	2.0	no	0.2
6	Barley	dec 1st[a]	100[b]	2,000,000[a]	95[f]	0[a]	0.9	1.4	no	0.2

DAP: Day 1 after sowing ; Fert. : Soil fertility; CCo: Plant density; CCx: Maximum canopy cover; Moff: Mulches off-season; Zx: Maximum effective rooting depth; Zexp: Average root zone expansion; RO: Surface runoff occurance; SB – Height of soil bunds;

[a] Field interview with farmer;

[b] Estimation (see suppl. Section 1.3 Soil fertlity);

[c] inferred from plant density (expert judgement on relationship between plant density and maximum canopy cover);

[d] Estimation (see suppl. Section 1.6 Parameter estimations for scenarios);

[e] estimated to be as in case 6 and 5;

[f] based on Abhra et al (Abrha et al. 2012)

Planting date (DAP) and plant or seed density (CCo) was aquired per interview with field owner. Maximum canopy cover (CCx) was inferred from seeding density or row and plant spacing by expert judgement on relationship between plant density and maximum canopy cover.

Maximum rooting depth (Z_x) measured in the field varies between 0.83 m and 1.7 m. In AquaCrop, maximum rooting depth was set to 0.9m for all cases to adjust for an artefact of AquaCrop calculation routine which is as follows: In the test runs of AquaCrop rooting depth was set to the default of 2.30 m. The roots are distributed over the whole rooting depth. Since only the top of the root zone is wetted in the simulation (this is the typical situation at the beginning of the rainy season in Mexico), the crop can only extract water in that wetted part, but there are not enough roots to extract since the rooting system develops anyway over the whole 2.30 meter. This leads to high water stress in the simulation which is an artefact of the AquaCrop simulation routine. According to the AquaCrop developers, AquaCrop should not have allowed to develop the root zone that deep when there is insufficient water as in the simulation (Personal communication with AquaCrop Developer Dirk Raes). This should be adjusted automatically in the simulation procedure. However, this is currently not implemented in AquaCrop. Therefore in order to avoid this artefact, rooting depth was set to 0.9m and root expansion rate was reduced to 0.7cm/day (instead of default 2cm/day). For irrigated systems root expansion rate was kept at 2 cm/day. This is plausible because in contrast to the rainfed cases the soil is wetted through irrigation applied before seeding. Therefore root expansion would not be hempered by dry soils as in rainfed cases.

AquaCrop predicts in some cases more aeration stress due to too much water in the root zone – which is probably not the case in reality where roots are spread out over more than 0.9 m. To avoid this artefact, the default for the threshold for aeration stress was set from saturation -5% to saturation-1%. AquaCrop simulations may react more sensitive to dry spells during or at the end of the cropping season because roots can not make use of water in deeper soils. A sensitivity analysis (deeper rooted plant) of the soil depth is not possible because when soil depth is increased, higher water stress in the expansion phase is simulated – i.e. the artefact described above. However, a rooting depth of 0.9 m is in a realistic range for the region of Guanajuato (soil depths of Case 4, 5 ,6 0.83m, 1.4m and 1.08m).

Crop parameters for barley were based on Abhra et al cultivar *Barley Italy/Ponente.* This long yielding cultivar matched the cropping duration reported by farmers.

Table A.2: AquaCrop parameters for barley.

			Time to reach:					
	CGC ($\%°C^{-1}d^{-1}$)	CDC ($\%°C^{-1}d^{-1}$)	Emergence (GDD)	Max canopy (GDD)	Flowering (duration) (GDD)	Senescence (GDD)	Maturity (GDD)	HI0 (%)
Barley	0.670	0.600	100-226	893	940 (180)	1063	1520	52

CGC: canopy growth coefficient; CDC canopy decline coefficient; GDD: Growing degree days; HI: Harvest Index

1.3 Soil fertility

Limited soil fertility might hamper canopy development, result in a steady decline of canopy in mid season, and reduce yields. To account for the effect of soil fertility stress in simulations, the case studies were categorized in a soil fertility category in the "Field management module".

The soil fertility level was set, so that in a good rainy year the yield value reported by the farmer was simulated. I.e. first good rainy years were identified (for C1 this is e.g. 2001) and then the soil fertility level was adjusted in AquaCrop. This method results in a ranking of cases by fertility level which roughly coincides with the ranking of cases by N-input (Table A.3). The three low input cases for maize are at the lower end of the ranking, the three high input cases at the upper end. Differences within the group can be explained by the fact that N inputs are an important but not the only limiting factor for yields. Inputs of P_2O_5 and other nutrients as well as pests also determine yields. For the purpose of this study the accuracy of this soil fertility classification is sufficient.

Table A.3: Ranking of cases by AquaCrop soil fertility level and by fertilitser input reported by farmers.

Case	Ranking AquaCrop Fertility level (%)	Ranking Fertiliser input (kg N $ha^{-1}yr^{-1}$)
1	23	20
2	25	102
3	50	104
4	85	475
5	80	348
6	85	494

1.4 Soil profile

Soil sampling was conducted beginning of may 2013 at the end of the dry season. In each case study plot one soil profile was excavated to determine horizon depth, texture, bulk density and soil moisture. Texture of each soil horizon was determined by fingerprobe according to Siebe et al (Siebe et al. 2006). To determine bulk density and volumetric soil water content two samples were taken from each horizon with a cylinder (diameter: 4.9cm, height: 5.4 cm). Weight of fresh samples was determined and samples were dried 24h at 105°C. Then oven dry weight was determined. From this volumetric soil water content and bulk density were calculated.

To determine N, P and organic carbon content, soil samples were taken in 40 measurement points per plot at a depth of 0-30 cm with a Pürckhauer bohrer. Samples were aggregated to a mixed sample and N, P and organic carbon content measured in the laboratory (Table A.4).

Wilting point (PWP), field capacity (FC), total pore volume (SAT) and saturated hydraulic conductivity (ksat) were inferred from texture, bulk density and organic matter content as described in the “Manual para la descripción y evaluación ecológica de suelos en el campo” by Siebe et al (Siebe et al. 2006) (*Table A.5*). These were used in the AquaCrop soil module. Soil water content for AquaCrop initial conditions module are shown in Tables A.1 and A.2.

Note that initial soil moisture in simulations was based on measurements of just one year (2013). Soil moisture at the beginning of the season varies from year to year. In order to get a reliable result for the concrete plot soil moisture would need to be sampled each year and then the simulation would need to be done with the climate data of the corresponding year. In addition, the simulation does not include the effect of the second crop (vetch) of C4 on the initial soil water content. For the purpose of this study we consider this approach accurate enough because the goal is not to give recommendations for the concrete plot but to show general tendencies.

Table A.4: Soil organic matter content

Case	Organic matter (%)
1	4,21
2	3,23
3	1,90
4	2,74
5	3,43
6	2,97

Table A.5: Soil profile caracteristics of case studies. PWP: permantent wilting point; FC: field capacity; SAT: total pore volume; Ksat: saturated hydraulic conductivity

Case/ Horizon	Texture	Thickness (m)	PWP (%)	FC (%)	SAT (%)	Ksat (mm/day)
1/1	Silt loam	0.48	16	43.5	53	100
1/2	Sandy loam	0.29	10	34	45	500
1/3	Sandy clay loam	0.43	30	45	49	100
1/4	Silty clay	0.4	36	51	55	300
1/5	Silt loam	3.0	14	41	50	100
2/1	Silt loam	0.74	16	43,5	53	100
2/2	Loam	0.39	20	40	48	300
2/3	Clay loam	0.14	29	45	50	300
2/4	Silt loam	0.18	14	41	50	100
2/5	Silt loam /Silty clay loam	2.3	18	41,5	49,5	100
3/1	Clay loam	0.27	31	47.5	53	300
3/2	Clay loam	0.24	28	40.5	45	40
3/3	Loam	0.19	18	33	40	100
3/4	Sandy loam	0.1	9	29	38	40
3/5	Silty clay loam	0.2	20	36	42	40
4/1	Silty clay loam	0.24	22.5	40	47.5	40
4/2	Silty clay	0.16	35.5	53	58.5	300
4/3	Silty clay	0.28	34.5	50.5	55	300
4/4	Silty clay loam /Silt loam fine	0.15	18	41.5	49.5	100
5/1	Silty clay	0.2	30.5	44	49.5	40
5/2	Silty clay	0.7	30.5	44	49.5	40
5/3	Silty clay loam	1.1	28	44.5	52	100
5/4	Silty clay loam	1.4	26	42	49	100
6/1	Silty clay loam-clay loam	0.27	25.5	41	47.5	300
6/2	Clay loam	0.15	31.5	49	55	300
6/3	Clay loam	0.26	29	45	50	300
6/4	Silt loam fine	0.4	14	41	50	100

Figure A.1: Soil moisture before start of the rainy season for low input cases status quo and scenario B.

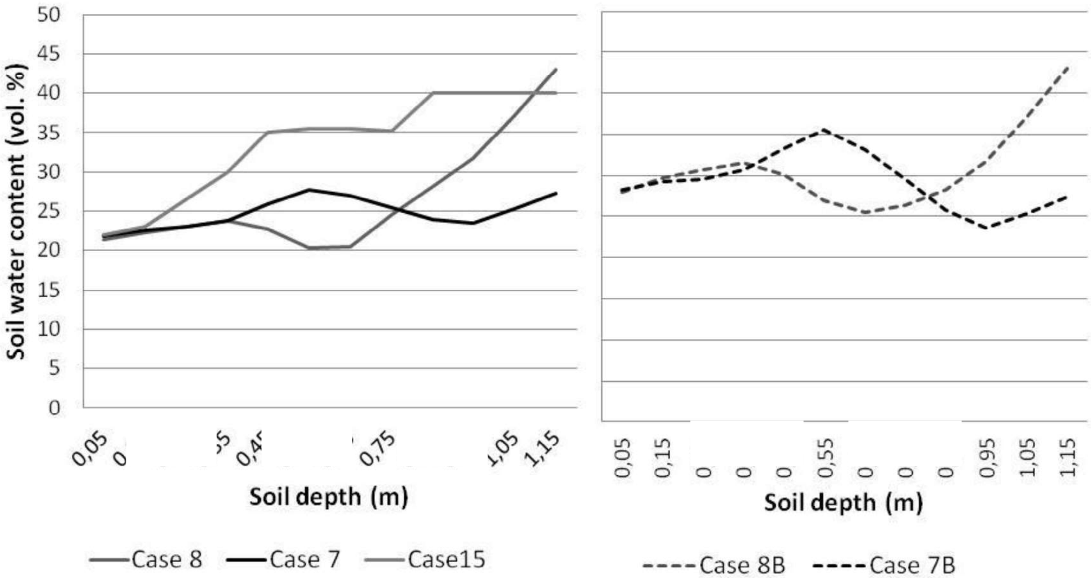
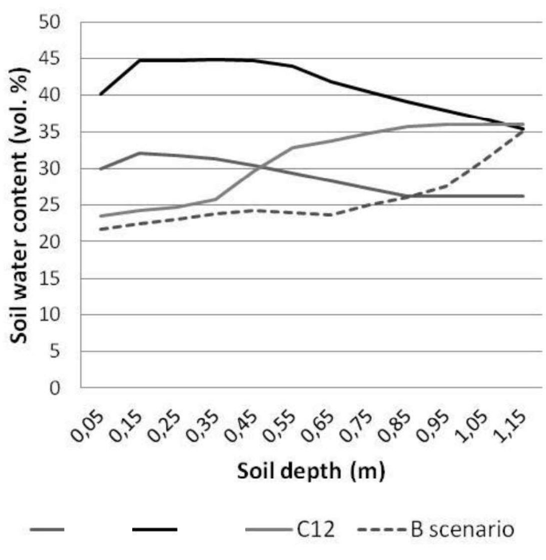


Figure A.2: Soil moisture before start of the rainy season for high input cases status quo and scenario B.



1.5 Irrigation rates and dates

Irrigation was applied in case 4, 5 and 6. Water was pumped from a water well and applied by furrow irrigation. Application dates and rates reported by farmers are shown in Table A.6.

Table A.6: Irrigation rates and dates for status quo cases.

Case	Crop	Irrigation date	Net application (mm)
4[a]	Maize	May 2nd	223
4[a]	Maize	Jun 4th	119
4[b]	Barley	Dec 28th	119
4[b]	Barley	Feb 6th	223
4[b]	Barley	Mar 10th	223
5[a]	Maize	Feb 5th-7th	124
5[a]	Maize	Mar 10-11th	93
5[a]	Maize	Mar 25-26th	93
5[a]	Maize	Apr 9-10th	93
5[a]	Maize	Apr 24-25th	93
6[a]	Maize	Apr 27	250
6[a]	Maize	Jun 2nd	150
6[b]	Barley	Dec 1st	250
6[b]	Barley	Jan 19th	150
6[b]	Barley	Feb 18th	150

[a] First crop; [b] Second crop

1.6 Parameter estimations for scenarios

Based on the outcomes of the status quo simulations, scenarios with an improved management were selected. AquaCrop modules were modified as follows:

Michoacán scenario A: In scenario A it was assumed that the farmer increases soil fertility by adequate measures (e.g. green manure, more synthetic fertilizers) and increases plant density to intensify production. These measures can not be simulated directly in AquaCrop. To include the effect of increased soil fertility, fertility levels were increased in the AquaCrop soil fertility module from 23% (C1 status quo), 25% (C2 status quo) and 50% (C3 status quo) to 60%. Planting density was increased in the AquaCrop

crop module from 51.000 plants/ha with 70% cover (C1) 76.000 plants/ha with 90% cover (C2) and 63.000 plants/ha with 80% cover (C3) to 76.000 plants/ha and to 90% maximum canopy cover.

Michoacán scenario B: Scenario B included the measures taken in scenario A and additionally an 80% mulch cover before seeding to increase the initial soil water content. The mulch cover conserved the soil moisture better in the soil from the rainfall before seeding. This measure was chosen to avoid yield failures that occurred due to water shortage in the beginning of the cropping season. It was simulated in AquaCrop by including 80% organic mulch cover in the off-season module.

For C3 the mulch cover was sufficient to avoid the yield failures of the status quo scenario. For C1 and C2 sensitivity analysis have shown that applying a mulch cover (80% or even 100%) is not sufficient to avoid yield failures and only partly mitigates yield decreases. Hence additional measures needed to be applied.

A number of management measures increase soil moisture. Irrigation was not an option for the scenarios because the aquifer is already overused (Aquifer "Lagunillas Patzcuaro" of C1 and C2 had an 80% extraction rate, Aquifer "Morelia-Querendaro" of C3 had an extraction rate of 141% (CONAGUA (Mexican National Water Comission) 2008). Another option to increase soil moisture are to adapt tillage operations.

Verhulst et al have shown in field trials in central Mexico that a management change from conventional tillage to zero tillage with residue retention increases soil water content at the beginning of the growing season (Verhulst et al. 2011b). This measure was chosen for the scenario B because its effect has been assessed in field experiments in the same region as the case studies. The effects of different tillage operation (zero tillage vs. conventional tillage) on soil water retention cannot be simulated in AquaCrop. To include the effect of a change in tillage operations, the initial water content was adjusted manually in top 60cm in AquaCrop.

For C2, soil moisture in top 60 cm was increased by around 40mm. This is the minimum increase in initial soil water content needed to avoid the yield failure in the critical year of the 10 year simulations, 2005. The additional soil moisture was distributed evenly over 60 cm depth. Numbers were converted in vol % soil water content and inserted in the AquaCrop initial soil moisture module. Soil water was increased from 23% to 29% (top 11cm), from 23.3% to 30% (11-33cm) and from 28% to 36% (33-60cm) (Figure A.1). No change in soil water content at depths below 60cm.

For C1, soil moisture in top 60 cm was increased by around 60mm. This is the minimum increase in initial soil water content needed to avoid the yield failures in the critical years 2005 and 2008. The additional soil moisture was distributed evenly over 60 cm depth. Soil water was increased from 22% to 30% (top 13cm), from 24 to 32% (13-39cm) and from 19% to 25% (39-60cm) (Figure A.1). No change in soil water content at depths below 60cm. This approach is plausible because initial soil water contents as well as increases in soil water contents are within the range observed by Verhulst et al. However, please note that this approach has uncertainties:

Uncertainty 1

Verhulst et al measured soil water content during three crop cycles. In the data from Verhulst in 2009, at the beginning of the growing season (DAP 0-4) soil water content was 50mm higher in the treatment zero tillage and residue retention compared to conventional tillage. In 2008, the difference in soil water content was 15 mm at the beginning of the growing season. In 2007, both systems had a similar water content at the beginning. This does not give a statistically sufficient evidence that an increase 40 mm or 60mm water in top soil would be reached in average (as assumed in the scenario). Perhaps additional measures would need to be applied to achieve a soil moisture increase of 60mm at the beginning of the growing season. This could be for example contour tillage since the field of C1 has a slope of 3% and is tilled in direction of the slope which leads to higher runoff.

Hence, before a recommendation for the concrete plot could be made, the proposed measures (zero tillage with residue retention and contour farming) would need to be tested in field trials and soil water sampling would need to be done frequently. However, for the purpose of the study this rough approach is sufficient because aim of the study is not to give a recommendation to the concrete farmer for his plot but to show general tendencies. The result of the simulation gives a good indication that yield failures and yield decreases could be avoided for C1, C2 and C3 by a change in management and provides a solid recommendation for future field trials.

Uncertainty 2

Initial soil moisture in simulations was based on measurements of just one year (2013). Soil moisture at the beginning of the season varies from year to year. In the field trials of Verhulst for example soil water content at beginning of growing season varied between 110mm-220 mm depending on treatment and year. In order to get a reliable AquaCrop simulation result for the concrete plot soil moisture would need to be sampled each year and then the simulation would need to be done with the climate data of the corresponding year. Also, the simulation does not include the effect of the second crop (vetch) of C2 on the initial soil water content.

However, in general soil water contents of case studies were in a similar range as in Verhulst trials. Soil water content at beginning of growing season varied in Verhulst trials between 110mm-220 mm depending on treatment and year. For case 1 soil water content at top 60 cm was 131mm. For C2 it was 150mm and for case 3 220mm. Thus it is reasonable to assume that a change in treatment would result in a similar increase in water content at beginning of the season.

Guanajuato scenario A: In scenario A the three plots were cropped completely without irrigation (rainfed). The measure was chosen because the aquifer is strongly overused (Aquifer Cienega Prieta-Moroleon annual extraction rate: 168% of renewable water is extracted).

Without irrigation, maize cannot be seeded in the beginning of may because rainy season has not started yet. Therefore the seeding date of scenario 4A, 5A and 6A was shifted to june 30th. This date was chosen because farmer of C5 has chosen this as seeding date in his first crop, which he grew rainfed.

Without irrigation in the dry season, soil water content would be lower at the beginning of the cropping season because more water would evaporate during winter. Case 4 was irrigated around one week before soil sampling, case 5 one month before soil sampling and case 6 was irrigated around 2.5 months before soil sampling which reflects clearly in the measured upper soil water content (Figure soil water). The initial soil water content in AquaCrop no-irrigation scenarios was therefore modified as follows.

As a proxy, the average soil water content of rainfed cases 1 and 2 was used as initial soil water content of scenario 4A, 5A, 6A. This is a reasonable estimation because soil textures of irrigated and rainfed cases were in a similar range (Table A.5). Also organic matter content is in a similar range: Around 3% for irrigated cases and 2-4 % for rainfed cases.

Case 3 was not included in the proxy because here the groundwater table was at 1.1 m. The moisture gradient was therefore significantly higher than one would expect for case 4, 5, 6 where the groundwater table is not as high.

Uncertainties

Since annual average rainfall in the region with the irrigated cases is with around 600mm lower than in Michoacán with 900mm, the estimation is rather conservative, i.e. soil water content might be even lower. On the other hand, the wilting point of the upper soils of the irrigated cases is a bit higher (22.5, 30.5, 25.5 vol%) compared to the rainfed cases (16%,31%)

Guanajuato scenario B: In scenario B supplemental irrigation was applied in those years where yield decreases >10% occurred in the simulations. It was applied when stomatal closure occurred for more than 3 days on the first day of stomatal closure. The minimal amount of water needed to avoid stomatal closure and maximize yields was applied (trying step by step which amount of water was sufficient in the simulation).

When building the schedule for the supplemental irrigation it was taken into account that water can only be applied after the onset of rainy season, so that storage facilities can be expected to contain sufficient water. Table A.7 shows the cumulative rain until rainwater harvest application date. The water shortage in year 2000 could not be mitigated by rainwater harvest. 280 mm of supplemental irrigation would be necessary to avoid yield failure which cannot be covered by 319 mm of rain.

Table A.7: Irrigation rates and dates for supplemental irrigation from rainwater harvest for Guanajuato B scenarios.

Case	Year	Irrigation date (Day after planting)	Net application (mm)	Cumulative rain until irrigation date (mm)
4B	1999	100	60	504
4B	2000	1; 54; 84; 104 [a]	280 [a]	319 [a]
4B	2001	33; 115	50; 30	263; 524
4B	2002	34	70	258
4B	2005	29	50	164
4B	2009	39	40	270
5B	1999	95; 109	40; 40	504; 504
5B	2001	18; 54; 114	40; 40; 30	228; 338; 524
5B	2002	28; 44	40; 40	257;284
5B	2005	26; 121	40; 20	174; 528
5B	2006	19	40	
5B	2009	38	40	269
6B	1999	106	50	
6B	2000	1,44,65,91 [a]	70,70,70,70 [a]	
6B	2001	32; 121	50; 20	
6B	2002	35; 54	50; 30	

[a]Supplemental irrigation was not applied in B scenarios of year 2000 because there is not sufficient precipitation to fill rainwater harvest storage facilities.

Sensitivity analysis has shown that mulching (80%) before cropping considerably reduces number of years and severity of yield decreases. For example, if mulch cover before seeding would be applied in case 5 yield failure would only occur in 4 out of 11 years instead of 5 out of 11 years. Yield decreases would be less severe with 13% and

21% instead of 18% and 29%. Hence, less water would need to be applied by rainwater harvest if additional measures would be taken that increase soil moisture.

Uncertainties

Due to technical reasons it might be necessary to apply larger volumes of irrigation water. The FAO irrigation training manual states that if the stream size is too small in furrow irrigation this will "result in inadequate wetting of the ridges. Even if the plants are located at the sides of the ridge, not enough water will be available. A small stream size will also result in poor water distribution along the length of the furrow. The advance will be slow and too much water will be lost through deep percolation at the head of the furrow" (Brouwer et al.). With sprinkler or drip irrigation finer control of the applied volumes would be possible. However, furrow irrigation was chosen in the simulations because this is a common technique in that region for maize and for economic reasons the most probable choice.

2. BIOGRACE INPUT DATA

2.1 Input data status quo

For the cultivation step input data for calculation of greenhouse gas emissions of bioethanol production were acquired by field interview on-site for case studies if not indicated otherwise below (Table A.8).

Straw removal: Information on the fate of field residues was acquired by field interview. If farmer reported that straw was removed, straw yield was estimated based on a straw/grain ration of 1.0 for maize and 0.7 for barley (German Federal Government 2012b). Data on straw removal was necessary for calculation of N2O emissions.

Electricity use: Amount of irrigation water pumped and depth of water well was reported by farmer. For Case 4 and 5 pumping depth was 200m, for case 6 pumping depth was 15m. Efficiency of the electricity pump was estimated with 0.65 which corresponds to an average-good pumping efficiency.

Input data for transports, ethanol plant and refinery were was taken from BioGrace default data on *Ethanol from corn (natural gas CHP)*. Field data could not be acquired because currently there is no ethanol production from maize in Mexico.
Emission factors were taken from BioGrace list of standard values which is integrated in the tool. National electricity grid emission factor for Mexico was taken from the BioGrace additional list of standard values: 215.85 gCO_2/MJ electricity (2013a).
All agricultural by products were allocated except for straw. Straw was not considered a by-product to be in line with the European Renewable Energy Directive (EU 2009). Please note that in Table A.8 inputs for barley are listed in a separate column for better overview. However, total emissions of the maize-barley cropping system were calculated and than part of the emissions were allocated to barley.

For allocation of agricultural by-products lower heating values for maize (18.5MJ/kg maize) and barley (17.0MJ/kg barley) were taken from BioGrace (2013b). The LHV of bean (18.5 MJ/kg bean) was taken from EcoInvent (Nemecek and Kägi 2007). The LHV of vetch was considered to be the same as for bean because the nutrient composition of bean and vetch seeds is almost identical.

For calculation of N2O emissions from fertilizers IPCC methodology Tier 1 on the estimation of N2O emissions from managed soils was used, as implemented in the BioGrace Tool (Eggleston et al. 2006, 2013b).

Table A.8: Basic data for maize production and harvest for status quo. Data was acquired by field interview if not indicated otherwise.

Case	1	2	3	4	4	5	6	6
Crop	Maize + bean	Maize + vetch	Maize	Maize	Barley	Maize	Maize	Barley
Yield (kg^{-1} ha^{-1} yr^{-1})[a]	1687	2962	4769	12116	7000	20803	12105	7000
Moisture content[b]	0,15	0,15	0,15	0,15	0,15	0,15	0,15	0,15
Straw (kg^{-1} ha^{-1} yr^{-1})	0	2962	0	12116	4900	20803	12105	0
Co-product (kg^{-1} ha^{-1} yr^{-1})	337	2400	0	0	0	0	0	0
Diesel (MJ^{-1} ha^{-1} yr^{-1})[c]	1889	1889	1889	5361	5361	10723	5361	5361
Electricity (MJ^{-1} ha^{-1} yr^{-1})	0	0	0	10323	17054	14821	906	1132
N-fertiliser (kg^{-1} ha^{-1} yr^{-1})	20	92	104	237	237	348	247	247
Manure (kg N^{-1} ha^{-1} yr^{-1})		60		0	0	0	0	0
CaO-fertiliser (kg^{-1} ha^{-1} yr^{-1})[b]	0	0	0	1600	0	3200	1600	0
K2O-fertiliser (kg^{-1} ha^{-1} yr^{-1})[b]	0	0	0	26	16	52	26	16
P_2O_5-fertiliser (kg^{-1} ha^{-1} yr^{-1})	0	0	31	41	41	120	66	66
Pesticides (kg^{-1} ha^{-1} yr^{-1})[b]	0	0	2	2	2	2	2	2

[a]Corn yields were averaged over 10 year AquaCrop simulations.

[b]BioGrace default value (2013b)

[c]Literature data for low and high input agriculture of maize in Mexico (Riegelhaupt et al. 2010)

2.2 Input data scenarios

In the B scenarios (Michoacán) agricultural production was intensified. The estimation of yields, diesel use and fertilizer use in the B scenarios are described in the following.

Table A.9: Basic data for maize production and harvest for scenario B.

Case and Scenario	1B	2B	3B	4B	5B	6B
Crop	Maize + bean	Maize + vetch	Maize	Maize	Maize	Maize
Yield (kg^{-1} ha^{-1} yr^{-1})[a]	7783	7812	7239	10640	10064	10576
Moisture content[b]	0,15	0,15	0,15	0,15	0,15	0,15
Straw (kg^{-1} ha^{-1} yr^{-1})	0	7812	0	10640	10064	10576
Co-product (kg^{-1} ha^{-1} yr^{-1})	1557	6330	0	0	0	0
Diesel (MJ^{-1} ha^{-1} yr^{-1})[c]	5361	5361	5361	5361	5361	5361
Electricity (MJ^{-1} ha^{-1} yr^{-1})	0	0	0	0	0	0
N-fertiliser (kg^{-1} ha^{-1} yr^{-1})[d]	142	142	135	237	174	247
Manure (kg N^{-1} ha^{-1} yr^{-1})	0	0	0	0	0	0
CaO-fertiliser (kg^{-1} ha^{-1} yr^{-1})[b]	1600	1600	1600	1600	1600	1600
K2O-fertiliser (kg^{-1} ha^{-1} yr^{-1})[b]	26	26	26	26	26	26
P_2O_5-fertiliser (kg^{-1} ha^{-1} yr^{-1})	76	76	76	41	60	66
Pesticides (kg^{-1} ha^{-1} yr^{-1})[b]	2,4	2,4	2,4	2,4	2,4	2,4

[a]Corn yields were averaged over 10 year AquaCrop simulations.
[b]BioGrace default value (2013b)
[c]Literature data for high input agriculture of maize in Mexico (Riegelhaupt et al. 2010)
[d]Scenario estimation

Yield co-product: Yields of bean and vetch were scaled up according to yield of main-product (maize).

N-fertiliser input is one of the most influential factors for GHG emissions of cultivation. To estimate N-fertiliser input of the scenarios a regional yield-nurtrient curve for maize was set up. It was set up based on the fertilizer-yield relationship observed in the case study (Table A.10).

Table A.10: Fertiliser input and yields of case studies.

Case	1	2	3	4	5	6
N-input (kg N ha^{-1} yr^{-1})	20	92	104	237	174	247
Yield (kg maizeha^{-1} yr^{-1})	1687	2962	4769	12116	10401	12105

These data points were fitted with Matlab (Figure A.3) to the solution of the logistic equation $u'_{(x)}=ku(G-u)$ which is given by the logistic function

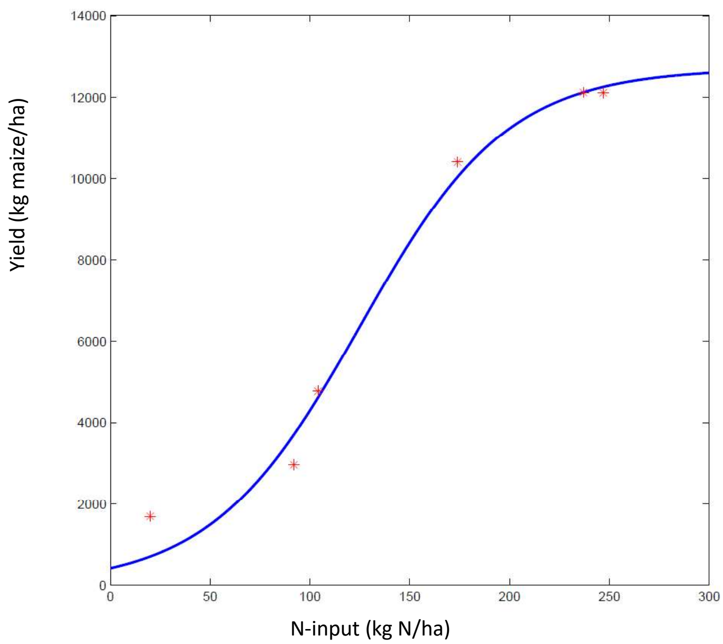
$u_{(x)} = G / [1+\exp(-k*G*x) * ((G/u_0) -1)]$ with $u_{(x)}$ = Yield, x = N-input.

This was done with the Matlab function lsqcurvefit which has used the least-squares-method to calculate the following parameters:

G =12706, K = 2.1234E-06 and u_0 = 422.4635

With this equation the amount of N-fertiliser needed to achieve yields of B scenario was calculated. Please note that this is a rough estimation, because the yield-nutrient curve for a certain crop depends on regional characteristics (rainfall patterns and soil characteristics) as well as crop variety. These parameters vary to a certain extent between the case studies.

Figure A.3: Matlab fit of yield-nutrient curve (blue) to field data (red).



3. LOW INDIRECT IMPACT BIOFUELS (LIIB)

RSB, Ecofys and WWF elaborated a methodology for individual producers to distinguish biofuels with a low risk of causing indirect impacts (LIIB) (vande Staaij et al. 2012). This methodology basically calculates the amount of biofuels produced on a certain plot from yield increases above the regional baseline of yield increases.

Using this methodology the volume of LIIB-biofuels produced by a management switch in Michoacán from status quo to scenario B was calculated in three steps following the methodology by van de Staaj (vande Staaij et al. 2012):

1) The average annual yield growth rate (g) was determined for similar producers in the same region. Annual yield data for rainfed maize in Michoacán for the last ten year was aquired from the Mexican agricultural information service (2009) and plotted in Figure A.4. The best linear fit was taken as yield trend line. It is given by the formula:

$Y_{gr}(x) = 0.0391x + 2.3209$

An average annual growth rate of g = 1.68% was calculated from the yield trend line i.e. 1.68% based on starting year value (1999).

2) Then the future yield baseline $y_b(x)$ of the study case was set. The yield baseline is an extrapolation of the yield trendline starting either (a) at the current year of the trendline, which is 2.7 T/ha (Figure A.4) or (b) at the average yield of the case study site during the last ten years, which is 3.0 T/ha for Case 2, 1.7 T/ha for Case 1 and 4.8 T/ha for Case 3 (Table 2.2 in main article). Staaj et al require that the higher value of these two options is taken, thus the baseline yield for Case 1 was corrected (Figure A.4)

The projected baseline $y_b(x)$ is given by:

$Y_b(x) = y_b(0) + (y_b\ (0) * g * x)$

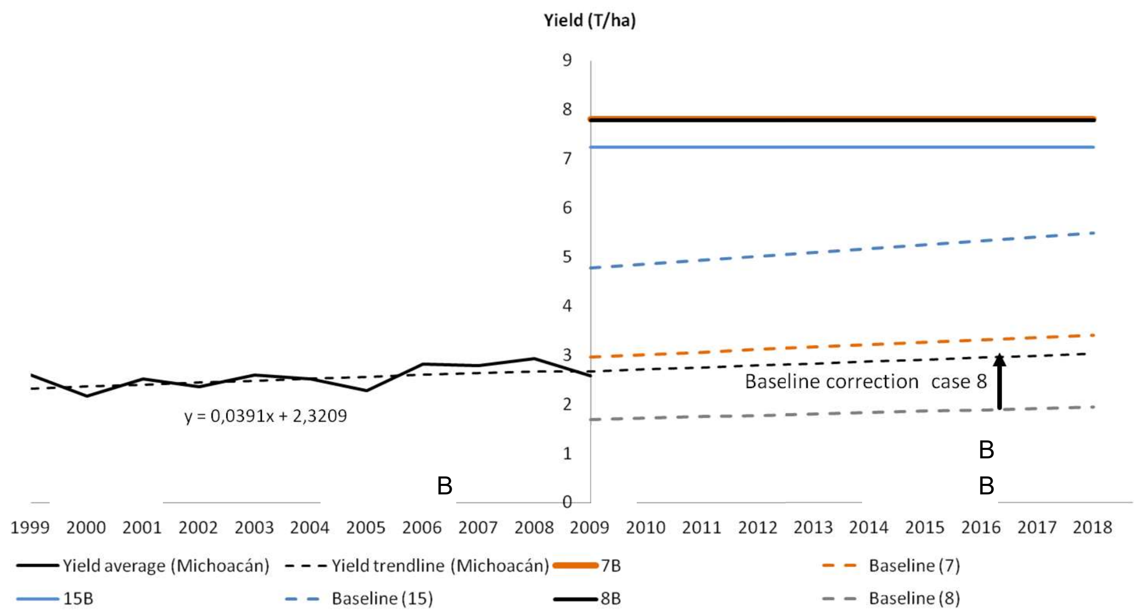
with

$Y_b(x)$ = Baseline yield in year x

g = average annual yield growth rate (1.68%)

$y_b(0)$ = starting point of yield baseline; $y_b(0)$=3.0 T/ha for C2; 4.8 T/ha for C3 and 2.7 T/ha for C1.

Figure A.4: Regional trendline of yield growth (left), case study baseline and scenario yields (right).



3) **The volume of LIIB-maize (V_{LIIB} (x) was calculated.** It includes all production in the study plot scenarios above the baseline. The yield in the scenarios is 7.2 T/ha for case 3 and 7.8 T/ha for case 2 and 1 (Table 2.3 in main article). The volume of LIIB-maize is given by:

V_{LIIB} (x) = (Y(x) - Y_b(x)) * A

with

V_{LIIB} (x) =Volume of LIIB-maize in year x

Y(x) = Yield in year x (T/ha)

Yb(x) = Baseline yield in year x (T/ha)

A = Plot area (ha)

Results for the volume of LIIB-maize and LIIB-biofuel from each study plot in year one and year ten after management switch are given in Table 2.4 in the main article.

4. UPSCALING

Two approaches were applied to estimate what it would mean at a national level if the measures applied in the scenarios were to be applied to all comparable rainfed agriculture in Mexico. The first approach is described in the main paper.

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APPENDIX 2: SUPPLEMENTARY INFORMATION FOR CHAPTER 3

Supporting Information: Maps and data for calculation of Greenwater Efficiency Indicator

Figure B.1: Effective Precipitation from june to october (Data source: Pfister et al (Pfister 2011))

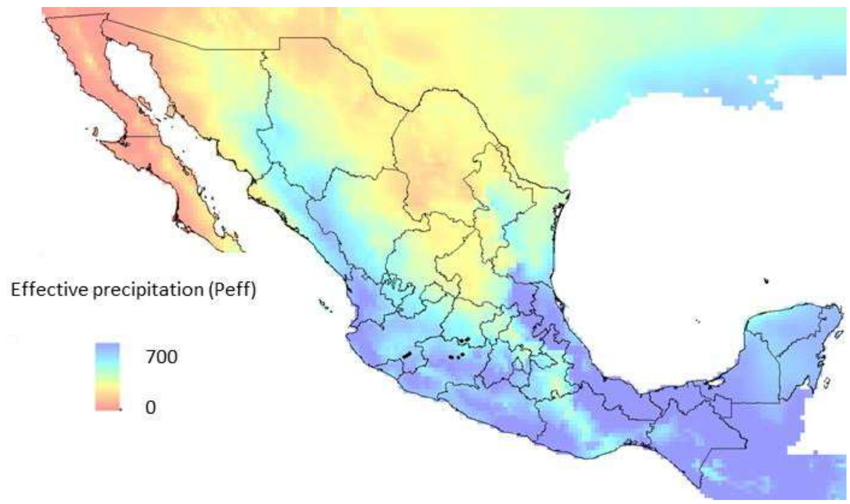


Figure B.2: Transpiration coefficients (calculated) in mm.

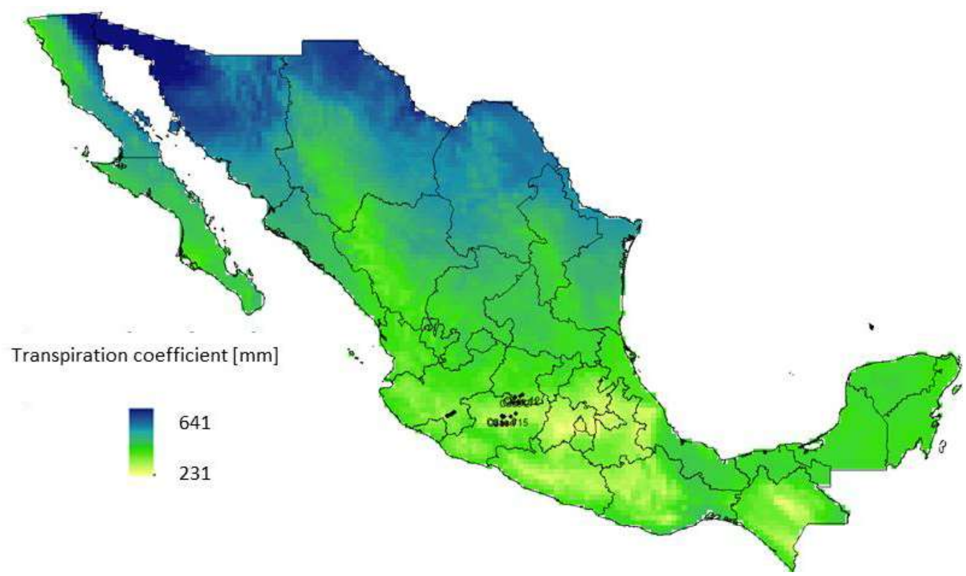


Figure B.3: Transpiration of rainfed maize (calculated) in mm.

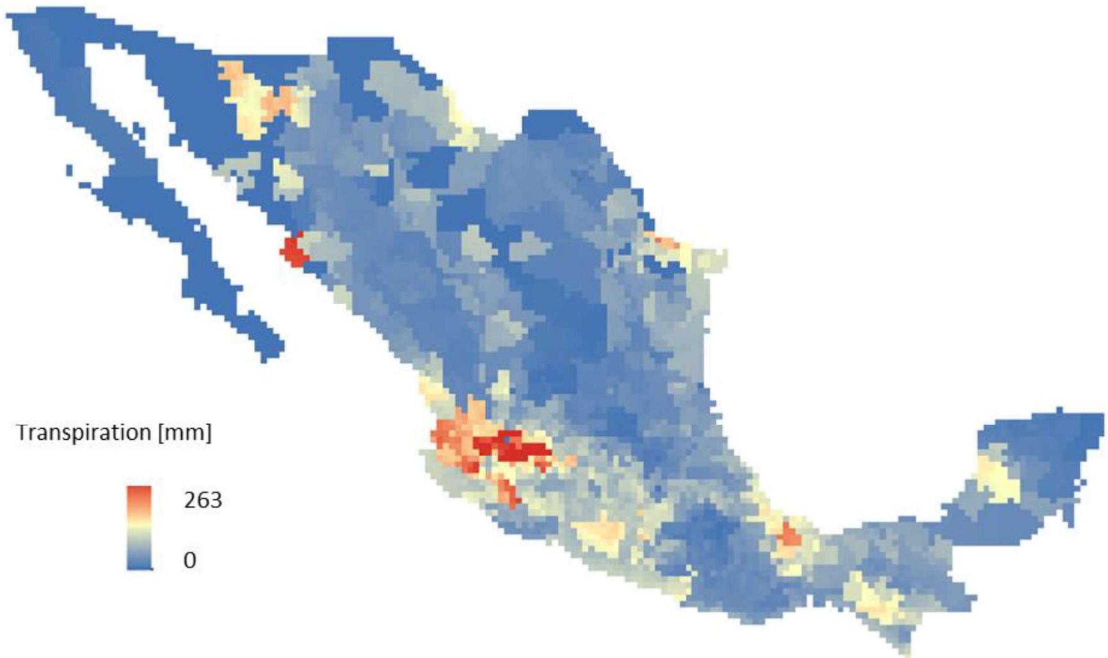


Figure B.4: Harvested area rainfed maize (share of pixel). Data source: SIAP(Mexican Ministry of Environment (SAGARPA) 2012b)

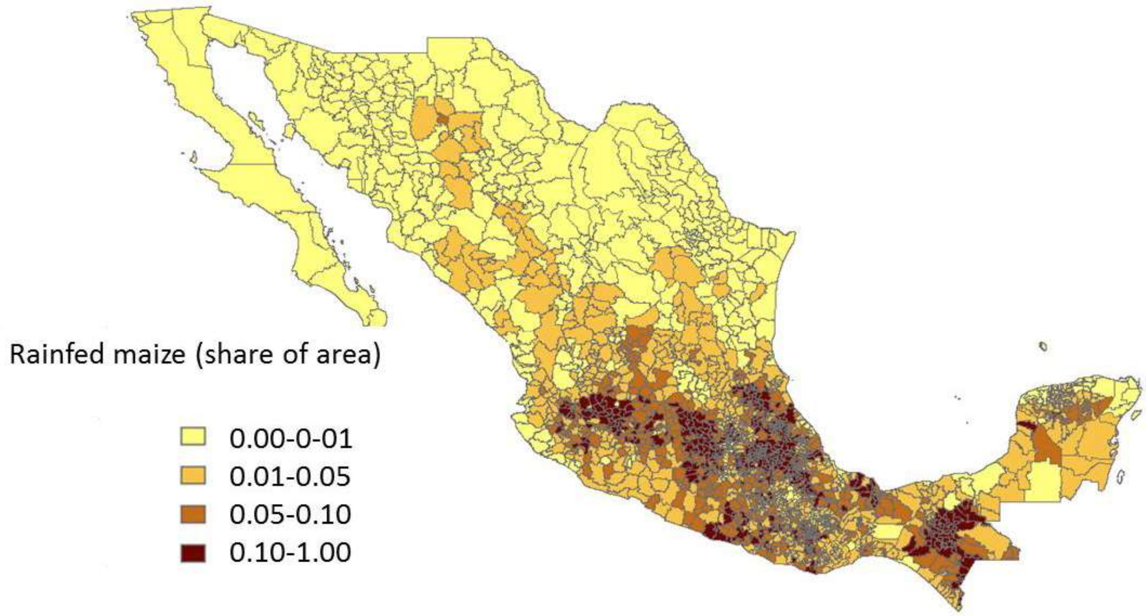


Figure B.5: Comparison of transpiration coefficients for different yields determined by the methodology of the current paper and determined for the case studies in Hennecke et al(Hennecke et al. 2015) using the AquaCrop model (unpublished data).

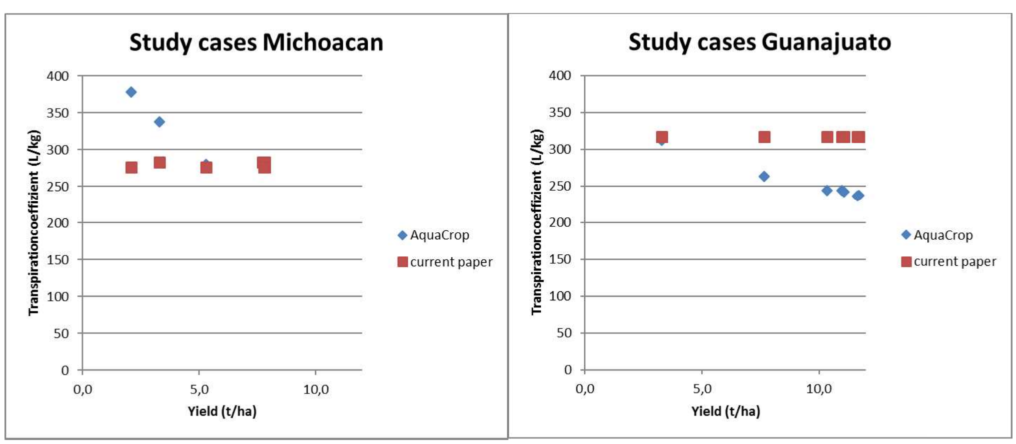
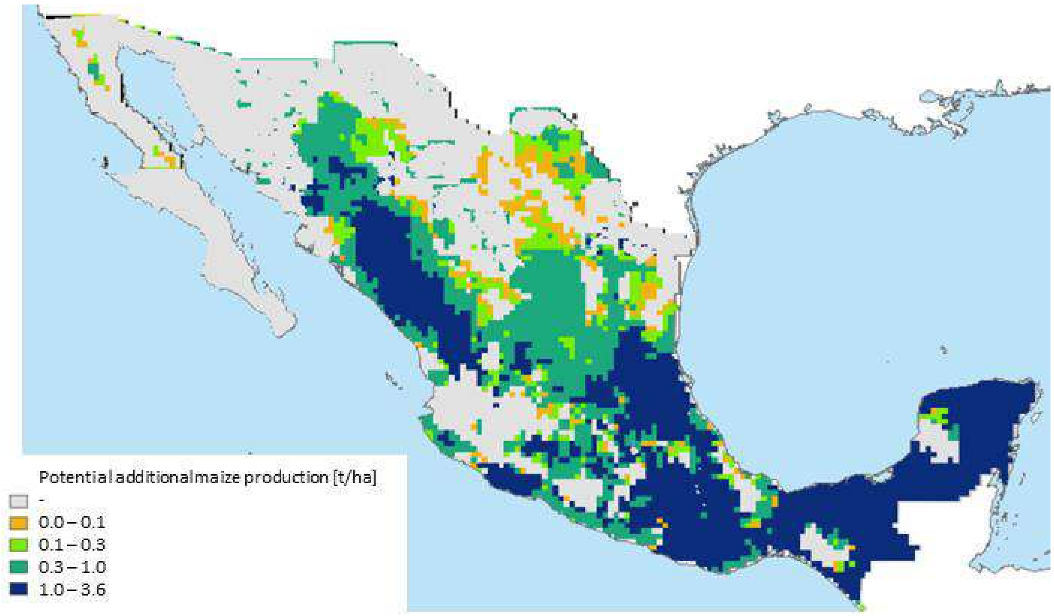


Figure B.6: Potential additional maize production that can be achieved by increasing GEFF to 0.15 in all areas cultivated with rainfed maize.



Potential additional maize production [t/ha]
-
0.0 – 0.1
0.1 – 0.3
0.3 – 1.0
1.0 – 3.6