The Environmental Impacts of Achieving Global Food Security: From Agricultural Intensification to Large-Scale Land Acquisitions

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Abstract

Producing more food while minimizing environmental impacts is one of humanity's most pressing challenges for achieving sustainable development. Rising affluence, demographic growth, increased crop-based biofuel use, and an intensifying livestock sector are contributing to unprecedented demands on crop production – and the resources required to support it – while climate change already shows evidence of affecting historical crop yield trends. Because of these pressures on a finite amount of suitable agriculture land, many countries and investors have begun acquiring large tracts of land in the global South, where land is relatively inexpensive, the potential to increase crop yields is generally high and property rights are often poorly defined. By acquiring land, investors can realize large profits and countries can substantially alter the land and water resources under their control, thereby changing their outlook for meeting future demand. In permitting such investments, targeted countries also hope to promote the rapid inflow of agricultural technologies into their underperforming agricultural areas. However, many of the impacts of agricultural intensification and land acquisition remain poorly understood.

To this end, this dissertation examined the major historical impacts of agricultural intensification on rural livelihoods and the environment as well as the potential of the global food system to meet future demand while minimizing environmental impacts. The work contained herein showed that the livestock sector has led to important food-environment tradeoffs and has become more efficient in terms of land use and greenhouse gas emissions. This dissertation also demonstrated that a combination of enhancing crop yields and moderating diets has the potential to greatly increase the number of people able to be fed globally. Following these studies of food supply and its environmental impacts, this work assessed the impacts of large-scale land investments on livelihoods and the environment in targeted areas. The results of these studies showed that millions of people in the developing world could potentially lose their livelihoods as a result of displacements and that land concessions have significantly and substantially enhanced rates of forest loss in Cambodia. Finally, this dissertation showed that there is a large potential to reduce the amount of resources associated with food production while meeting future demand, thereby increasing the self-sufficiency of nations and minimizing the need for land acquisitions elsewhere. The many novel contributions of this dissertation help to integrate the various benefits and impacts of the global food system and inform responsible decision-making that incorporates human well-being and environmental stewardship.

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It is in the nature of the good to give of itself in the same way that it is the nature of light to emanate rays and to illuminate what is around it.

Seyyed Hossein Nasr

The Garden of Truth

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INTRODUCTION

The global food system is facing unprecedented pressure. Population has more than doubled over the past 50 years and is expected to reach 9.7 billion by 2050[1]. Global per capita calorie demand rose by 31% (50% for animal products)[2], with diets projected to become even richer in the coming decades[3]. Biofuel production has increased seven-fold since the start of the century alone[4]. Animal demand for feed now makes up 15% of global crop production. Climate change – having already affected historical crop yields to some extent[5] – is expected to significantly impact regional yields of major crops in the near future[6].

In the face of these ongoing demands and stressors on global food production, the pervasive environmental impacts of agriculture have also become apparent. These impacts are well studied, ranging from the depletion of rivers and groundwater for irrigation[7,8] to nutrient pollution from the large-scale anthropogenic fixation and application of reactive nitrogen for fertilizers[9,10] to greenhouse gas emissions from mechanized cultivation, land use change, ruminant production and food trade[11]. With humanity already exceeding its sustainable use of Earth's systems in a number of ways[12,13,14,15,16], there is growing concern that the potential resource requirements of future food demand portend yet more profound and pervasive consequences[17,18]. Thus, there is widespread agreement that food production must increase substantially while also minimizing environmental impacts, an approach known as 'sustainable intensification'. Potential solutions to address this apparent dilemma include closing crop yield gaps, reducing food waste, moderating diets and reducing inefficiencies in resource use[19].

Regardless of how sustainable these potential solutions may prove to be if pursued, many countries are faced with more immediate food security concerns. In recent years, import-reliant countries in particular have been made aware of their vulnerability to perturbations to the global food system, as they rely on food sources that are produced in areas beyond their boundaries and therefore beyond their direct control[20]. This susceptibility to system shocks became especially apparent during the 2008 food crisis, during which droughts in major producer countries were followed by spikes in world food prices[21,22,23]. To curb the domestic escalation in food prices, some governments went so far as to ban grain exports, much to the concern of import-dependent countries[24]. It thus became clear that food security was at risk in many importing countries and that the adequacy of resources for long-term food and energy security required redefinition. At first, many import-reliant countries tried to negotiate long-term contracts for supplies of major grains[25]. Finding this option largely unsuccessful, governments and corporations then began acquiring rights to land in the global South[26], as part of a phenomenon often referred to as the global land rush[27]. As a result, 43 million hectares of land have been acquired to date[28].

Proponents of these land investments contend that large-scale land acquisitions will bring economic development, energy security and improved crop production in underperforming agricultural land, and indeed there are many instances in which these land deals produce positive outcomes (e.g. job creation, infrastructural improvements)[29]. However, the potential for profit has frequently led to the treatment of these lands as a commodity, which can in turn prioritize a purely economic perspective and downplay the potential impacts on local populations (e.g., lost livelihoods, food insecurity) and the environment (e.g., soil erosion, deforestation)[26,30,31,32,33]. In addition, the resources acquired through land deals are oftentimes no longer available in, or are removed from, the targeted areas[26] and exported for sale elsewhere[25,30,34]. Therefore the process can often entail the displacement of small-hold farmers or the exclusion of previous users from access to the land (see ref. 35). Overall, that the suite of benefits and impacts is unique to each land deal and can differ between local and national scales means that more detailed assessments of these land deals is still much needed[29]. This global land rush is evidence of a persisting and apparent disconnect between efforts to improve global food security and the local human and environmental impacts of achieving it. As such, this dissertation addresses a pressing need to integrate concerns – at both global and local scales – of food security, rural livelihoods and environmental impacts so that potential solutions for achieving agricultural intensification might truly be sustainable.

While the first portion of this introduction was meant to convey a brief introduction to the global food system and its human and environmental impacts that this dissertation explores, the remainder of this section provides a brief overview of the studies contained in the subsequent chapters – highlighting their focus and key findings. Using the example of the livestock sector, this dissertation first highlights how the need to rapidly increase food production has led to a substantial expansion of humanity's environmental footprint related to agriculture. Specifically, Chapter 2 examines the major environmental impacts from animal production over the past 50 years. With two contrasting processes – 1) greater demand for animal products and the feed to support it, and 2) a transition to more efficient non-ruminants (e.g., pigs and poultry) – occurring within historical animal production, it is unclear whether increasing consumption or enhanced efficiencies have played a more important role in determining resource requirements. To investigate this, this study calculated the land use, nitrogen application and greenhouse gas (GHG) emissions required per animal calorie, how these efficiencies have changed through time, and whether any important environmental and food security tradeoffs have occurred. This study found that the efficiency of land use and GHG emissions has improved with time while nitrogen use efficiency has markedly increased – due in large part to a growing reliance on feed. Despite improvements in the efficient use of certain resources, overall the land demand, nitrogen demand

and GHG emissions associated with animal production continue to increase. This chapter was published in *Environmental Research Letters* (10: 125013) and was co-authored by Kailiang Yu, Mario Herrero, Petr Havlik, Joel A. Carr, and Paolo D'Odorico.

Chapter 3 examines the number of people that could potentially be fed under selected scenarios of yield enhancements, dietary changes, crop-based biofuel demand and reductions in food waste. This study used national inventories on food production and consumption patterns – as well as projections of future biofuel use and dietary demand – to consider how improving one (or a combination) of the various pressures on food supply can enhance future food security. The results of this study showed that increasing dietary demand will be largely to blame should future crop production fall short of demand. However, depending on the extent to which yields can improve by 2050, it is possible to feed billions more people if appropriate solutions are adopted in time. This study demonstrates that by combining both demand- and supply-side approaches it is possible to better ensure future food security, but only if long-term sustainability is the focus. This chapter was published in *Earth's Future* (2: 559-565) and was co-authored by Paolo D'Odorico and Maria Cristina Rulli.

Chapter 4 explores the mechanism of large-scale land acquisitions as a linkage between the global-scale changes in the global food system – demographic growth, increasing affluence, biofuel demand and the feed requirements of an intensifying livestock sector – and selected local environmental and livelihoods impacts. Specifically, this study estimated the number of people living in targeted areas whose livelihoods would potentially be lost as a result of large-scale land acquisitions. To do this, this study utilized information on current crop yields, agricultural area leased to investors, and crop prices to calculate the value of crop production on acquired land if they were fully put under production. This study then converted the value of this potential production – using average per capita income – into the number of people whose incomes might potentially be affected by large-scale land investments. This study showed that millions of people may be affected and that – for certain countries – this could be a substantial portion of a nation's population. This chapter was published in *Population and Environment* (36: 180-192) and is co-authored by Paolo D'Odorico and Maria Cristina Rulli.

Chapter 5 examines the human and environmental impacts of large-scale land acquisitions occurring in Cambodia, a country located in one of the regions most targeted by the global land rush. This case study combined a high-resolution (30 m) tree cover map with polygons of economic land concessions. To compare rates of forest loss inside and outside these concessions, this study employed a covariate matching approach that controls for confounding characteristics that may make an area more susceptible to deforestation – regardless of whether it is located in a concession. This study showed that rates of forest loss were markedly higher within acquired areas and that the more than 200,000 people currently living in these large tracts of contracted land are at increased risk of experiencing livelihood impacts, land insecurity and displacement. This chapter was published in *Nature Geoscience* (8: 772-775) and was co-authored by Kailiang Yu, Maria Cristina Rulli, Lonn Pichdara, and Paolo D'Odorico.

Chapter 6 considers the potential impacts of future food production – in terms of water, nitrogen, land and GHGs – as well as possible strategies to prevent further increases in these resource requirements. To begin this analysis, this study first calculated the total food-related environmental burdens for water, GHGs, nitrogen and land in the year 2050 under constant (circa 2009) footprint intensities (i.e., resource use efficiencies) and for several future diet scenarios. By examining these changes relative to the year 2009, this study then determined the improvement in footprint intensity required to prevent an overall increase in the environmental burden of a resource and compared the required change to projections of historical improvements in production efficiencies. This study showed that efficiency enhancements alone cannot prevent an increase in the environmental footprint of the global food system if affluence continues to determine consumption patterns. However, combining efficiency with a transition to less environmentally burdensome dietary choices can effectively offset the increased demand from population growth. This chapter has been accepted for publication in *Global Environmental Change* and was co-authored by Jessica A. Gephart, Kyle A. Emery, Allison M. Leach, James N. Galloway, and Paolo D'Odorico as co-authors.

This dissertation concludes in Chapter 7, which briefly summarizes the novel contributions of this dissertation and how these findings further current knowledge on food production and its human and environmental impacts. This concluding chapter also describes how better integration is required across the various dimensions of the global food system as well as between global and local efforts.

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HISTORICAL TRADE-OFFS OF LIVESTOCK'S ENVIRONMENTAL IMPACTS

Abstract:

Human demand for animal products has risen markedly over the past 50 years with important environmental impacts. Dairy and cattle production have disproportionately contributed to greenhouse gas (GHG) emissions and land use, while crop demands of more intensive systems have increased fertilizer use and competition for available crop calories. At the same time, chicken and pig production has grown more rapidly than for ruminants, indicating a change in the environmental burden per animal calorie (EBC) with time. How EBCs have changed and to what extent resource use efficiency (RUE), the composition of animal production and the trade of feed have played a role in these changes have not been examined to date. This study employed a calorie-based perspective, distinguishing animal calorie production between calories produced from feedcrop sources – directly competing with humans for available calories – and those from non-feed sources – plant biomass unavailable for direct human consumption. Combining this information with data on agricultural resource use, this study calculated EBCs in terms of land, GHG emissions and nitrogen. This study found that EBCs have changed substantially for land (-62%), GHGs (-46%) and nitrogen (+188%). Changes in RUE (e.g., selective breeding, increased grain-feeding) have been the primary contributor to these EBC trends, but shifts in the composition of livestock production were responsible for 12%-41% of the total EBC changes. In addition, the virtual trade of land for feed has more than tripled in the past 25 years with 77% of countries currently relying on virtual land imports to support domestic livestock production. These findings indicate that important tradeoffs have occurred as a result of livestock intensification, with more efficient land use and emission rates exchanged for greater nitrogen use and increased competition between feed and food. This study provides an integrated evaluation of livestock's impact on food security and the environment.

Introduction

Animal production is one of the most extensive and impactful means by which human activities affect the environment[1,2]. Large amounts of land (3.86×10^9 ha yr^{-1}), water (2091 km^3 H_2O yr^{-1}), fertilizers (101 Mtonne N yr^{-1}) and GHG emissions (2.45 Gtonne CO_2eq yr^{-1}) are needed to support feed production, grazing lands and animal maintenance (circa 2000; refs. 3,4). Animal biomass demand (i.e. feed, grasses, and crop residues) has increased 108% over the past half-century while animal calorie production has more than tripled in response to rapid growth in demand for animal products[5]. Thus, intensification of livestock systems has been responsible for much of the growth in animal calorie production[2]. Though whether this intensification has in fact minimized the environmental burden of animal production appears to be a more complicated storyline. The transition of the livestock sector from ruminants towards non-ruminants has meant improved efficiency per animal calorie produced, both in terms of land area and methane (CH_4) emissions. At the same time, this shift has meant greater reliance on crops for feed and greatly increased the amount of nitrogen fertilizers and non-CH_4 GHG emissions associated with livestock production. While the history of livestock's growing environmental costs as well as its future potential impacts have been well studied in recent years[1,4,6,7,8,9], the efficiency with which a given resource (e.g., land, CO_2, nitrogen) can be converted into animal calories, how this has changed over the past 50 years, and to what extent environmental trade-offs have occurred have not been quantified to date.

Changes to the environmental burden of producing an animal calorie are influenced in two main ways. First, at the scale of the individual animal is its resource use efficiency (RUE), namely how much of a resource is required to produce a given amount of that animal. Through selective breeding and higher quality diets[10,11], a species' ability to incorporate calories and nutrients can improve with time, so that less feed or fodder is required to produce the same amount of animal calories. Such changes are well documented in the animal science literature[1,12]. Second, at the scale of all livestock production, changing the relative amounts of the various animal products that comprise all of livestock calorie production (e.g., eggs/milk vs. meat; pigs/poultry vs. ruminants) makes it possible to alter total resource use, even if the RUE of individual animal products remains constant. Recent studies[13,14] have shown that it is possible to substitute resource-intensive animal products (e.g. beef) with lower impact ones and still meet human dietary demand and nutritional requirements. Yet, it is still unclear to what extent the environmental burden per animal calorie has changed through time as well as what the relative contribution of RUE and livestock composition to these changes has been.

Thus, our purpose here is two-fold: 1) to quantify the changing environmental burden of global livestock production in terms of land, reactive nitrogen and GHG emissions over the past 50 years, and 2) to determine what fraction of these changes are attributable to shifts in RUE and livestock composition. I examine historical data for 5 main animal products – cattle meat, pig meat, poultry meat, eggs and milk – in 173 countries from the year 1961 to 2010. Combined with information on agricultural inputs and emissions, I calculate trends in the animal productivity of land (kcal ha^{-1}), GHG (kcal kg CO_2eq^{-1}) and nitrogen (kcal kg N^{-1}) and explore the possibility of environmental trade-offs as the livestock sector has transitioned to more intensive systems of production. I conclude our analysis by determining what percentage of the changes in animal productivity of land, GHG emissions and nitrogen fertilizer application is attributable to changes in RUE and what portion is attributable to changes in livestock composition.

Methods

Data sources

Country-level data on feed supply, animal production (i.e., poultry, beef, pig, eggs and milk), crop yields, pasture area, trade, and synthetic nitrogen fertilizer application came from FAOSTAT[15].

Country-level data on GHG emissions from agriculture also came from FAOSTAT[15]. These emissions included enteric fermentation (CH_4), direct and indirect emissions from nitrogen fertilizers (N_2O/CO_2), energy use for feed production (CO_2), rice cultivation for feed (CH_4), manure left on pastures (N_2O), manure applied to fields (N_2O) and manure management and storage (N_2O/CH_4). GHG emissions for agricultural energy use were multiplied by the ratio of feed production to total crop production, while GHG emissions for rice cultivation were multiplied by the ratio of rice production used for feed to total rice production. Emissions from transport or land use change were not included.

Crop-specific application values of synthetic nitrogen fertilizers for the year 2010 came from a recent study by the International Fertilizer Industry Association (IFA)[16]. These values are reported for 26 countries, the EU-27 and the rest of the world (Suppl. Tables 2-3). Thus, any countries not among the EU-27 or the 26 other countries were all assigned the same crop-specific N application values, as the application of synthetic N in these countries is only 6% of the global total[16]. Annual country-specific estimates of the percentage of total synthetic N consumption used for grassland fertilization for 1961 through 2009 came from Lassaletta and colleagues[17]. In addition, while manure applied to soils continues to be an important source of nitrogen for enhancing crop production (23% of total applied N in 2010)[15], it was not included in our analysis because: 1) manure can be considered a recycling of reactive N from a nitrogen cascade perspective[18], and 2) information on crop-specific application rates was not available.

Due to a lack of comprehensive historical data, the water demand of livestock – though an important impact – was not included in our analyses.

Animal calorie production from feed and non-feed sources

Animal production was converted into calories using data from FAOSTAT's Food Balance Sheets and Commodity Balances[15]. Calories (kcal) were used in lieu of the S.I. unit (kJ) to follow convention of the food security literature and to better relate the findings to human demand and diets. This animal calorie production was partitioned between feed-fed and non-feed animal calorie production by country for the years 1961 through 2010 following the methodology of Davis and D'Odorico[5]. To define 'feed sources', I considered 40 main crops used for animal feed (see Supplementary Table 1); these crops were selected because: 1) each contributed at least 100,000 tonnes to global feed use in the year 2009, and 2) together they comprised at least 93% of global feed production for any given year. All other sources of plant biomass for animal diets were considered a 'non-feed source', consistent with the assumption used by Davis and D'Odorico[5]. Under this definition, fodder crops (e.g., alfalfa, clover, green maize), crop residues and permanent grasslands are considered as non-feed products – even though their production may compete with other uses of cultivatable land – because human and animal demands do not directly compete over the consumption of most of these crops. In addition, because some of these crops (e.g. alfalfa) can also be directly consumed by humans, our method of calculation means that, to a limited extent, I underestimate the feed calories available for animal consumption.

The fraction of animal calorie production derived from non-feed sources (i.e., non-feed fraction, *NFF*) for country x in year t was calculated as:

$$NFF_{x,t} = 1 - \left(\frac{\sum(k_i c_i)}{\sum(FCR_j k_j a_j)}\right) \quad (1)$$

where k_i is calories per tonne of crop i, c_i is the tonnes of crop i used for feed, FCR_j is the feed conversion ratio (FCR) for animal product j, k_j is calories per tonne of animal product j and a_j is the tonnes of production of animal product j. FCRs were derived from Herrero et al.[4] for 28 geographic regions. Thus, in calculating *NFF* for country x, I used the FCR for the geographic region in which country x was included. FCR values – along with the countries corresponding to each geographic region – are reported in Davis and D'Odorico[5]. I should note that the calorie-based approach used here limits our findings somewhat, in that it is only possible to determine the fraction of total animal calorie production attributable to feed and non-feed sources but not for an individual animal product.

Animal productivity of land

The non-feed animal productivity of land (kcal ha^{-1}) was estimated as the non-feed animal calorie production divided by the area of 'permanent meadows and pastures', the same definition from FAOSTAT used by Ramankutty et al.[19] to map global pastures. Under this definition it is important to note that permanent meadows and pastures are actively used for grazing to varying degrees, thus our estimate of the non-feed animal productivity of land is likely conservative. The feed-fed animal productivity of land was estimated in a similar way – by taking the ratio of feed-fed animal calorie production to the cropland area required to grow feed sources. However, the calculation of feed-fed animal productivity of land also accounted for the trade of feed (and the difference in crop yields between the importing and exporting country). The effect of trade was accounted for by assuming that for country x the percentage of feed from imports was the same as the proportion, p_x, between its imports and domestic supply (i.e. production minus exports plus imports) (Supplementary Table 4). The total area required to grow the domestic supply of feed of country x, h_x, was determined in two parts. First, the amount of land required for domestically produced feed, $h_{x,dom}$ was calculated as:

$$h_{x,dom} = \sum \left((1 - p_x) \frac{t_{cg,x}}{r_{cg,x}} \right) \tag{2}$$

where $t_{cg,x}$ is the tonnes of domestic feed supply for a given crop group in country x and $r_{cg,x}$ is the yield of that crop group in country x (crop mass per unit area). Second, the amount of land virtually imported by country x, $h_{x,imp}$, was found by:

$$h_{x,imp} = \sum \left(\frac{t_{cg,y,x}}{r_{cg,y}} \right) \tag{3}$$

where $t_{cg,y,x}$ is the tonnes of a given crop group exported from country y to country x and $r_{cg,x}$ is the yield of that crop group in country y. Thus,

$$h_x = h_{x,dom} + h_{x,imp} \tag{4}$$

Several countries in the study did not report areas for permanent meadows and pastures (i.e., Egypt, Kiribati, Malta and Netherlands Antilles). In these cases, the feed-fed productivity was assumed to equal the overall productivity. In addition, because Davis and D'Odorico[5] assume global historical changes in FCR, this likely means that, to a certain extent, I overestimate changes in the feed-fed animal productivity of land for Africa and Asia and underestimate for Europe and the Americas (see Suppl. Table 4 for list of countries included in each region).

Assumptions in partitioning global resource use

To determine the global use of each environmental burden by feed-fed and non-feed animal calorie production, several assumptions were made. First, I assume that all emissions related to energy use arise from intensive systems and are attributable to feed sources [1]. These data on GHG emissions from energy were only available from the year 1970 onwards but initially only contributed ~5% of total GHG emissions in the first years when the data were available. Second, substantial differences exist in enteric emission rates between animal production systems (especially as a result of feed quality; see refs. 4,11). For simplicity, however, I assume that all enteric emissions originate from animal calories derived from non-feed sources, as the lowest quality ruminant diets (i.e., those with low concentrations of protein and calories and derived almost entirely from non-feed sources) have emission rates sometimes two orders of magnitude higher than higher quality diets[4]. To check the sensitivity of this assumption on how enteric emissions are attributed to feed and non-feed sources, I also performed our analysis assuming that enteric emission rates from animal calories derived from feed-fed and non-feed sources were the same and found that, while the calculated GHG emissions per animal calorie were somewhat different, this had no important effect on the temporal behavior of our findings (Supplementary Figure 1). Third, following Liu and colleagues[20], I assume that any managed manures originated from intensive systems and all manure deposited on grasslands remained on those grasslands. Fourth, the fraction of synthetic N fertilizer consumed for feed production in a given year was assumed equal to the amount of feed production in that year divided by total crop production. To validate this assumption, I divided crop-specific fertilizer application amounts (reported by the International Fertilizer Industry Association[16,21]) by crop production to determine crop-specific rates of N fertilizer application for crop groups (Supplementary Table 3). Multiplying these rates by feed production, the amount of N used for feed production in 2006, 2007 and 2010 was calculated and in good agreement with our estimates. Nitrogen consumption to support animal calorie production from non-feed sources was estimated by multiplying country-specific estimates of the percentage of total synthetic N consumption used for grassland fertilization[17] with country-specific data on total N consumption[15]. Because the Lassaletta et al.[17] dataset did not report a value for the year 2010, I calculated N application to grasslands for this year as a linear extrapolation of nitrogen consumption for non-feed animal production for 1961 through 2009.

Obtaining current global footprints of animal products

Crop-specific nitrogen efficiency for plant products (i.e. kg of applied N per kg of crop) was calculated as the amount of nitrogen applied in 2010[16] divided by the amount of crop production. Production-weighted averages were used to combine the nitrogen efficiencies of individual crops into the larger commodity groupings. Because pulses were included with 'other crops' in the IFA data, the nitrogen efficiency value calculated for soybeans was used for pulses, as soybeans were the only N-fixing crop for which a value was reported. Using dry matter intake values and feed rations reported by Herrero et al.[4] (Supplementary Table 5), the current global N efficiency of animal product *j*, η_j, was calculated as:

$$\eta_j = DMI_j \sum \left(\frac{r_{cg,j}\eta_{cg}}{100}\right) \tag{5}$$

where DMI_j is the dry matter intake per kilogram of animal product *j*, $r_{cg,j}$ is the feed ration of a given crop group for animal product *j* (reported as a percentage of total biomass intake) and η_{cg} is the N use efficiency of that crop group. Nitrogen applied to pasture land was split between beef and milk production (92% and 8%, respectively) following the methodology of Eshel and colleagues[13]. The same methodology was used to determine current land use efficiency for specific animal products.

Global GHG emission rates (kg CO_2eq kg animal^{-1}) for each animal product were calculated based on data reported in two FAO life-cycle assessment (LCA) studies of major animal production systems[10,11]. These calculations are detailed in Supplementary Table 6. Because the LCA studies (1 CH_4 = 25 CO_2eq; 1 N_2O = 298 CO_2eq) and FAOSTAT (1 CH_4 = 21 CO_2eq; 1 N_2O = 310 CO_2eq) employed different global warming potentials (GWPs), emission rates from the LCA studies were corrected using the ratio of the FAOSTAT GWP to the LCA GWP. All resource use efficiency values are summarized in Supplementary Table 7.

Attribution of change to RUE and livestock composition

We considered two modes of change in the environmental burden per animal calorie (EBC): resource use efficiency (RUE) and livestock composition. The overall historical EBC, EBC_{hist}, was calculated as the magnitude of the environmental impacts (resource used for animal production, or emissions of GHGs and pollutants) divided by the total animal calorie production. To determine the contribution of changing livestock composition in year t to $EBC_{hist,t}$, I calculated what the EBC would be holding RUE constant at year 2010 values as follows:

$$EBC_{constRUE,t} = \frac{\Sigma(\eta_{j,2010} p_{j,t})}{\Sigma(p_{j,t})} \quad (6)$$

where $\eta_{j,2010}$ is the RUE value for animal product j in the year 2010, and $p_{j,t}$ is the amount of that animal good produced in year t. I then calculated the changes in EBC_{hist} and $EBC_{constRUE}$ relative to 1961 values as:

$$rEBC_{hist,t} = \left(\frac{EBC_{hist,t} - EBC_{hist,1961}}{EBC_{hist,1961}}\right) \quad (7)$$

and

$$rEBC_{constRUE,t} = \left(\frac{EBC_{constRUE,t} - EBC_{constRUE,1961}}{EBC_{constRUE,1961}}\right) \quad (8)$$

where Equation 7 keeps all components as dynamic and Equation 8 keeps RUE constant but allows all other variables (i.e., livestock composition and animal calorie production) to change with time. With these two scenarios calculated, the relative contribution to EBC when holding livestock composition constant, $rEBC_{constLS}$, in year t was then calculated simply as the difference of $rEBC_{hist,t}$ minus $rEBC_{constRUE,t}$. In this way, I were able to determine the contribution of changing livestock composition and changes in RUE to the overall per calorie environmental burden of livestock. As a point of note, I found that $EBC_{hist,2010} \neq EBC_{constRUE,2010}$. This discrepancy is because the amounts of feed reported by FAOSTAT and the Herrero et al.[4] study were different. However, even when correcting for this discrepancy, I found that it had no important effect (Land: ±1.7%; GHG: ±0.0%; N: ±6.4%) on the relative contribution of RUE and livestock composition to overall change in EBC. As a final note, the relative contribution of RUE and livestock composition to overall change in EBC likely varies between regions. However, I were unable to assess this aspect of the present study, as detailed trade data do not exist for the beginning of the study period.

Results

The overall productivity of land for animal calories has increased by 165%, from 87000 to 231000 animal kcal per hectare (ha) over the study period (Figure 1a). During this time, the productivity from feed sources was 2.1-3.7 times greater than from non-feed sources, with Europe having a particularly high feed-fed productivity. On the other hand, Europe is also the only region with no clear positive trend in overall productivity – largely a result of its decreasing production of animal calories using non-feed sources (see ref. 5). Africa and Oceania have maintained relatively low productivities while Asia has markedly increased the efficiency with which it utilizes land resources to produce animal calories. In addition, I find that those countries with high levels of animal production (e.g., USA, China, Brazil, India) are not necessarily the most efficient users of land for livestock (Figure 2).

The trade of feed has also played an important role in these changing productivities. From 1986 to 2010, the virtual trade of land for feed more than tripled from 110 Mha to 337 Mha and is currently equivalent to 7.6% of the total land required for livestock production (Figure 3). While the Americas have been consistent exporters of feed during these 25 years, the majority of inter-regional imports has gradually transitioned from Europe to Asia. I also find that Asia has not been able to achieve self-sufficiency of its animal calorie production (i.e. domestic supply exceeding domestic demand) despite its increased involvement in acquiring feed imports (Supplementary Figure 2). Looking at the country scale, five nations - Argentina, Brazil, India, Ukraine and the US – exported 238 Mha yr^{-1} (71% of all virtual land traded internationally for feed), while China alone accounted for 19% of virtual land imports for feed (Suppl. Table 8). In total, 133 out of 173 countries were net importers of virtual land for feed (Figure 3b).

This study also examined the trade-offs between some of the main environmental impacts of livestock and how they differ between animal calorie production from feed and non-feed sources. Specifically, I found that animal calories produced from feed sources were more efficient than non-feed sources in terms of land use and GHG emissions, using on average 65% less land and emitting 59% less GHGs per animal calorie, respectively (Figure 4a-b). Conversely, the production of animal calories from non-feed sources was substantially more efficient in terms of fertilizer use – an average of 80% less nitrogen per animal calorie over the time period (Figure

4c). These results indicate that as animal production has increasingly relied on feed sources, the amount of land and GHG emissions associated with the production of an animal calorie has decreased, while the opposite has occurred for required fertilizer.

Together, changes in RUE and in the composition of livestock production contributed to change the EBC for GHGs, land and nitrogen by -46%, -62 % and +188%, respectively (Figure 4d-f). I found that shifts in the composition of livestock production were responsible for 41% (GHGs), 32% (land) and 12% (N) of these total changes in EBC. Thus the majority of the change in EBC for all three environmental impacts was attributable to RUE.

Discussion

Changing environmental burdens

Livestock production has increased rapidly to meet the demands of population growth and dietary changes[1,5,7]. To support this development, resource use, GHG emissions and pollution from synthetic fertilizers have also expanded – by our estimation, 20% for land, 74% for GHG and 820% for N (Supplementary Table 9) – despite apparent gains in certain EBCs. How these environmental impacts have changed relative to animal production is the result of multiple underlying factors (e.g., feed trade, RUE, livestock composition). This study clearly demonstrates that RUE has played a major role in altering the environmental burden of animal production. Selective breeding, higher quality diets, improved access to vaccinations and reduced exposure to extreme climate (i.e., climate-controlled industrial systems) have combined to enable these substantial improvements[7]. In just the past 30 years, advances in animal science have doubled the grain feed conversion efficiencies of chickens and pigs[1,2,12]. Because RUE is in large part dictated by technology, animal physiology and access to feed, affluent regions have been able to produce animals more efficiently[4]. Indeed, this is apparent for animal calories from feed sources where the land productivities of many developed countries were markedly higher than much of the developing world (Figure 1). These high productivities in many industrialized countries also highlight a 'livestock yield gap' for many developing countries where there is a large potential to increase livestock yields in the coming decades.

While RUE of animals has been a more important contributor to changing livestock's environmental burden, our analysis shows that the changing composition of livestock production has played a significant role as well. Though changes in livestock composition were modest in influencing nitrogen use intensity, this factor contributed considerably to minimizing the per calorie impact in terms of GHG emissions and land requirements. For both of these environmental metrics, much of this contribution can be explained by declining relative contribution of cattle (Supplementary Figure 3), whose methane emissions substantially influence the overall GHG emissions from the livestock sector and whose land requirements still currently equal 74% of all area used for animal production. This is not surprising, as the transition towards intensive systems goes hand-in-hand with the shifts in composition from beef to chickens and pigs[22]. Industrialization is responsible for much of this transition, having steadily lowered the prices of non-ruminant products and, in turn, shifted consumption patterns significantly[12]. Yet, while these shifts have led to certain improvements in EBCs, this switch towards non-ruminants has also raised concerns related to disease risks and animal welfare[22].

Environmental trade-offs and impact displacement

The intensification of livestock production has led to important trade-offs in EBCs, with lower land and GHG footprints due to the predominance of non-ruminants and increased per calorie demand for nitrogen (and irrigation water[1,2]) to support rising feed requirements. While the impact of GHG emissions is by and large global, other environmental consequences are more limited to the location where the animal or feed production occurs. A globalizing livestock sector has meant a separation of feed's production and consumption and, combined with increasing global affluence, may have enhanced the displacement of land use and land use change into producer countries[23,24]. This shifting of impacts is apparent in our quantification of the virtual trade of resources where the countries producing the feed are the ones assuming many of the environmental costs (e.g. ref. 25) (Figure 3b; Supplementary Figure 4). In addition to virtually exporting environmental costs through the purchase of feed, importing countries can also conserve their locally available resources for other uses, potentially attain levels of livestock production above the local livestock carrying capacity and minimize the influence of local climatic variability and extremes. Though the trade of feed does not appear to impact domestic calorie provision in the main exporting countries[26], the increased use of feed still does not guarantee the self-sufficiency of animal calorie production (i.e., domestic production of animal calories exceeding domestic demand) for the importing country. This is especially apparent in Asia where – despite rapid increases in both productivity (Figure 1) and feedcrop imports (Figure 3a) – large imports of animal goods are still required to meet regional demand (Supplementary Figure 2; ref. 15). For places importing animal products, embedded nutrients in those products can also have environmental impacts[17], highlighting the fact that the virtual trade of resources associated with livestock production occurs at two levels: the trade of feed and (to a lesser extent) of the animal production itself[25,27].

Food security implications

Changes in the livestock sector have also had important implications for global food security and crop availability. While increased grain-feeding has contributed significantly to improving livestock yields, this intensification has required the use of lands of high agricultural value instead of using areas not suitable for crop production (e.g. rangelands). Recent work quantifying the competition for crop use as a result of this intensification found that 4.9 billion people could be fed by the crop calories currently used for animal feed and that eliminating beef from the diet would result in a crop calorie savings of 2.13×10^{15} kcal[5]. Another recent study demonstrated how global diets link improved human health to environmental sustainability[14]. The authors showed that diets which reduce incidences of cancer, heart disease and diabetes are also the ones composed of foods which are less resource-intensive to produce, translating into significant land sparing and GHG reductions. Thus, a combination of modifying diets while encouraging healthy choices appears to be a promising way to minimize the sector's environmental burden while meeting increasing human demand.

Counter to these recommendations, increasing global affluence is expected to make future diets more meat-demanding[12,14,28] while cultural and economic factors leave consumption patterns entrenched. Thus while efforts should be made to reduce a consumer's dietary footprint, utilizing a suite of options is the most realistic for minimizing livestock's impact. As one alternative to modifying diets, Havlik et al.[8] showed that closing crop yield gaps can at the same time help to improve livestock efficiency due to higher feed quality. However, this approach used in isolation would likely lead to an increase in the overall resource demands of the livestock sector, as the rates of historical improvement in EBC have yet to realize a decrease in absolute resource use and emissions. Another important avenue for reducing livestock's footprint is through the reduction of waste along the food supply chain[29]. As West et al.[30] showed, the waste of animal products can have a much larger impact on available calories because of the inefficient conversion of feed to animal calories. The point along the food supply chain in which waste of animal products occurs differs greatly between regions. For example, in sub-Saharan Africa, approximately 26% of initial meat production is wasted before even reaching the consumer, a consequence of high animal mortality and insufficient storage and transport infrastructure. Conversely, more than half of the waste of meat and dairy products in Europe and North America occurs at the level of the consumer (e.g. retailers, households). Finally, much of the environmental burden of livestock production is due to its heavy reliance on feed and the resources required to produce those crops. Making advanced technologies (e.g., fertilizer banding) more readily available and affordable could thus contribute substantially to avoiding the inefficient application of fertilizers for feed production that continue to occur. Indeed, the gradual but consistent decrease in the EBC of nitrogen is encouraging in this regard and suggests that wasteful application of synthetic fertilizers has been reigned in to a certain extent (Fig. 4e).

Conclusion

The current structure of the global livestock system – a system which has placed greater reliance on feed – appears to be largely unsustainable. Continued growth in human demand is expected to outpace improvements in EBC and, in turn, lead to greater resource demands and environmental impacts of the livestock sector. In addition, that a large number of countries import feed for animal production raises concerns about their long-term food self-sufficiency[31], especially considering that producer countries may be less willing to export crops in the coming decades as a result of demographic growth and climate change[32]. Current knowledge points towards a global food system that has become increasingly homogenized and more susceptible to shocks as a result[33]. Exemplifying this vulnerability for the livestock sector, globalization and the transition to intensive systems have been accompanied by the increasing risk of infectious diseases and antibiotic resistance[22]. It is critical that countries seek to adapt domestic animal production to minimize reliance on trade and improve resilience by maintaining a balance of species. A variety of production systems, plant biomass sources and consumption patterns all offer benefits toward achieving sustainable intensification. This mirrors recent thinking that both addressing supply- and demand-side trends simultaneously[9,34] as well as better integrating the nutrient and energy cycles of crop and animal production[35,36] are the most promising pathways to securing livelihoods, food and environmental stewardship. As this study shows, countries can integrate environmental and food security considerations in order to better understand how improvements in one aspect of livestock production and consumption may result in adverse consequences in another. In doing so, each country can ultimately tailor a suite of approaches most appropriate for its unique socio-ecological landscape, aimed at minimizing livestock's environmental burden while maximizing food security.

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Figures

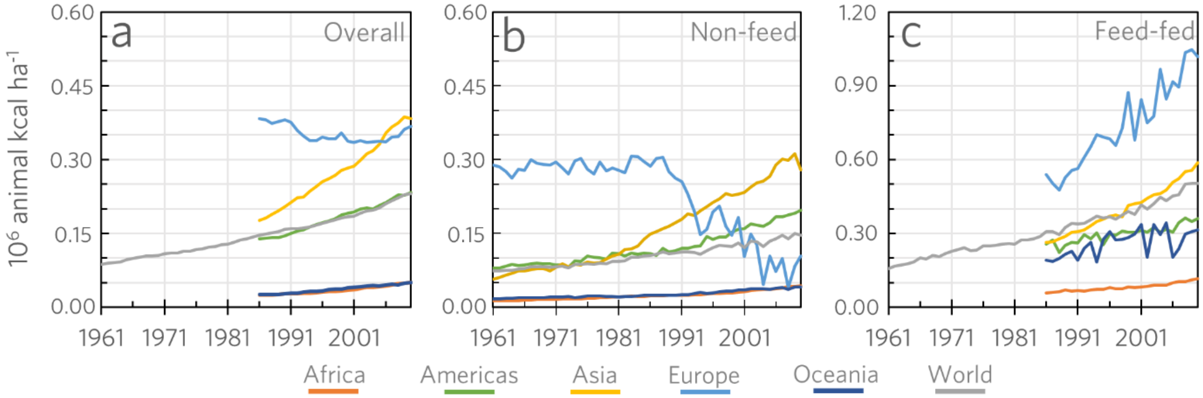

Figure 1. Trends in animal productivity of land. Regional changes in the animal productivity of land (kcal ha^{-1}) for (a) total animal calorie production and production derived from (b) non-feed and (c) feed sources. Regional data for 'overall' and 'feed-fed' productivities begin in 1986 because this was the first year for which detailed trade information was available. The reader should also note that the y-axis scale for panel *c* is different from that of panels *a* and *b*.

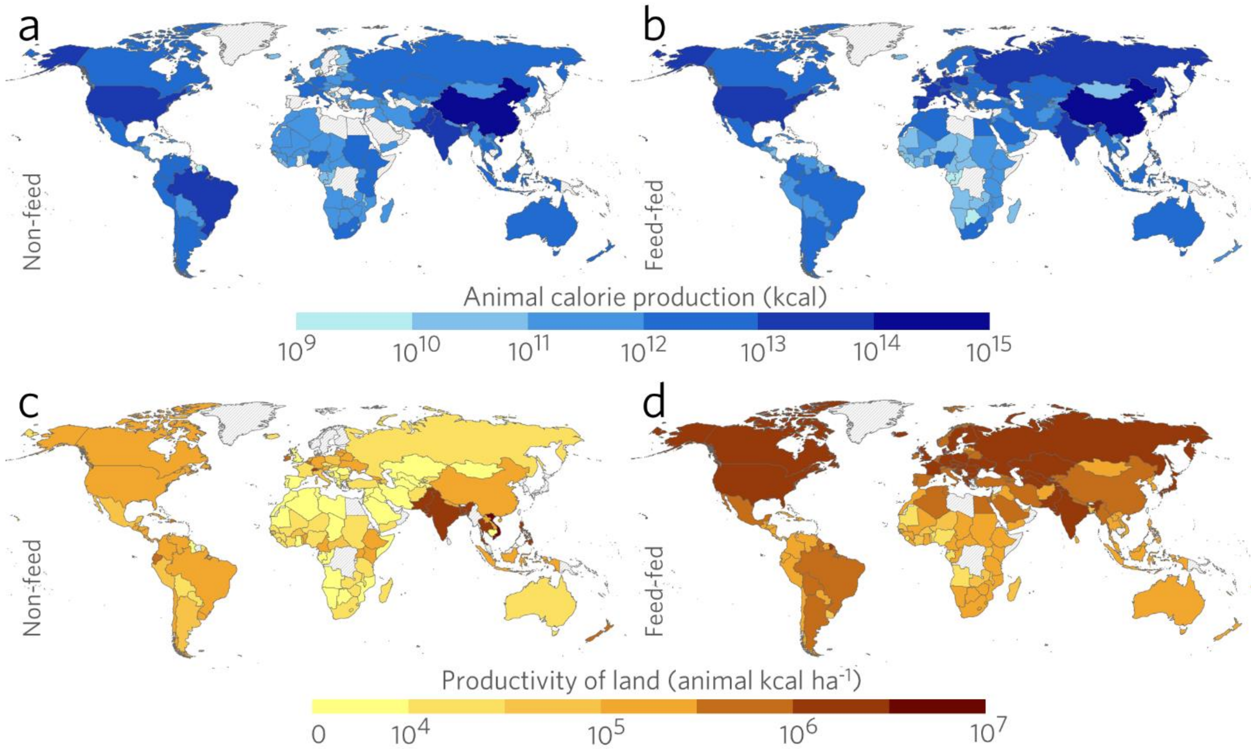

Figure 2. Animal production and productivity of land. a,b) Animal calorie production from feed and non-feed sources (year 2000-2009 average). Only feed-fed and non-feed calorie production values above 10^9 kcal are shown. Non-feed production values are only shown for countries with a pasture area greater than $0.5x10^6$ ha. Countries with grey cross-hatching either fell below these thresholds or had no data. c,d) Animal productivity of land for feed and non-feed sources (year 2000-2009 average). Values for these maps are presented in Supplementary Table 10.

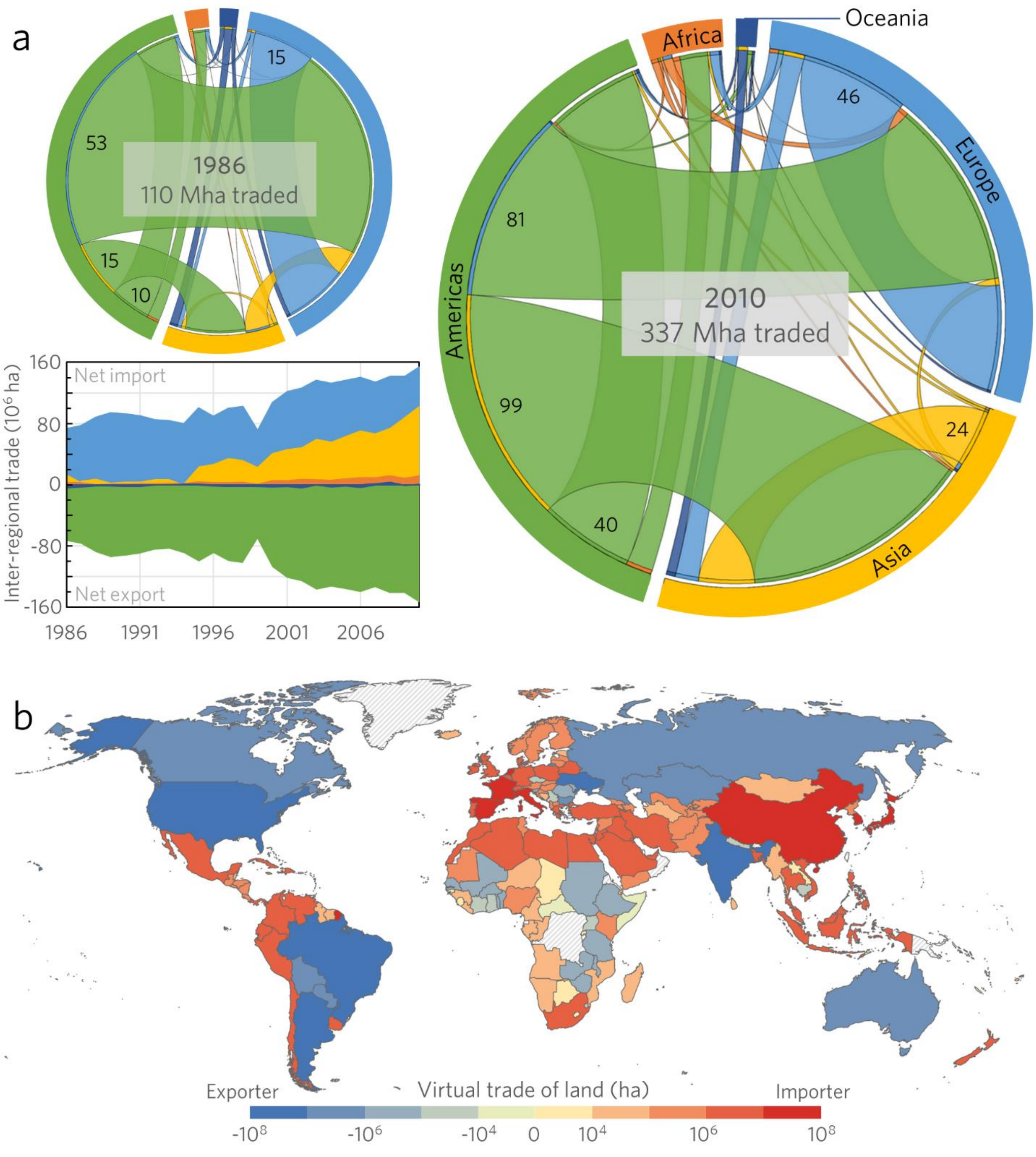

Figure 3. Trends and patterns in virtual land trade. a) Inter-regional flows of virtual land via feed trade. The color of each band corresponds to the exporting region, while the numbers within major bands represent the magnitude of the virtual flow of land (in Mha) along that link. Circle areas are scaled to the total virtual land traded in 1986 and 2010. Inset plot shows the steady transition of virtual land's destination, from almost entirely Europe in 1986 to roughly equal parts Europe and Asia in 2010. b) Net virtual trade of land for feed by country (year 2000-2009 average). Values are reported in Supplementary Table 10.

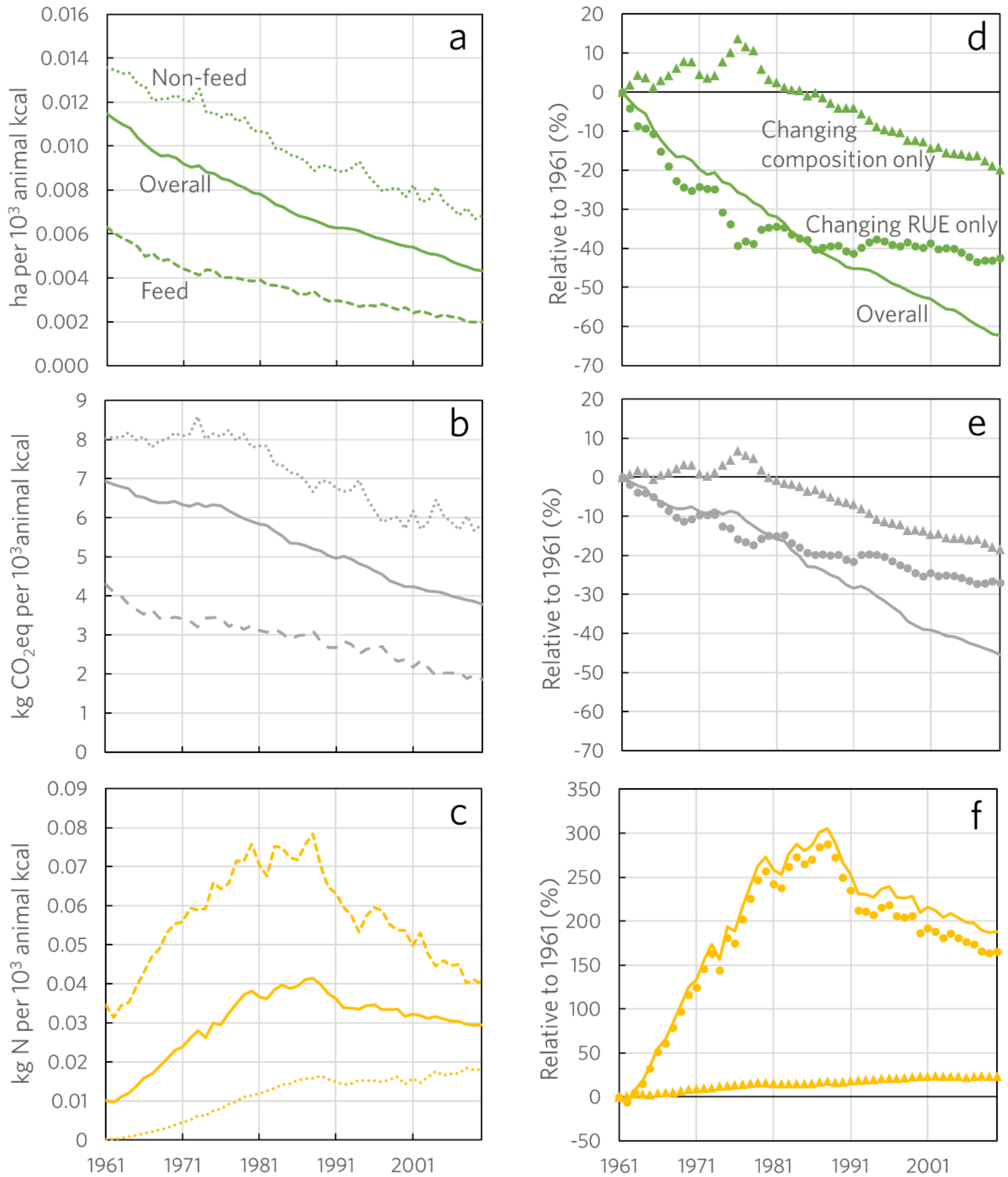

Figure 4. Changing EBC for land, carbon and nitrogen. a-c) Changes in EBCs for feed-fed systems, non-feed systems and total animal calorie production. d-f) Relative change in EBC as contributed by changing RUE of individual animal products and changing composition of livestock production (i.e., greater contribution of non-ruminants relative to ruminants). For the 'Changing composition only' scenario, resource use efficiency of each animal product was held constant at year 2010 levels to determine the contribution of changing livestock composition. This relative change was subtracted from the overall relative change in EBC to determine the importance of RUE changes (i.e., 'Changing RUE only') in altering EBC.

MODERATING DIETS TO FEED THE FUTURE

Abstract:

Population growth, dietary changes and increasing biofuel use are placing unprecedented pressure on the global food system. While this demand likely cannot be met by expanding agricultural lands, much of the world's cropland can attain higher crop yields. Therefore, it is important to examine whether increasing crop productivity to the maximum attainable yield (i.e. yield gap closure) alone can substantially improve food security at global and national scales. The study presented here shows that closing yield gaps through conventional technological development (i.e. fertilizers and irrigation) can potentially meet future global demand if diets are moderated and crop-based biofuel production is limited. In particular, this study finds that increasing dietary demand will be largely to blame should crop production fall short of demand. In converting projected diets to a globally adequate diet (3000 kcal/cap/day; 20% animal kcal) under current agrofuel use, this study also finds that ~1.8 to ~2.6 billion additional people can be fed in 2030 and ~2.1 to ~3.1 billion additional people in 2050, depending on the extent to which yields can improve in those time periods. Therefore, the simple combination of yield gap closure and moderating diets offers promise for feeding the world's population but only if long-term sustainability is the focus.

Introduction

The world's population has rapidly increased over the past two hundred years and is projected to continue doing so to century's end, when the global population is expected to reach a maximum[1]. Predictions of the level and timing of this maximum are typically based either on historic demographic data[2] or on limiting global resources[3]. The first approach leads to the question: 'How much of Earth's resources will be needed to support these people?', while the second asks: 'How many people can these resources support?'. When the population reaches the maximum size allowed by the available resources, one of two occurrences can be the result: either demographic growth ceases as the result of a Malthusian ceiling[2,4], or innovation and adoption of new technology raises the ceiling of resource availability[5,6]. To prevent any sort of forcible natural constraint on population, humans have historically preferred the latter option[2]. Thus, it is likely that technology will keep intervening to increase the ceiling until population can stabilize as an effect of demographic and developmental drivers[7]. With potential for agricultural expansion limited[7,8,9], increasing crop productivity towards the maximum attainable yield (i.e. yield gap closure) offers an important avenue by which technology can substantially improve global food supply[9,10], though the literature has reached the consensus that increasing crop yields alone will be largely insufficient to meet future demand[7,9,11]. This is because population growth, dietary changes and biofuel use will play an important role in determining human demand and whether increases in crop supply can keep pace. Thus a combination of four main solutions[7,9] has been put forward: 1) agricultural intensification (i.e. increasing yields and harvests on current cropland), 2) increasing resource use efficiency and sustainability (e.g. fertilizers, irrigation water, soils), 3) reducing food waste and 4) moderating diets (especially the demand for meat and animal products).

Here I use an integrated calorie-based approach to examine the effect of diets (both current and projected) on the global carrying capacity, as constrained by domestic (country-level) crop production. By examining yield gap closures under different diet and biofuel use scenarios, I seek to accomplish two objectives. First I seek to demonstrate to what extent fertilizers and irrigation can increase the ability of global crop production to support the world's population to mid-century. Second, I examine the effect of moderating diet on meeting current and future demands when combined with yield gap closure. Unlike previous analyses[12,13,14], I relate food to population size using country-specific dietary requirements[15], account for the caloric conversion from plant to animal calories[16] and, most importantly, consider the number of people able to be fed under current, future and globally moderated, calorie-adequate (3000 kcal/cap/day; 20% animal) diets. Since a country's diet can substantially differ from the global average or reference diet typically used in these previous studies, the novelty of our study lies also in considering several detailed diets and how many people can be supported under these scenarios, in comparing these estimates to population growth at both global and national scales, and in examining the self-sufficiency of each country's domestic food calorie production. Moreover, in order to consider the transformation from plant to animal calorie and its efficiency, I have introduced a proper conversion factor. Our approach also allows for comparisons between the food-supply benefits of a calorie-adequate global diet versus the current and projected distributions of diets under different biofuel use scenarios and levels of yield gap closure. Thus, I ask to what extent different levels of yield gap closure of major food crops taken in combination with moderated diets (as well as reduced biofuel use) can potentially contribute to global food security by 2030 and by mid-century.

Methods

We consider agricultural production data for year 2000 yields and yield gap closures for 16 major food crops (Supplementary Table A1)[10]. Country-specific information on average individual diet came from FAO Food Balance Sheets[15] for the year 2000. Data on total animal-source (i.e. meat, eggs, dairy, animal fat and offals) production for each country were obtained from FAOSTAT[15]. Caloric data (i.e. crop-specific energy to weight ratio) from the Food Balance Sheets (FAOSTAT) were used to convert the Mueller production data for each major crop (excluding cotton) and the FAO animal production data to total caloric production by country.

To calculate the number of people who could be fed under different diet and yield gap closure scenarios I compare the major crop production to the per capita demand of those crops. The latter is therefore calculated considering only the portions of the diet (in terms of calories) contributed by the major crops, their processed derivatives and feed-produced animal products (excluding fish) but excluding all the food products (e.g., fruit, rangeland meat) whose production is not supported by the major crops. A caloric conversion factor (for animal-source calories derived from feed as indicated in the Food Balance Sheets) was calculated for each country for the production of animal-source calories. Using energy input ratios[16] (i.e. how many plant calories are required to produce one animal calorie; Supplementary Table A2), the conversion factor ($q > 1$) was calculated as a calorie-weighted average for the animal products contributing to total animal production within each country (see Supplementary Table A3). Major crop production used as feed was divided by this country- and crop-specific conversion factor to obtain the total equivalent animal calories from feed. The ratio of feed-produced animal calories to total animal-source calories (r) was then multiplied by the total per capita consumption of animal calories minus seafood animal-source calories to obtain the portion of the country-specific diet contributed by feed-produced animal products. This portion was then multiplied by q to give the equivalent vegetal calories needed to support the animal portion of the individual diet. Similarly, for the vegetal portion of the diet, the ratio (k) between the dietary calories from major crops (including their derivatives) and the total vegetal consumption was calculated. The country-specific factors q, r and k were used in all diet scenarios (Supplementary Box A1). For each country, the fraction of production available for human consumption (w) was calculated for all foods from the Food Balance Sheets as the sum of the amounts categorized as 'food', 'feed' and 'processing' divided by the 'domestic supply' (i.e. Production – Exports + Imports). This was held constant through all scenarios where waste was taken into account. In scenarios of no waste, the amount of each crop categorized as 'waste' in the Food Balance Sheets was added to 'food', 'feed' and 'processing' while calculating w. In introducing the 'domestic supply' value (and the 'import' and 'export' values needed to calculate it), I should note that inconsistencies are possible at national and regional scales within the FAOSTAT agricultural trade data (see ref. 15 for in depth description). However, since much of our study is focused on global calorie production relative to demand, the presence of these inconsistencies does not impact our findings. The fraction of oil palm, maize, rapeseed, sunflower and sugar cane currently used for biofuel was calculated from the Food Balance Sheets as the amount of each crop used for 'other use' divided by domestic supply. For sugar cane, this value was obtained using the values of centrifugal sugar. Oil palm, rapeseed and sunflower were assumed to be fully converted to their oils. The country-specific fraction of conversion efficiency for each of the three oilcrops was calculated as total production of that oil divided by total production of that oilcrop, using production data from FAOSTAT[15]. For countries lacking these data or with conversion efficiencies greater than one (due to processing without domestic production of the raw oilcrop), the global fraction for that oilcrop was used. This global fraction was calculated as the production-weighted average of countries with existing data and with conversion efficiencies less than one. Projected increase in biofuel production increase under a 'business-as-usual' scenario was linearly extrapolated from current and projected production for ethanol and biodiesel production[17]. Data on the current extent and depth of undernourishment by country were obtained from the World Bank's World Development Indicators database[18].

Four diet scenarios were considered: 1) current country-specific diet (as described above from FAO food balance sheets), 2) the FAO recommended calorie-adequate diet (i.e. 3000 kcal/cap/day; 20% animal calories), 3) projected diet for the year 2030, and 4) projected diet for the year 2050. These projected country-specific diets were calculated using regional values from Alexandratos and Bruinsma[19], where the percent increases in total and animal calorie demand for 2000 to 2030 and for 2000 to 2050 were then applied to the current (circa 2000) country-specific demand of the countries contained within each region (Supplementary Table A3). For expansion onto land originally used for cotton, the fraction of total land used by each major crop was calculated for each country, and the area of cotton land was divided accordingly. The recommended FAO calorie-adequate diet of 3000 kcal with 20% animal protein was treated with the factors q, r and k and subsequently used for the calorie-adequate diet (see Supplementary Box A1). Projected changes in diet for the years 2030 and 2050 took into account the percent increases both in total dietary calories and in animal-source calories[19]. Based on sub-region (Supplementary Table S3), the current total and animal-source caloric intakes for a person in each country were multiplied by the appropriate regional percent increases (as calculated from ref. 19) to give the projected dietary demands of 2030 and 2050. Neither of the future diet scenarios considers current or projected depths and extents of undernourishment. Population estimates were taken from the UN Population Division[1]. The number of people who could be fed under different diet and yield gap closure scenarios is presented in the supplementary information (Supplementary Table A4). Diets of adjacent countries were used for countries for which dietary information was not available (Supplementary Table A5).

Results

Our estimate for the number of people able to be supported by global production of major crops in the year 2000 is 5.83 billion people (when accounting for waste and biofuel use). This is consistent (4.9% difference) with the UN estimate of 6.13 billion people[1]. Under the current scenario (diet, waste and biofuel use in 2000), I calculate that complete yield gap closure would support 3.94 billion additional people. This represents a gain in vegetal production of major food crop calories of 3.50×10^{15} kcal (compare to 5×10^{15} kilocalories for 95% closure without waste or biofuel use calculated previously[9]). Under the status quo, this level of production would be more than capable of feeding the world in 2030 (8.42 billion people) and at mid-century (9.55 billion people). However, this does not consider future changes in diet and biofuel use (Figure 1), nor the rate at which yield gap closure can occur[11]. If biofuel production were to increase in a 'business-as-usual' scenario[17] with the projected diet of 2030, the population able to have their dietary needs met at complete yield gap closure would be substantially reduced to 7.19 billion people, a deficit of ~1.23 billion people globally. Conversely, if a calorie-adequate global diet is consumed in 2030, closing the yield gap would support 9.32 billion people, even if biofuel production continues to increase as it has.

While the global average daily diet was ~2700 kcal per person in 2000, diets varied widely by country, from Eritrea (1506 kcal/cap/day, 8% animal) to Austria (3809 kcal/cap/day, 33% animal). When I consider a transition from current diet to a calorie-adequate diet with current biofuel use and waste, ~820 million additional people can be fed, showing that modifying diets (in terms of calories) to be more globally uniform can substantially improve the number of people fed[7,9] and that caution must be used when drawing conclusions based on average global diets. Our findings also add to a recent study (~110 million additional people assuming a 3000 kilocalorie diet) based on improved agricultural use of water resources[12] and indicate that other yield-increasing inputs (e.g. fertilizer and pesticide use) may need to feature more prominently in closing yield gaps, as has been the case throughout the 20th century[20].

We estimate ~56% of the total production of (non-seafood) animal products originated from rangeland in the year 2000, representing a significant contribution to diets globally[7]. Also, our global estimate (derived from the FAO Food Balance Sheets) of wasted food (~14%) agrees well with the ~16% previously found for lost or wasted food within the food supply chain[13].

Lastly, by comparing the number of people potentially supported by domestically produced calories with the current (year 2000) population of each country, I determined which countries are currently most dependent on imported calories (Figure 2). I found that ~917 million people (~15% of global population) needed foreign-produced calories in the year 2000 (figure 2), a value that agrees well with Fader and colleagues[21]. Moreover, the countries with larger populations also tended to be more self-sufficient in terms of domestic crop production. Furthermore, when diets are adequate globally, a greater number of countries can achieve self-sufficiency in terms of calorie production. Specifically, the percent of countries in obvious calorie deficit (and which are therefore reliant on food trade) modestly decreases from 77% under the current diet to 70% under a calorie-adequate global diet.

Discussion

The relationship of humankind to the planet's natural constraints is dependent on human choices relating to diet, energy and demographic changes[3]. Technology has continually played a role in increasing the planet's carrying capacity, allowing the combination of agricultural expansion and elevated yields to meet increasing human demand[5,6]. However, decisions on how to feed a global population have become more difficult as the environmental impacts from increased agricultural production continue to mount. Further agricultural expansion exemplifies this dilemma in that it may provide immediate benefits to food availability but compromises the ability of ecological systems to maintain biodiversity and carbon storage[9]. Moreover, many agree that present global consumption far exceeds long-term sustainable levels[22,23,24].

These findings make apparent the current dependence of many countries on global food trade and the potential for this dependence to increase (see also ref. 21). As seen with the trade of virtual water, a greater dependence on trade will likely decrease societal resilience[25]. Further, in response to recent spikes in food prices resulting from droughts or other climate extremes in years of increasing demand for agricultural products, the governments of exporting countries have banned or limited their exports to ensure their own food security (e.g. ref. 21). , Thus the food security of import-dependent countries (much of the world is reliant on food trade to meet domestic needs) is strongly affected by the uncertainty and unreliability of the food trade market. With this in mind, our study's comparison between domestic calorie production and demand thus asks what would happen if trade did not occur. This in turn sheds light on self-sufficiency (both present and future) of domestic calorie production under a number of scenarios in terms of domestic crop production and shows that in some cases improved yield can potentially increase food security (particularly in places of slow population growth). In all of this, our study evaluates countries' self-sufficiency considering the very extreme case that international food trade would cease completely, which is clearly unlikely. Overall, our findings reinforce that simply closing yield gaps is not sufficient to meet future dietary needs under a variety of scenarios, regardless of the rate of yield gap closure[11]. While closing yield gaps alone is largely insufficient, I do see that, when yield gap closure is combined with a calorie-adequate global diet, these two approaches alone can largely meet global demands to mid-century (Figure 1) and that minimizing the use of crop-based biofuels further improves the outlook. In some cases moderating diets can also serve to meet a nation's calorie demand domestically. To achieve this greater self-sufficiency however would entail a reduction in per capita demand and would likely prove difficult given the economic, social and cultural implications of diet. Overall, it is apparent that, while moderating diets can reduce global demand, food trade will still need to feature prominently under such scenarios to ensure food access and security. In contrast to the global calorie-adequate diet, I find that projected changes in diet as a result of increasing global development and affluence will likely result in greater food insecurity globally, as even the highest attainable yields cannot meet the appetite of a rapidly growing population over the next several decades. In addition, the fact that greater affluence leads to richer diets is compounded by recent trends in the livestock sector towards intensification (i.e. grain-fed, high-density animal production)[26,27]. From a resource perspective, it is encouraging that much of this intensification (and the increase in animal production overall) is attributable to more resource-efficient animals (e.g. chickens, pigs). Yet while a greater reliance on these non-ruminant species with small area requirements may alleviate stress on grazing systems, this can also mean increased competition between food-crop and feed-crop production for land and water resources[28] and further separation of consumers from the environmental impacts of their food production [e.g. ref 29]. If supply in fact becomes constraining as a result of livestock production and other unprecedented demands, this may mean that future biofuel dependence and dietary demand will need to decline, or they may profoundly impact food availability for human consumption in the near future. As a brief aside, I should also note that other important ways to potentially improve global food supply are through 1) increasing the frequency of crop harvesting, where it has been shown that many regions have large "harvest gaps"[30] and 2) reducing food waste, where it has been shown that halving food losses could feed 1 billion additional people[13].

In this study, yield gap closure is achieved by increasing nutrient and water availability through investments in fertilizers and irrigation technology, a process often delayed by social, cultural, technical, and financial obstacles. Moreover, crop yields remain susceptible to stagnation of actual[31] and potential yields (i.e. yield ceiling)[32] and the effects of climate variability and change (e.g. ref. 33). Changes in growing season length and drought occurrences thus constitute serious threats to the predictability and reliability of global agricultural production[33,34]. On the other hand, with recent increases in large scale land acquisitions in the developing world (and the rapid improvement in agricultural technology that they can bring), there may be a global potential for major crop yields to improve more rapidly than historically observed. This may mean that crop production is better able to attain the doubling in supply that has been predicted to meet mid-century demand[11,35]. In highlighting these various additional influences on future crop yields I should clearly state that the effects on global food security of climate change, carbon dioxide fertilization[36], genetically modified organisms (GMOs) and access to crop production[37] were not considered in this study. In addition, feedbacks resulting from potential social (e.g. modified diet in response to availability), economic (e.g. increased food prices) and policy (e.g. biofuel additive cap[38]) responses to a strained food supply were not considered.

Use of food crops as biofuels is another significant factor influencing future food security and demonstrates that the outlook for meeting human demands largely depends on the decisions made now regarding biofuel policy and the pace of and extent to which they are implemented[6,39]. Providing possible insight into how major biofuel producers (and societies in general) may be expected to prioritize agricultural resources in the coming decades, recent European legislation placed a cap on the amount of food-based biofuel added to transportation fuel[38]. Thus, due to the relatively rapid changes in policy that can occur regarding the use and production of crop-based biofuels, I do not consider biofuel scenarios for 2050.

Though changes in biofuel policy can improve the outlook for meeting future human demand, I have set out in this paper to examine the consequences of dietary change in particular. Future diets will be characterized by transitions to greater percentages of meat, reflecting economic and developmental improvements[35]. As diet has social, cultural and economic implications, encouraging smaller proportions of meat may be one of the more difficult avenues to pursue in seeking to decrease demand[7], but can also offer some of the largest benefits in increasing the number of people able to be fed[6,14]. This is particularly true if diets transition towards less demand for animal-source products, as the calories from these products require substantially more resources to produce. Alternatively, it may be possible to rely less on grain-fed animal production, increase the animal production of rangelands, and enhance the reliance on fisheries (particularly aquaculture) to ensure resource savings (e.g. ref. 40). In this way, a greater amount of cereals will be available for direct human consumption (e.g. ref. 14). Though this offers promise for global food security, it also appears that progression towards an adequate diet for countries below this recommended level [41] conflicts with the need to rapidly feed more people.

The United States and Brazil, two of the world's major biofuel producers, serve as cautionary examples of how current calorie surplus can be quickly exhausted in the future as a result of energy choices, dietary behavior and demographic change. Largely used to meet the country's high demand for animal-based calories, maize in the United States made up ~86% of plant-based feed (by weight) in 2000 and is increasingly diverted for biofuel production. Thus, as a greater percentage of maize production is used for energy, the remaining percentages for animal production and food for direct human consumption are reduced, and could result in less willingness to export to countries dependent on this production. A reduced ability to export major crops may also occur in Brazil. Here, increases in yield alone would be insufficient to prevent Brazil's transition to calorie deficit, if biofuel production and dietary demand continue to increase. Unlike the United States where the frequency of crop harvesting is close to the maximum, Brazil has the potential to more than double the frequency with which it harvests crops[30] and can in this way greatly increase domestic calorie delivery. Under this scenario, Indonesia and Papua New Guinea undergo a similar fate due to expanding oil palm production but again also have a large potential for increasing crop harvest frequency.

China and India offer a different perspective in that they are not major biofuel producers, but population growth (particularly in India) and increased consumption of animal-based calories (particularly in China) may serve to take these countries below the threshold of self-sufficiency for domestic calorie production, even if yield improvements are realized. China has especially limited options in terms of agricultural intensification, as the country's potential for increasing harvest frequency is also low[30]. As further evidence of China moving towards maximizing its domestic resources, in 2010, it had already become a net importer of virtual water[42] and food (in tons[15]). The transition of India, however, can be expected to occur later (if at all) as it reaches its peak population some 30 years after China[1] and has more time to be proactive. More broadly, rapid population growth in Asia and Africa may exacerbate issues of food security and malnourishment[41], as improvements in crop production may not keep pace with growing demand. In addition, the impacts of climate change on domestic crop production in these most vulnerable countries are expected to become more severe with time[34,36]. However, in these regions, the large potential to increase yields and harvesting frequency (outside of China and India) offers hope in the ability to increase food supply[30]. Overall, the long-term sustainability of such agricultural practices will become a more pressing issue in the coming decades.

Conclusion

Closing yield gaps offers great benefits for additional global food supply, especially in areas of high food insecurity[41], but will likely not meet increased future global demand on its own. This is particularly true given recently observed crop yield stagnations and the potential for this to occur in more places in the future[31,32]. As in the past, new technologies and innovation will likely act to increase global food supply, but the multiple demands on the global food system dictate that yield gap closure can only *ever* be part of the solution towards meeting future needs. While population growth, dietary changes and biofuel production can act synergistically to the detriment of many countries' prospects for food security, the combination of moderated diets and improved crop yields offers great promise but can also be one of the more difficult avenues to pursue. Our approach considering country-specific dietary requirements highlights the fact that a greater focus on making dietary demand more equitable can be one of the most beneficial solutions for the prospects of global food security but can make some of the poorest countries less able to feed their populations.

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Figures

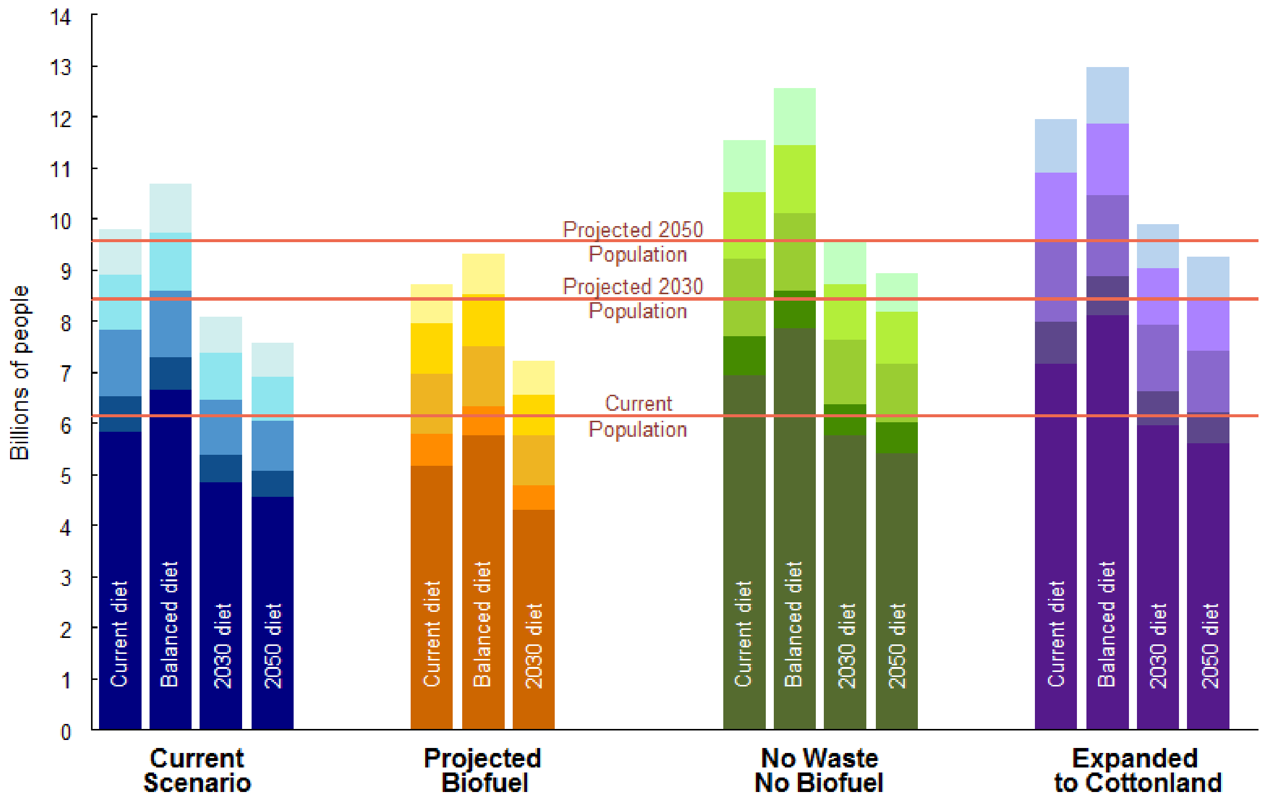

Figure 1. Yield gap closure for different scenarios of dietary change, biofuel use and waste. The five segments of each column represent the population potentially supported by domestic production under year 2000 yields and yield gap closures of 50%, 75%, 90% and 100%. For expansion onto land originally used for cotton, the fraction of total land used by each major crop was calculated for each country, and the area of cotton land was divided accordingly.

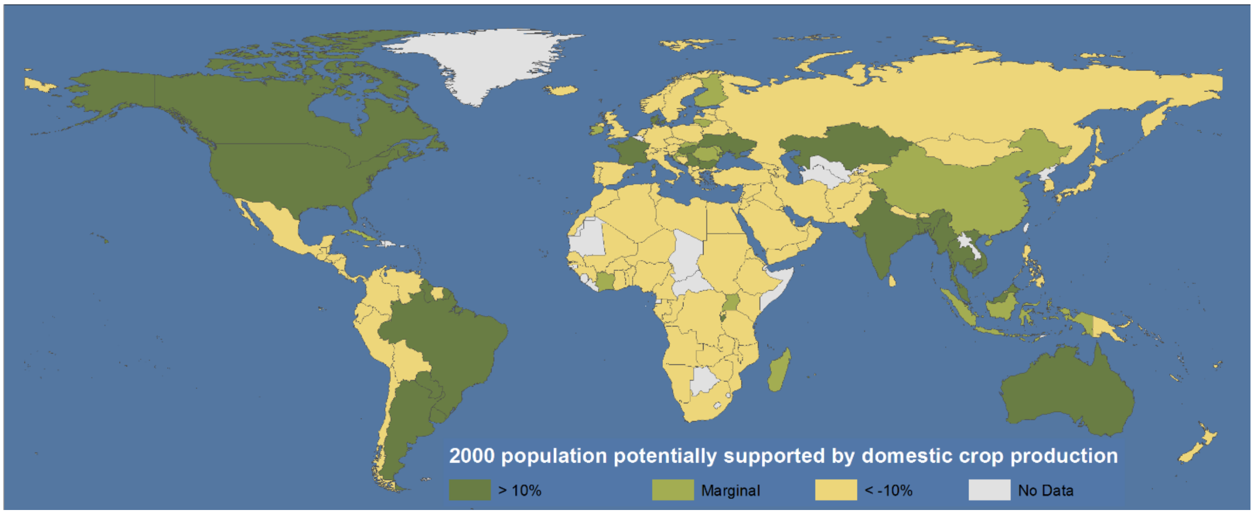

Figure 2. Countries in conditions of food calorie self-sufficiency, deficit, and marginal self-sufficiency or dependency, based on current diet, waste and biofuel use and the year 2000 yields. 117 out of 154 countries are in obvious calorie deficit. For self-sufficient countries, the domestic caloric production from crops is at least 10% greater than what is required by the domestic population. For deficit countries, the domestic caloric production from crops is at least 10% less than what is required by the country's population.

LAND GRABBING: A PRELIMINARY QUANTIFICATION OF ECONOMIC IMPACTS ON RURAL LIVELIHOODS

Abstract:

Global demands on agricultural land are increasing due to population growth, dietary changes and the use of biofuels. Their effect on food security is to reduce humans' ability to cope with the uncertainties of global climate change. In light of the 2008 food crisis, to secure reliable future access to sufficient agricultural land, many nations and corporations have begun purchasing large tracts of land in the global South, a phenomenon deemed "land grabbing" by popular media. Because land investors frequently export crops without providing adequate employment, this represents an effective income loss for local communities. This study examined 28 countries targeted by large-scale land acquisitions (comprising 87% of reported cases and 27 million hectares (ha)) and estimated the effects of such investments on local communities' incomes. This study found this phenomenon can potentially affect the incomes of ~12 million people globally with implications for food security, poverty levels and urbanization. While it is important to note that this study incorporates a number of assumptions and limitations, it provides a much needed initial quantification of the economic impacts of large-scale land acquisitions on rural livelihoods.

Introduction

Population growth, dietary changes and increasing use of crop-based biofuel are placing ever greater demand on food production and its requisite resources[1]. In addition, climate change is projected to adversely affect reliable and sufficient food supply in the future[2]. These changes in the demand and supply of agricultural products threaten food and water security as well as sustainable livelihoods. Due to these demographic and environmental pressures and the 2008 food crisis, many nations and corporations with the requisite capital are making large-scale investments in agricultural lands both domestically and abroad to either accumulate a reliable reservoir of land and water resources in the event of increased climatic uncertainty or to speculate on the price of cultivatable lands[3]. While the potential benefits (e.g. insurance against food price shocks, increased global food supply) of these deals may be apparent, such transactions often take place at the expense of and without informed consent from the prior land users[3,4,5]. This fact has been the source of wide discussion in the land rush literature[6,7,8,9] but is all too often overlooked by the involved governments and investors. These large scale land acquisition projects often emphasize the rapid increase in yield that they can produce and the additional employment they can provide. However, the benefits of this additional agricultural production are often not felt locally[5,10], so that the loss of access to land can ultimately spell significant dietary, social, cultural and economic consequences for rural communities in the targeted areas[8,9]. Given the lack of transparency in many of these transactions, it is understandable that a quantitative literature on the human impacts of this phenomenon is sparse. Despite this apparent difficulty, several studies have been able to broadly assess the amount of land appropriated (e.g. refs. 3,11). However, knowing the area controlled by investors can only inform the discussion so much, and a more pointed quantification of the specific impacts of the global land rush is now necessary. That is why steps are now being taken in the land rush literature to turn the focus from studies purely assessing the area affected by such land deals toward quantification of the potential environmental and human impacts[12,13,14]. One such study sought to quantify the potential for these land deals to impact malnourishment in the affected areas, estimating 200 - 300 million people at risk of greater food insecurity as a direct result[15]. Though this reduced ability to feed people locally is important to consider, it is only one way in which rural communities may experience the impacts of this global land rush. I focus here on a single question, namely: how many people in rural communities of targeted areas may potentially experience income loss as a direct impact of these agricultural land deals? I argue that, since the communities in these areas rely on agriculture for income, the loss of access to land and water resources as a result of land deals represents an inability to produce household income. While I only quantify this potential impact in lands intended for food crops, I should also note that large scale land acquisitions can occur for several other reasons. For instance, recent increases in demand for agricultural land for biofuels is an effect of new energy policies[16,17] aimed at curbing the increase in atmospheric CO_2 concentrations. Further, some large-scale investments in forested land can be driven by prospects of profitable investments in the carbon credit market for climate change mitigation[18,19,20].

Targeted countries typically have lower levels of development and economies heavily reliant on the agricultural sector, in terms of both employment and value of domestic product (Figure 1), making the livelihoods of their citizenry especially sensitive to climatic change, land degradation and this recent global land rush[21]. Specifically here, I consider how targeted land resources that would otherwise be used for local crop production translate into a reduced ability to sustain the livelihoods of the current population in the affected areas. This is especially important considering that rural households in agriculture-based economies are limited in their opportunities for non-farm employment unrelated to agricultural production[17,22,23]. The income lost from targeted agricultural land represents a reduced ability of the area to support a certain number of people. Thus, while there may be various contributing factors to the problem, the sole impact I explore here is the income loss by rural communities as a result of large scale land acquisitions and how this impact varies across the most targeted countries. By calculating the total lost income due to confirmed large-scale land deals, I examine the portion of a country's population with the *potential* to be directly economically impacted by these land deals and briefly suggest (while citing the limited available evidence) that this may result in increased urbanization and human migrations in order for rural communities to diversify their incomes[17,24,25]. This study provides an *initial* but much needed quantification of the number of rural people whose livelihoods may be potentially impacted by large-scale land acquisitions. By providing empirical support, the intent of our work here is to act as a stepping stone for further studies with more definitive conclusions and to direct the attention of land deal research toward better quantifying the impacts of land deals on rural populations. Just as with tropical deforestation or urbanization, the issue of large scale land acquisitions is a rapidly evolving phenomenon[26]. This is compounded by the fact that information on land deals and their rural economic impacts suffers from a lack of transparency[5,13]**.** Yet despite the difficulty that these issues can present in staying current with the dynamics of the phenomenon, our study provides a novel alternative approach with the potential to fill an important knowledge gap in our growing understanding of large scale land acquisitions and their many possible impacts.

Methods

We study the 28 countries (Table 1) most targeted by large-scale land acquisitions (comprising 87% of reported cases and 27 million hectares (ha)). I define large scale land acquisitions as transactions that target agricultural areas and that entail the transfer of rights to use, control or ownership through sale, lease or concession to commercial farming. Based on current yield scenarios[27], country-specific crop yields for the year 2000 (which reflect national average yields before the land deal) were multiplied by land areas under contract from the new (June 2013) Land Matrix database[28] to calculate the agricultural production for each edible crop. Recently, criticism has been raised towards quantitative studies on large scale land acquisitions relying on previous data sets of large-scale land acquisitions[12,29,30]. Part of the criticism was based on lack of on-ground verification of the acquisition and on the fact that a substantial number of announced deals fail in the course of the negotiation stage. The new Land Matrix data set[28,31] improves upon these criticisms and specifies whether each deal is just intended or concluded and also reports the area under contract. It also indicates whether the land has already been put under production by the investors[28]. Here I consider *only* concluded deals for which the contract area was specified (Table S1), regardless of whether the land is under production because I assume that at this stage previous land users have already been excluded from accessing the acquired land. These criteria ensure that land rights have legally changed hands and that the ability of rural communities (who typically rely on traditional land tenure systems)[5,6,32,33] to access that land has been affected. The fact that these deals deny rural communities further access to agricultural land is all that is necessary for their incomes to be impacted. I readily acknowledge (as do the authors of the Land Matrix database) that conclusions from this database must be arrived at with caution and make every effort to ensure that our estimates are conservative. Also, since data are not available for the crops previously grown on targeted lands, our estimates of production represent the *potential* amount of crops able to be grown on these lands at current yields had the land continued to be available to local communities. I assume that the intended crop types were grown on the land prior to the land deal. This is reasonable since most prior land use is by smallholder agriculture[5,34]. This assumption in turn can lead to inconsistencies in certain instances (for sugar cane in particular) between FAO estimates of production and our own. As stated before, this is likely because targeted land may not yet be actively cultivated, but it is no longer accessible by rural communities. Thus, even if a community intended in the coming years to expand cultivation onto land that is now incorporated, this community would no longer have that option. Gross agricultural production values (USD $ for crops used as food, feed or seed) and gross agricultural production (tons of crop production used as food, feed or seed) were obtained from the FAOSTAT data base[35]. Unit prices of crops were calculated as the total gross agricultural production value of each crop by country divided by the gross agricultural production of that crop for that country[35]. A value of $484 per metric ton was used for missing oil palm data, as this was the unit value given by the FAO for all African countries considered where data was available. To account for production costs, I first took the sum of the gross capital stock for the year 2007 (the most recent year available) for land development, plantation crops, machinery and equipment[35]. I then divided this by the total gross production value of crops by country to obtain the national average fraction of gross agricultural value lost to production costs. This further ensures that our estimate is conservative since a portion of the gross capital stock considered also takes into account land development, machinery and equipment used for livestock production and thus is an overestimate. I do not consider the cost of fertilizers, as sub-Saharan Africa, Latin America and South-East Asia have low levels of synthetic fertilizer consumption[36,37]. I also do not consider transportation since the gross agriculture value represents the value of the production at farm gate. Oil palm production was converted to palm oil production by country-specific ratios of palm oil production to oil palm fruit production obtained from FAOSTAT[35]. The appropriate unit price was multiplied by the quantity of lost agricultural production, and the sum of these crop values gave the total lost agricultural income by a country as a result of recent land deals. This total was then divided by the average income per capita[38] to give the number of people who could *potentially* lose their income as a result of large scale land acquisitions (see supplementary materials for more details). Since data on average rural income were not available for the countries of interest, average income per capita was given as the gross national income (GNI) per capita in terms of purchasing power parity. These data were from the World Bank's World Development Indicators database[38], as were population data for each country and percent value added by the agricultural sector. The use of GNI (as opposed to rural per capita income) may, in turn, underestimate the total number of people affected, thus ensuring that our estimate errs on the conservative side. Jatropha was conservatively excluded from these calculations because: 1) it is not yet clear if the crop is profitable and 2) it is typically grown on marginal land[39]. Data for the percent value added by agriculture to GDP were from the World Bank's World Development Indicators database[38].

Our work here addresses the major limitations of the Land Matrix dataset by: 1) using an up-to-date database (that has addressed much of the criticism of its preceding versions)[31] and rigorous criteria to select land deals to include in our analysis, 2) using a simple, conservative yet powerful analysis to estimate the impact on rural income and 3) seeking only to approximate the *potential* number of rural people affected by large scale land acquisitions. Lastly, I should note that while the database used in this study is a significant improvement on its previous versions it is still subject to certain biases (e.g. countries' data policies, focus on international investments), which should be taken into consideration when drawing any conclusions.

Results

We estimate that in the 28 countries most affected by land deals from the year 2000 to present, more than 12.1 million people are potentially affected by the direct economic consequences of land acquisitions (Table 1). The percent of a population potentially affected by lost income due to this phenomenon falls below 1% for all but 7 countries (Gabon, Liberia, Malaysia, Mozambique, Papua New Guinea, Sierra Leone and South Sudan/Sudan). However, the impact on lost livelihood varies widely by country. In Papua New Guinea for example, an income that could support nearly one quarter (23 %) of the population is potentially lost. Conversely, in countries such as Russia (< .01 %), Brazil (.02%), Peru (.05%) and Uganda (.05 %), the relative impact on employment prospects is minimal. Of the countries in this study, 16 have a potential lost income equating to greater than 100,000 people, and 4 have greater than 1.5 million people potentially affected. In absolute numbers, Mozambique tops the list with more than 2.7 million people, followed by Indonesia (1.8 million), South Sudan/Sudan (1.7 million), Papua New Guinea (1.5 million) and Ethiopia (0.78 million). Since there are no data in the peer-reviewed literature supporting these findings[34], comparisons are limited. However, several reports from NGOs indicate that our estimates are reasonable. For instance, our estimate for Ethiopia agrees well with a report[40] placing the number of affected people at 1 million. Our approximation for Uganda corresponds well to an estimate for select affected districts of 20,000 people[41]. Also, a major land deal in Tanzania will reportedly displace more than 160,000 people[42]. According to our findings, the regions with the potential to be most heavily impacted in terms of lost agricultural income are sub-Saharan Africa and Southeast Asia. While Africa accounts for 43% of the appropriated area in this study, Africans comprise roughly two-thirds (8.2 million people) of all those potentially affected (Figure 2). I estimate total lost income globally at ~$34 billion, a number comparable to the ~$35 billion loaned by the World Bank for development and aid in 2012[43]. The local agricultural livelihoods of smaller countries in West Africa appear to be particularly vulnerable to the potential effects of land acquisition (Figure 2). Again, I stress that the results presented here are conservative estimates. The analysis here thus provides a new and simple way to quantify a phenomenon with a reputation for lack of transparency and to gain a first approximation of how severely impacts on rural income may be across countries.

Discussion

From the outset of this discussion it is important to note that while this study contributes important empirical evidence of the economic impacts of the global land rush on rural communities, the findings should by no means be viewed as conclusive. They should instead be considered as an upper limit (of potential impacts on rural people) against which future case studies can be measured. This is particularly noteworthy given the significant assumptions incorporated into our methodology (especially related to prior use and crop type) and, in turn, how they may influence our findings. Despite this, I find that where data is available our results agree quite well with case studies where rural communities were either displaced or their livelihoods were affected.

While the loss of income and employment opportunities by rural communities is an important impact to consider, I also acknowledge that with each land deal comes a unique set of benefits to, impacts on and responses by the affected local communities[44]. The fact of varying benefits, impacts and responses is true both between and within countries, as was highlighted in

McCarthy's work[45] in Indonesia. Here he noted that the options presented to smallholders and the ways in which they choose to interact with commercial agriculture ultimately dictate whether change is positive or negative. In addition, as Borras and Franco[46] have previously described, the perspective from which a land deal is viewed plays an important role in how benefits of land deals are defined and whether they have been realized. For instance, a land deal that improves crop production or rural employment opportunities may result in environmental degradation. While potential benefits and impacts vary with each case and for each stakeholder, Li's work[47] examining existing data on the land rush phenomenon and taken from a labor perspective demonstrates that poverty reduction is an unlikely result of large-scale land acquisitions. However, the question of benefits is far more certain at the national level for the target country where land deals are more likely to result in some economic and political benefits[47]. This was notably the case in a study by the organization Welthungerhilfe of a recent land deal in Sierra Leone[21,48]. In this instance, local farmers were denied access to land without prior consultation and experienced a drastic loss of reliable income, making them less able to afford food for their households and school fees for their children. Except for a small one-time payment to farmers of USD 220 and minimal annual area-based payments of USD 6.25 per hectare for oil palm land only (compared to an average annual GNI per capita of USD 880), farmers are unable to obtain income from the land. Conversely, the various levels of government administration receive the other 50% of the investors' yearly lease payment.

While this is a compelling example of what I seek to examine here, what is more broadly essential to consider is to what extent the potential benefits from land contracts (and the activities that follow) actually find their way to the populace just as the original agricultural income would. One way by which these changes in land tenure can potentially benefit and sustain the livelihoods of local communities is by providing employment opportunities with adequate income. While investing corporations regularly make estimates on new job creation, the actual number of jobs created is typically well below expectations, due to transitions to plantation-style agriculture preferring mechanization and wage laborers[3,4]. In most cases the opportunities for employment are low-quality, limited or nonexistent[3,4,47]. Moreover, land acquisitions largely affect rural (and generally poorer) communities in countries where wealth tends to be distributed less equally. Overall this means that vulnerable communities within vulnerable countries (i.e., those most impacted by changes in food prices) are also those more susceptible to livelihood loss due to the land rush. Where agricultural production is primarily contributed by subsistence farming, the loss of cropland can also be interpreted as a reduced ability to meet the dietary requirements of a targeted country's population[15,40].

The extent to which these land deals potentially affects employment prospects within a country varies widely and is unique to each case. The number of people potentially affected ranges from thousands to millions (Table 1), highlighting the fact that countries are differentially affected by and sensitive to consequences of large-scale land acquisitions. As per capita income can vary greatly between targeted countries (e.g. USD 330 per year in the Democratic Republic of the Congo vs. USD 14,680 per year in Malaysia)[38], a person's average income is an important consideration in assessing the consequences of such land deals. While I make every effort to keep our estimates conservative, our approximations of the potential number of people affected by income loss due to the global land rush provide important insight into which countries may expect to experience this impact most heavily (even if the country's land area under contract is comparatively small; e.g. Mozambique). How much of a country's income comes from agriculture (Figure 1) and how many of its people are employed in that sector (Table 1) contribute to how vulnerable a country may be to the effects of large scale land acquisitions. The strength of a targeted country's legal system, the extent of enforcement and the ease for investing countries in navigating its land tenure system also help determine which places are preferentially targeted[3,5,13]. Ultimately, this can lead to the sudden marginalization of rural communities and leave them with limited options for alternative forms of household income. To worsen this vulnerability, less developed countries (and rural areas in particular) are predicted to experience a disproportionately large amount of the adverse consequences of climate change[49,50]. While the analysis here focuses on people, the consequences of environmental change are likely to be compounded with a transition to a more commercialized means of agricultural production. These adverse effects typically associated with transition to commercial-scale agriculture include pollution from increased fertilizer usage and soil loss from mechanized planting and harvesting[39]. However, since the land rush has only taken place in the past several years, many of these potential adverse effects may require more time to be fully discernible. This is true not only for environmental impacts, but as Cotula and colleagues[26] point out, also for impacts on rural livelihoods, since land deals across the world are at various stages of establishment and implementation. The fact that many land deals have taken longer than expected to implement can also mean significant opportunity costs, where it becomes less likely that positive outcomes will counter negative impacts[26].

From the perspective of local communities, the economic consequences of land deals can often be thought of as analogous to those of crop failures. In both cases, the financial (e.g. transportation fare) and infrastructural (e.g. roads, bridges) means to seek employment through non-farm activities are often left intact but the enduring livelihoods of households are threatened. Given the proximity of many land deals to urban areas[5], the prospect of migration becomes all the more reasonable. In Bangladesh, a place visibly experiencing the early effects of climate change through increased flooding, it was found that crop failures (and not flooding) better explained people's propensity to migrate permanently[51]. Thus, as with land acquisitions, loss of local profit from crop production for the foreseeable future can make migration a reasonable option for securing a household's income[24,25]. Similarly in China, migration due to the conversion from subsistence farming to commercial agriculture has been reported[52]. Also, in Ethiopia, large scale land acquisitions have reportedly caused transboundary displacements of local farmers and pastoralists into Sudan[40].

Conclusion

Overall, how affected communities are able to financially cope with the impacts of these land deals depends upon their access to assets, infrastructure and opportunities[17].The effects of large-scale land investments can be multitudinous, with advocates on either side touting their positives (e.g. technology sharing, increased crop yields) and negatives (e.g. lost livelihoods, unjust land appropriations, environmental degradation)[10,53]. This study offers a first insight into the impact of the recent land rush on rural livelihoods. Our conservative estimate of over 12 million people losing their incomes is more than one third of the number of internally displaced people due to conflict (29 million people)[54] and one quarter of the number of migrations induced by natural hazards in 2012 (32 millions)[55]. This relatively large number of people may contribute to issues of food insecurity and poverty in rural areas while challenging the sustainability of urban growth as affected people seek to diversify household income[17].

Losing access to land can carry with it a variety of economic, social, nutritional and cultural consequences[9], a full discussion of which is beyond the scope of this study. Income loss represents just one way through which these deals might adversely affect rural communities. By quantifying the number of rural people potentially affected by these land deals, I can also begin to understand the extent of the social and cultural impacts, an equally important aspect of the on-going conversation surrounding the global land rush[12]. Our study provides estimates from an economic perspective against which field studies can be compared. Given the lack of transparency of this phenomenon, these findings provide a much needed *initial* empirical evaluation of the direct impacts of large scale land acquisitions on rural communities and their livelihoods. While our study and others like it[14] are a good first step, on-ground verification is an essential next step toward firmly quantifying the human impacts of this process[12,29,30]. Where our estimates best agree with such verifications can provide valuable information as to the primary impact (i.e. income loss) on rural households (and its magnitude) in these areas and help direct possible ways to address the problem.

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Tables and Figures

Table 1. Summary findings for grabbed countries.

	Total lost income ($)	**Total people affected**	**% of population**
Angola	79,337,812	15,383	0.08
Argentina	345,949,205	22,342	0.06
Benin	16,783,119	10,614	0.12
Brazil	454,969,840	41,386	0.02
Cameroon	203,675,121	90,845	0.46
Colombia	403,308,909	44,722	0.10
Congo	13,127,064	4,136	0.10
DRC	105,572,483	319,605	0.48
Ethiopia	809,980,299	785,701	0.95
Gabon	1,440,146,140	110,167	7.32
Ghana	332,672,327	206,456	0.85
Guatemala	68,573,647	14,817	0.10
Indonesia	7,736,024,665	1,847,609	0.77
Liberia	225,161,293	478,476	11.98
Madagascar	158,298,340	165,997	0.80
Malaysia	8,956,266,573	608,958	2.14
Morocco	926,336,692	201,836	0.63
Mozambique	2,443,013,473	2,710,813	11.59
Nigeria	331,781,421	153,439	0.10
Papua New Guinea	3,758,184,784	1,564,440	22.81
Peru	119,124,632	13,524	0.05
Philippines	804,018,409	203,256	0.22
Russia	27,585,683	1,423	<0.01
Sierra Leone	501,467,190	610,031	10.40
South Sudan & Sudan	3,561,260,372	1,731,108	3.97
Tanzania	305,055,452	215,955	0.48
Uganda	19,237,881	15,379	0.05
Uruguay	115,090,195	8,483	0.25
Total	34,262,003,020	12,196,904	..

Figures

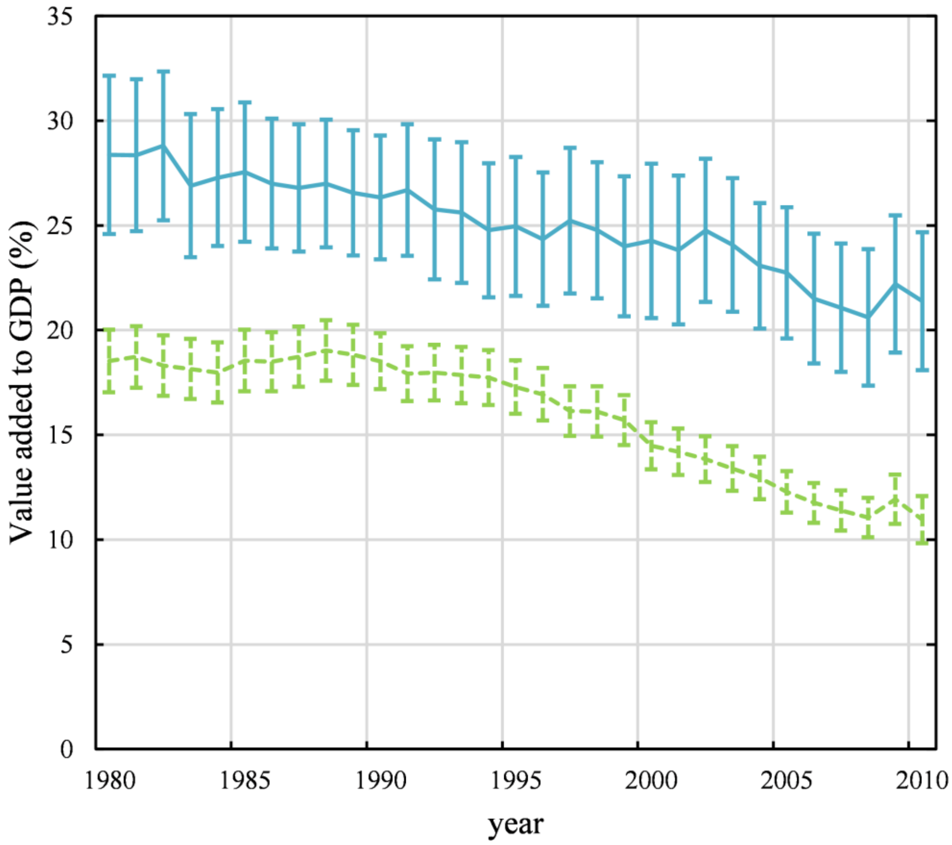

Figure 1. Average percent contribution by agricultural sector to gross domestic product for the 28 significantly grabbed countries (solid/blue) and all other countries (dashed/green) from 1980 through 2010. Error bars represent the standard error of the mean.

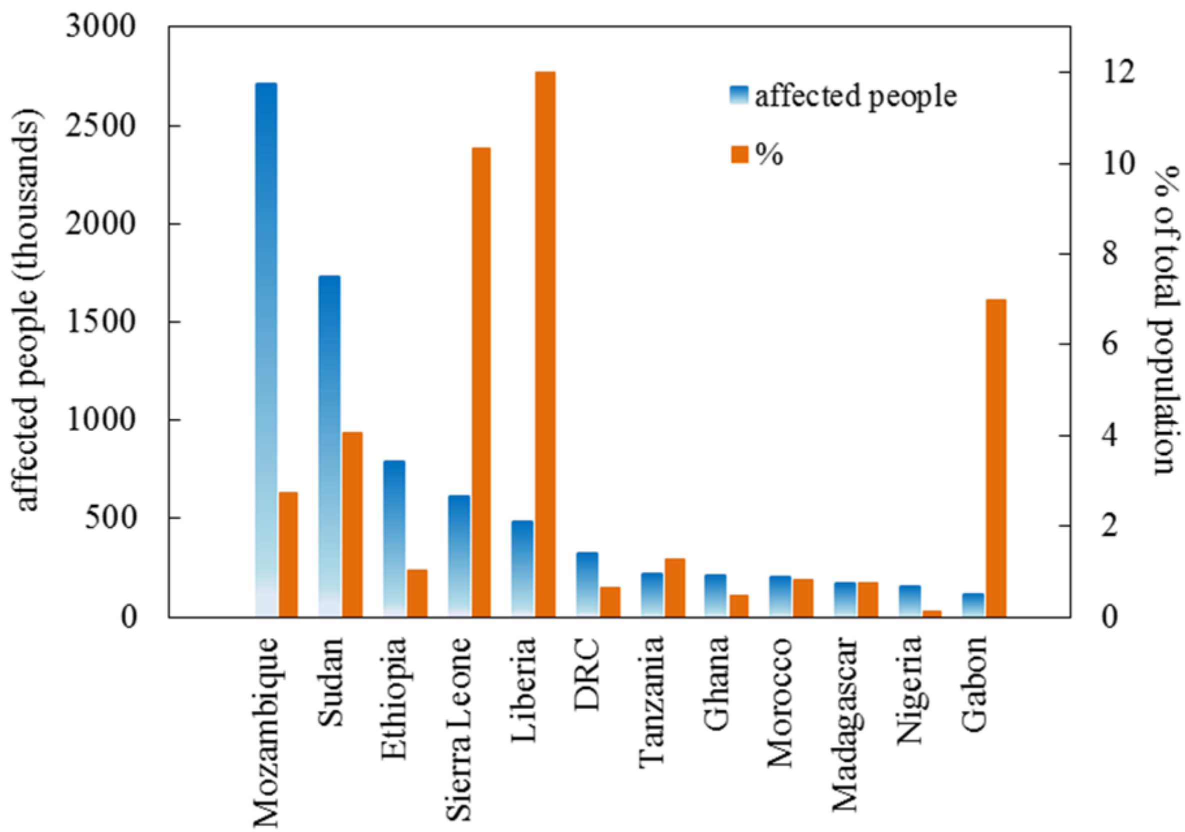

Figure 2. Summary for grabbed African countries. Chart shows African countries with more than 100,000 people potentially affected by land grabbing. Percent of total population is relative to the 2010 national populations.

ACCELERATED DEFORESTATION DRIVEN BY LARGE-SCALE LAND ACQUISITIONS IN CAMBODIA

Abstract

Investment in agricultural land in the developing world has rapidly increased in the past two decades[1, 2, 3]. In Cambodia, there has been a surge in Economic Land Concessions, in which long-term leases are provided to foreign and domestic investors for economic development. More than 2 million hectares[4] have been leased to date, sparking debate over the consequences for local communities and the environment[5]. This study combined official records of concession locations[4, 6] with a high-resolution dataset of changes in forest cover[7] to quantify the contribution of land concessions to deforestation between 2000 and 2012. This study then used covariate matching to control for variables other than classification as a concession that may influence forest loss. Nearly half of the area where concessions were granted between 2000 and 2012 was forested in 2000; this area then represented 12.4% of forest land cover in Cambodia. Within concessions, the annual rate of forest loss was between 29% and 105% higher than in comparable land areas outside concessions. Most of the deforestation within concessions occurred after the contract date, and whether an investor was domestic or foreign had no effect on deforestation rates. This study concluded that land acquisitions can act as powerful drivers of deforestation.

Introduction

Large-scale land acquisitions have been at the center of a debate between those who primarily see in them development opportunities and those concerned about the rights and livelihoods of local communities[8, 9]. Though promising an influx of technology and rural and economic development, land deals are often characterized by a lack of transparency and little or no involvement of previous land users[2, 10]. With only 13% of globally contracted area reportedly being put to productive use[11], many land acquisitions also appear to be speculative[3], and, in a number of cases, have reportedly led to evictions, violations of human rights and the loss of livelihoods[5, 12]. In addition to the frequent economic and social impacts of land deals on local communities, there are concerns that the exclusion of previous land users can also represent a loss of environmental stewardship[10]. Proponents of these land deals in turn argue that these lands are 'empty', 'marginal', 'virgin' or 'degraded' and can therefore be put to productive use without affecting the livelihoods of local communities[1, 2, 13]. While knowledge of previous land use remains largely incomplete, the leasing of 'empty' lands raises another set of concerns on land use change, deforestation and the associated environmental impacts[13, 14, 15].

As with potential impacts on previous land users, assertions about the environmental consequences of land acquisitions are often difficult to verify. Quantitative assessments of the previous land use (i.e. cropland, forests, rangeland) and of the changes in land cover are still missing[16, 17]. To that end, I focus on the case of Cambodia where lands acquired by foreign and domestic investors currently total 2.05 million hectares[4] (ha) – equivalent to 36% of the country's agricultural land[18] – and for which official government records of Economic Land Concessions (ELCs) and their associated geographic locations exist[4, 6]. By combining this information with remotely sensed data on forest cover[7], I determine the initial extent of forests in acquired lands for the year 2000 and analyze to what extent this forested area has changed annually through 2012. Because deforestation does not occur randomly across a landscape, I also employ a covariate matching approach to control for characteristics that may make an area more likely to undergo forest loss (e.g. distance from roads and cities). In doing so, I relate land acquisitions to deforestation and land use change and investigate whether such land deals enhance deforestation and habitat loss. Our analysis provides much needed quantitative evidence for the environmental effects of land deals and highlights how spatial data on large-scale land acquisitions can be profoundly useful in informing future concessions and land tenure policies[15].

Methods

The database on economic land concessions was produced by Open Development Cambodia[4]. The database used government data provided directly by the Cambodian Ministry of Agriculture, Forestry and Fisheries (MAFF)[6] for information on each deal including coordinates, area, contract date, investors and intended use. Data on village location and population also came from Open Development Cambodia[4] and were originally produced by Cambodia's National Institute of Statistics and Ministry of Planning as a product of the 2008 national census. Data on annual forest loss were from Hansen and colleagues[7]. This dataset provides the initial forest cover in the year 2000 (as a percentage of the pixel area) as well as the year in which a pixel (30m x 30m) gains or loses forest. For those initially forested pixels that undergo deforestation in a given year, I assume complete forest loss for that pixel in that year and all subsequent years. Forest gain from 2000 to 2012 was not considered in the calculation of deforestation rates because this was not reported on an annual basis. For all of Cambodia, the number of pixels experiencing apparent forest gain during this time was equivalent to 1% of initially forested pixels. Conversely, this value was 14% for ELCs, due in large part to the establishment of tree plantations, as our validation showed.

Validation of forest cover and tree plantations was carried out in two ways. The first approach was done using a new cropland cover map (1 km resolution)[29] – which was the product of fusing numerous published datasets on cropland extent and included oil palm areas as cropland – to evaluate the consistency between areas reported as forest by Hansen et al.[7] and non-crop areas. I resampled the 30 m forest cover data[7] to 1 km resolution and classified a pixel as forest when its tree cover exceeded 90%. In 99% of the cases (and in the entire area of ELCs), forested areas coincided with areas with no cropland. As further validation of the forest cover dataset, 29 land deals (15% of all ELCs) were randomly selected. Based on the Hansen dataset, the average forest area (> 30% tree cover) and tree cover of each of these deals was then calculated for the beginning of the year 2013 after accounting for tree loss. Then year 2013 high resolution satellite images from Google Earth Pro ® (Imagery © 2015 TerraMetrics) were imported to ArcGIS using the Arc2earth software[30] for visually delineating areas of tree plantations, which stand as areas subdivided into regular rectangular (or, in general, polygonal) parcels, or areas with trees growing in straight rows. These tree plantations were then digitized (for examples, see Supplementary Figures 2C-D) and used to calculate the percent overlap with forest area after accounting for forest loss between the years 2001 and 2012. For the 29 randomly sampled ELCs, on average only 2.5% of forested areas occurred within tree plantations (Supplementary Table 20). However, in certain individual deals, this percentage was more substantial (in one case >25% of forested area). Some of these 'false positive' areas are likely as a result of clearing for tree plantations or other intended crops during the year 2013 and may also have occurred in places where tree plantations were established before the year 2000 – the start of the Hansen dataset. From this analysis, I have demonstrated that our approach is overall sufficient for a national-scale analysis of deforestation in Cambodia and also shown that our estimates of forest loss are conservative. For calculating average percent tree cover, the digitized tree plantations areas were subtracted from the ELC area before again calculating the tree cover. Linear regression analyses were used to compare average percent tree cover within each randomly selected ELC both before and after accounting for the area of tree plantation ($R^2 = 0.99$). In this way, I were able to confirm that the effects of tree plantations on calculations of natural tree cover was minimal (Supplementary Figure 3).

A number of factors may also influence the likelihood that an area will be deforested, regardless of whether or not it is located in an ELC. To control for these characteristic covariates, I employed a covariate matching approach similar to that used by Andam and colleagues[31] for which they measured the effectiveness of protected forest areas. The goal of this approach is to establish ‘balance’, so that the covariate distributions of ELC and non-ELC pixels are ‘very similar’. Thus it is then possible to compare ELC and non-ELC plots to examine the potential effect of land acquisition on deforestation. To this end, I randomly selected 179,347 initially forested pixels (30m x 30m) – 28,439 of which were located within ELCs. Pixels in protected areas were not considered. For each pixel, I determined covariate information for distance from the nearest road, distance from the nearest waterway, distance from the nearest railway, distance from the nearest urban area (i.e. population density greater than 300 people km^{-2}), distance from forest edge, slope class, soil suitability and district area (Supplementary Tables 3-14). Distance from the nearest urban area was calculated using a year 2005 population density dataset from CIESEN/CIAT[32]. Classes for median terrain slope and agro-ecological suitability for rain-fed high-input oil palm (Supplementary Table 19) were assigned using data from the FAO/IIASA’s

Global Agro-Ecological Zones[33]. Matching was performed in R using the ‘Matching’ package[34]. I also examined the sensitivity of these results to hidden bias using Rosenbaum’s sensitivity test[35]. Matched ELC and non-ELC plots differ in their likelihood of being deforested by an unknown covariate by a factor of Γ, so that $\Gamma = 1$ means that ELC plots are equally as likely as their matched non-ELC plots to be deforested as a result of hidden bias. The higher that gamma can be increased while the result still remains significantly different from zero, the more robust the results are to hidden bias. Results were overall insensitive to hidden bias, though it is important to note that this was not the case in the absence of selection criteria for ELC contract date. In cases where the results are not robust to hidden bias, I note that while conclusions drawn from those results should be viewed with caution, this sensitivity does not guarantee the actual presence of an unobserved confounder. To determine the potential for leakage (e.g. displacement of forest loss into neighboring forests), I also considered the effect of a 2 km buffer (the same distance used by Andam and colleagues[31]) around protected areas and ELCs. In adopting this distance for our analysis, I should note that leakage can occur at various distances and, given the indirect pathways by which it is often driven, can also be difficult to fully quantify. Complete results of matching and sensitivity analyses are presented in Supplementary Tables 2 - 19. For examining the amount of deforestation that occurred before and after the contract date of a land acquisition, only those deals with contract dates between January 2001 and December 2011 were used. Also, to prevent overestimation of the percentage of deforestation that occurred after the contract date, I assume that any deforestation occurring on the same year of the contract took place before the contract.

Results

Considerable deforestation has occurred across Cambodia since the start of the century, a disproportionate amount of which has taken place within ELCs (Fig. 1a). While 12.4% of Cambodia's forests were contained in ELCs in 2000, 19.8% (or 0.26 Mha) of the country's forest loss through 2012 has been within these land concessions (Supplementary Table 1). In addition, the contribution of these acquired lands to Cambodia's annual forest loss rose from 12.1% in 2001 to 27.0% in 2012. However, while these differences appear stark (Fig. 1b), they do not directly address whether forested ELC areas are in fact more likely than non-ELC areas to experience forest loss because deforestation is not a random process. Using a covariate matching approach, I controlled for characteristics that influence deforestation (see Supplementary Materials). Our analysis showed that while ELCs and non-ELC areas both experienced increases in the relative rate of deforestation from the initial ~0.5% yr^{-1}, forest removal was particularly aggressive within land concessions. As a result, the rate of forest loss on acquired lands increased to 4.3-5.2% yr^{-1} by the end of the study period (2010-2012 mean), 29-105% greater than that for matched non-ELC areas (Supplementary Table 2). Regardless of selection criteria – reporting of ELC contract date, distance from protected area, distance from ELC boundary (for non-ELC plots) – ELC areas consistently exhibited higher deforestation rates (Fig. 1c). These results were overall insensitive to hidden bias (see Supplementary Tables 15-18). Areas more distant (> 2 km) from ELCs with earlier contract dates (2001 – 2006) were slightly less likely to undergo deforestation (Fig. 1a, 1d); this suggests 'spillage' in the areas immediately surrounding these ELCs - possibly as a result of investing companies exceeding their contract areas, from illegal logging and/or from the displacement of local communities to surrounding areas. The opposite was observed for the non-ELC areas matched with more recent (2007 – 2012) concessions, where more distant areas were more susceptible to forest loss and more proximal areas perhaps experienced an unintended protective effect.

Discussion

Abrupt land use change in ELCs is apparent when comparing the pattern of forest loss in acquired lands with that in other areas (Fig. 2). As opposed to the less targeted encroachment on forests generally observed throughout the country, large areas of forest within a number of ELCs were removed in a single year to make way for tree plantations and other crops. This clustered patterning of forest loss in ELCs likely explains why our random sampling underestimates the deforestation rate on ELCs (Fig. 1b-c). On average, 63% of cumulative forest loss on acquired lands has occurred after the date of the land deal contract (Supplementary Figure 1). I found this post-contract increase in forest loss to be consistent regardless of investor origin (i.e. foreign or domestic) and intended use. One requirement of any company that is granted an ELC contract is that it provide the State Land Management Committee with a detailed land use plan for the entirety of the contract, a condition intended to prevent irresponsible land use and speculative investments. However, many investors granted ELCs have not adhered to these land use plans, and only recently has the Cambodian Ministry of Agriculture, Forest and Fisheries begun reviewing and cancelling contracts that are inactive or improperly used[19]. Combined with this general lack of monitoring and enforcement, our findings show that little lag typically exists between when an ELC contract is signed and when investors begin to modify the land for productive use. As a result, a large portion of forest (0.67 Mha remaining within ELCs) are now at a heightened risk of removal (Supplementary Table 1).

The recent surge in land concessions and the deforestation that has followed provide strong indication that shorter-term economic goals are trumping long-term sustainability and that serious environmental consequences are already occurring. With 28% of forests within ELCs removed since the start of the century, the rapid deforestation and conversion to commercial agriculture can produce various environmental impacts including enhanced carbon emissions, biodiversity loss, soil erosion and nutrient runoff[20, 21, 22]. In addition to the immediate effects of these land use changes, the vast majority of ELCs considered in this study have a contract length of 70 years and thus will continue to exert significant influence on land use and land use change in Cambodia for most of this century. Furthermore, the potential for many of these environmental impacts to occur is made all the more likely given that many ELCs are intended for the production and export of agricultural goods (86 of 191 deals for rubber alone). Foreign consumers of these export-oriented crops may unconsciously place a lower value on minimizing their impacts as they do not directly observe the environmental consequences of their choices[10, 23, 24].

Equivalent to a third of Cambodia's agricultural land, ELCs may also have important implications for domestic food security and the livelihoods of rural people[10, 25, 26] – especially when the crops from these lands are mainly agroindustrial and intended for export[13]. With nearly half of the acquired areas initially forested in 2000 (Supplementary Table 1), what is apparent from the work here is that the areas targeted by ELCs were not entirely under crop cultivation before they were acquired and are continually undergoing rapid land cover changes. Beyond this knowledge of forest location, information on the distribution of previous land use remains incomplete, though anecdotal evidence suggests that many areas were communally held (as farms, forest or conservation land) and that the livelihoods of many villagers are dependent on forests[5,13]. Recent village census data[4] (from the Cambodian Ministry of Planning) show that 277 villages – home to 213,000 people – fall within ELC boundaries. Further, despite a number of legal protections for indigenous people in Cambodia, by 2012 nearly 100 ELCs had been granted at least partially on indigenous lands[27]. As a result, dispossession, evictions and conflict are some of the commonly reported impacts of ELCs on local communities[13, 19, 28], effects that often violate the right of indigenous people to free, prior, and informed consent as recognized by the Cambodian government and the international community[27]. While benefits from ELCs (e.g., job creation, improved infrastructure) are also certainly possible, quantitative studies examining the economic and social benefits and impacts of ELCs are still lacking. Systematic mapping, classification and registration of state public and private land in Cambodia have only partially taken place, while land use plans have not been adopted by provincial or municipal land management committees[19]. These lines of evidence are representative of the recent situation in Cambodia, where a legal framework for protecting local communities is well-established but proper implementation and monitoring has been largely absent as a result of weak local and national governance bodies. That these institutions have been unable to ensure investors' adherence to ELC land use plans has ultimately meant that many stakeholders are excluded from the potential benefits of ELCs. In spite of this, a recent moratorium on ELCs as well as a new land titling initiative could provide improved protection for the rural poor, distributing more than 200,000 land titles to households within the first year of the program[19]. However, the enduring effectiveness of these government actions remains to be seen.

The phenomenon of land acquisitions is especially fast-moving in Cambodia, where in just a few years a large area can go from a mixture of forests and smallholder farms to industrial plantation-style monocultures. Such rapid transitions in land use are also possible in other targeted countries where acquired land – much of which is not yet under production[11] – can be quickly put to productive use. In these places there is urgent need for swift evidence-based action that better involves all stakeholders and integrates sustainability, so that the potential benefits of acquisitions might be enhanced and their human and environmental impacts minimized. These decisions are only possible, however, if government agencies responsible for land tenure records make a concerted effort to improve access to the geographic coordinates of land deals. More open sharing of such information represents an important step towards improving the transparency of land acquisitions and – as evidenced by this study – will allow governments and the international community to better assess the environmental impacts of the global land rush to date and to advance the related policy debate.

Acknowledgements

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Figures

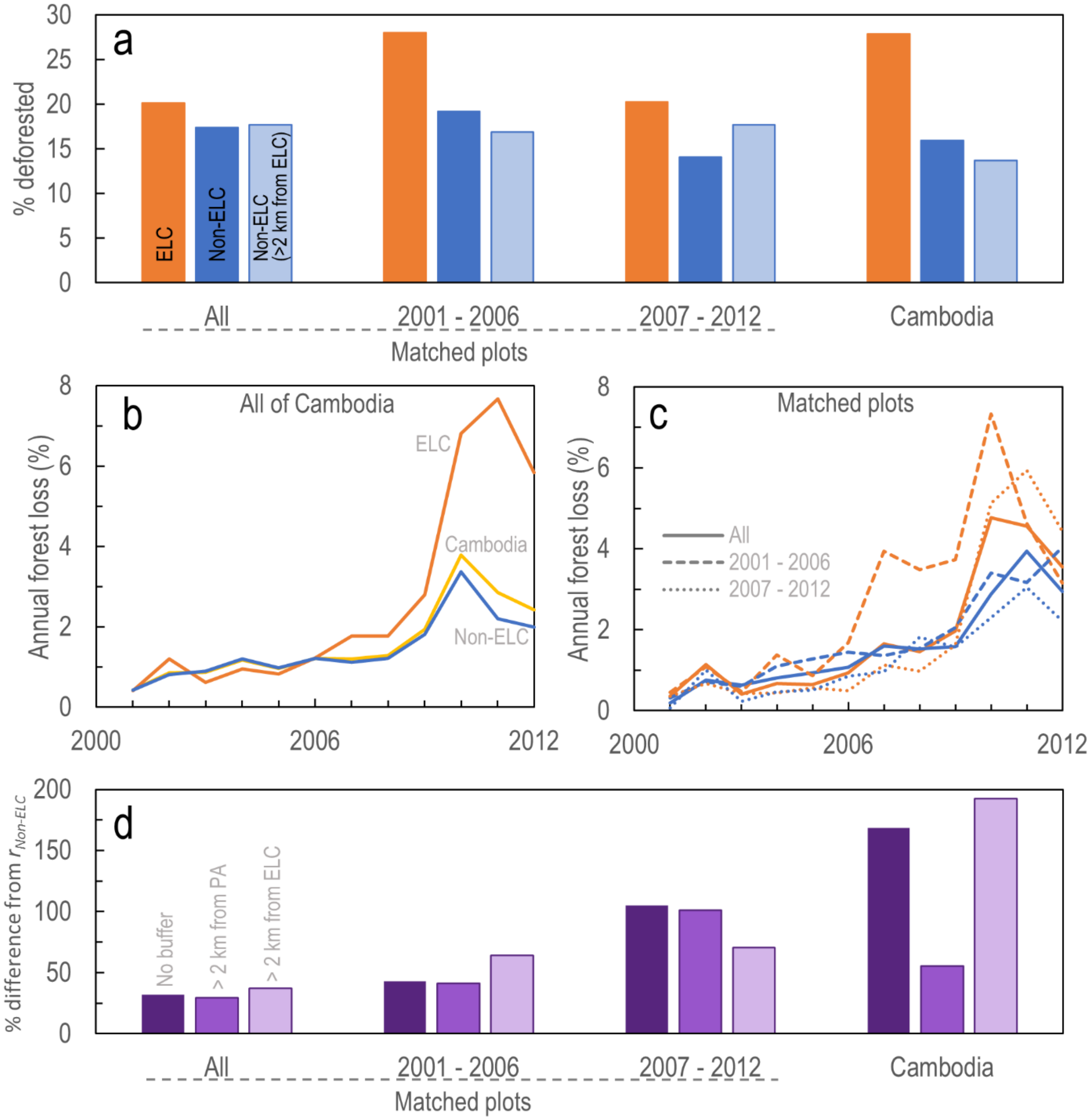

Figure 1. Deforestation in Cambodia. a) Deforestation for matched plots and all of Cambodia. 'All' – all matched plots; '2001–2006' and '2007-2012' – only matched plots with an ELC contract date within specified years; 'Non-ELC (>2km from ELC)' – excludes non-ELC plots within 2 km of ELC. b,c) Annual deforestation rates for Cambodia and for matched plots. d) Percent difference between 2010-2012 average deforestation rates of ELC and non-ELC areas, calculated as 100*(r_{ELC} - r_{NonELC})/r_{NonELC} (Supplementary Table 2). '>2km from PA' – excludes plots within 2 km of protected area. '>2km from ELC' – excludes non-ELC plots within 2 km of ELC.

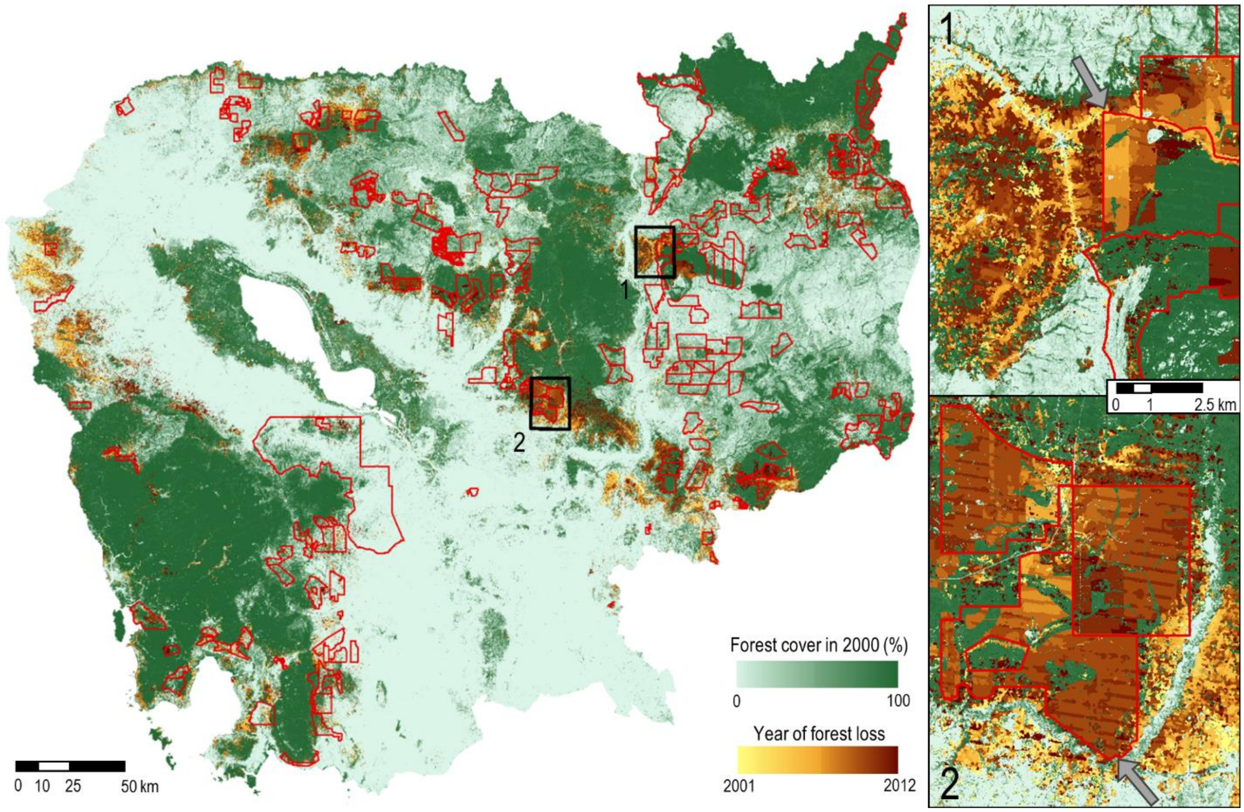

Figure 2. Map of forest cover, forest loss and confirmed ELCs in Cambodia. For the ELCs indicated by arrows, the establishment of tree plantations is also shown in Supplementary Figures 2 and 3.

MEETING FUTURE CROP DEMAND WITH CURRENT AGRICULTURAL RESOURCES: REQUIRED CHANGES IN DIETARY TRENDS AND PRODUCTION EFFICIENCIES

Abstract:

Meeting the food needs of the growing and increasingly affluent human population with the planet's limited resources is a major challenge of our time. Seen as the preferred approach to global food security issues, 'sustainable intensification' is the enhancement of crop yields while minimizing environmental impacts and preserving the ability of future generations to use the land. It is still unclear to what extent sustainable intensification would allow humanity to meet its demand for food commodities. This study used the footprints for water, nitrogen, carbon and land to quantitatively evaluate resource demands and greenhouse gas (GHG) emissions of future agriculture and investigate whether an increase in these environmental burdens of food production can be avoided under a variety of dietary scenarios. This study calculated average footprints of the current diet and found that animal products account for 43%-87% of an individual's environmental footprint – compared to 18% of caloric intake and 39% of protein intake. Interestingly, this study found that projected improvements in production efficiency would be insufficient to meet future food demand without also increasing the total environmental burden of food production. Transitioning to less impactful diets would in many cases allow production efficiency to keep pace with growth in human demand while minimizing the food system's environmental burden. This study provides a useful approach for evaluating the attainability of sustainable targets and for better integrating food security and environmental impacts.

Introduction

Global food production is one of the most significant ways by which humans have modified natural systems[1]. These impacts are well studied, ranging from the depletion of rivers and groundwater for irrigation[2,3] to nutrient pollution from the large-scale anthropogenic fixation and application of reactive nitrogen for fertilizers[4,5] to greenhouse gas emissions from mechanized cultivation, land use change, ruminant production and food trade[6]. With humanity already exceeding its sustainable use of Earth's systems in a number of ways[7,8,9,10,11], there is growing concern that the combination of population growth and increasing per-capita global affluence[12] portend yet more profound and pervasive consequences[13,14]. Thus, there is widespread agreement that food production must increase substantially while at the same time minimizing environmental impacts, an approach known as 'sustainable intensification'. Potential solutions to address this apparent dilemma include closing crop yield gaps, reducing food waste, moderating diets and reducing inefficiencies in resource use[15].

A number of recent studies have asked by how much food supply can increase if a single one of the above solutions was implemented. For instance, Mueller et al.[16] found that by maximizing crop yields (i.e. closing yield gaps), global crop production could increase by 45-70%. Kummu et al.[17] determined that an additional 1 billion people could be fed if food waste was halved from 24% to 12%. Also by changing from current diets to a globally adequate diet (3000 kcal cap^{-1} day^{-1}; 20% animal kcal), Davis et al.[18] found that an additional 0.8 billion people could be fed. Finally in another recent study, Mueller et al.[19] determined that nitrogen application, when more efficiently distributed across the planet, could be reduced by 50% while still achieving current levels of cereal production. While these and other studies[20,21] have certainly helped determine to what extent certain improvements are possible, they do not provide an integrated view of future human demand, food production and its multiple environmental impacts. In addition, many lack a temporal component. Thus it is unclear whether such advances can keep pace with projected increases in human demand.

This question of timing can be addressed in two ways. The first approach is based on past trends, where one estimates how much improvement is possible within a given period of time and whether this will achieve a pre-determined target. This is exemplified in a study by Ray and colleagues[22], where the authors asked whether historical rates of crop yield improvement would be sufficient to meet the doubling in human demand by the year 2050. While such an approach helps in understanding what may be expected if past trends continue, it is necessarily data-intensive. In addition, relying on past trends may not accurately capture future factors adequately (e.g., climate change, improved technologies). The second approach instead starts with a pre-determined target (e.g., a desired level of GHG emissions by 2050) and then asks to what extent improvements must be made in order to meet that target. This approach is useful when a continuation of past trends is undesirable and is especially valuable in situations where historical data may be lacking, both of which apply to the product- and country-specific environmental footprints of food production.

Here I combine both approaches to examine the extent to which production efficiencies (i.e., footprint intensities) and dietary patterns will need to change by mid-century in order to maintain current levels of resource use and emissions (i.e., environmental burdens), which many argue are already unsustainable[7,8,9,10,11]. I begin by calculating what the total food-related environmental burdens for water, GHGs, nitrogen and land would be in the year 2050 under constant (circa 2009) footprint intensities and for several future diet scenarios[23]. By examining these changes relative to the year 2009, I determine the improvement in footprint intensity required to prevent an overall increase in the environmental burden of a resource and compare the required change to projections of historical improvements in production efficiencies. In instances where the required change exceeds the relative potential enhancement in footprint intensity, the overall environmental burden of that resource must necessarily increase to support human demand. In considering these multiple environmental metrics and diet scenarios simultaneously, I also provide a much needed assessment of the tradeoffs that may occur and how dietary choices affect each environmental burden differently. In doing all of this, I present a quantitative, multi-metric assessment of how changes in efficiency and dietary patterns can combine to increase food supply and minimize environmental impacts from agriculture.

Methods

Data

Data on historic diets, harvested area, and agricultural production came from the FAO's FAOSTAT database[24]. Affluence-based dietary projections (i.e. based on projected growth in per capita GDP or a 'GDP-based scenario'), alternative diet scenarios and protein conversion ratios and feed compositions for livestock and animal products were from Tilman and Clark[23]. Alternative diet scenarios were Mediterranean, pescetarian and vegetarian (see Table 1; Supplementary Table 1a). In using the alternative diet values derived by Tilman and Clark[23] from various dietary recommendation studies, I also note that the definition of each alternative diet can vary substantially between studies and regions. This is particularly true for the composition of the Mediterranean diet utilized by Tilman and Clark and those recommended in other literature sources (refs. 25, 26, 27). While I utilize the former for consistency, our approach provides a straight-forward means by which to incorporate other alternative diets, additional nutrient requirements, or variations of the scenarios presented here (e.g., ref. 20). Country-level water footprint data for plant and non-seafood animal products (centered on the year 2000) were taken from two studies by Mekonnen and Hoekstra[28,29]. Our study only considered consumptive uses of irrigation water and rainwater (i.e. blue and green water footprints, respectively). Product-specific global carbon emission values for the year 2009 came from Tilman and Clark[23]. Crop-specific synthetic nitrogen application for the year 2010 (for 26 countries, the EU-27 and the rest of the world; Supplementary Table 2) was taken from a recent study by the International Fertilizer Industry Association (IFA)[30]. Historic population data and projections were from the UN Population Division[31].

Obtaining current global footprint intensities

The true footprint of a good can be defined as all of the inputs – both direct and indirect – needed to produce and deliver a certain good along its full supply chain (see ref. 32). To avoid confusion in terminology, I adopt the more general term of ‘footprint intensity’ to describe the product-specific ratio of inputs to product output. In describing the methods used in this study, it is important to highlight the differences between the approach I utilize here to develop certain footprint intensities (i.e., land and nitrogen) and what others have done in previous studies. While the footprint intensities for water and GHGs came from studies which employed life-cycle assessments and comprehensive input-output models (and are therefore true footprint values), a lack of comprehensive country- and crop-specific values for land and nitrogen required us to develop methodologies that captured their major direct requirements in food production. Thus I use the term ‘footprint’ when referring to water or carbon individually, and ‘footprint intensity’ when referring to land, nitrogen or any combination of the four environmental metrics.

For land, I calculated the footprint intensity as simply the harvested area of a crop divided by the production of that crop (i.e., the inverse of the yield). Though cropland represents the most extensive requirement of land in the production of a food item, Weinzettel et al.[33] have shown that calculating a true land footprint must also account for the other land requirements of an item's production (e.g., the space occupied by a barn or processing plant) – requirements which our approach does not include. Similarly for nitrogen, I calculated the footprint intensity simply as the ratio of synthetic nitrogen applied to an area and the crop production of that area, and assumed that all anthropogenic nitrogen inputs will eventually reach the environment[34]. While this approach does not capture potential recycling or losses at each step along the supply chain, it agrees broadly with the overall inputs and outputs of the nitrogen footprint model described by Leach et al.[35]. It is also worth noting that because our study only considers consumption patterns from a global perspective – country-specific values are only calculated for the footprint intensities of production – I avoid many of the difficulties associated with obtaining accurate footprint intensity values (e.g., accounting for virtual trade of resources).

Land

Country-specific land footprint intensity for primary plant commodities (i.e., ha per kg of crop) was calculated as the harvested area in 2010 divided by the amount of crop production[24]. These values agree well with the cropland footprints of production reported by Weinzettel et al.[33] (Supplementary Figure 1). The land footprint intensity of vegetable oils, η_{vo}, depends on the land footprint intensity of oil crops adjusted to account for the fraction of oil crops used for oil production as well as the production of oilcakes (for feed) from the byproducts of oil crop processing. Therefore, the land footprint intensity value for vegetable oils was as calculated as:

$$\eta_{vo} = \eta_{oc}\left(\frac{p_{vo}}{(ap_{oc}) - p_{cake}}\right) \qquad (1)$$

where η_{oc} is the land footprint intensity for oilcrops, p_{vo} is the production of vegetable oil in metric tons, a is the fraction of oilcrop production used for processed goods, p_{oc} is the production of oilcrops and p_{cake} is the production of oilcakes. Global land footprint intensity values for each plant commodity group (e.g., cereals) were calculated as the production-weighted average of country-specific land footprint intensity values. Variability of the global land footprint intensity value for a plant commodity group was calculated as the production-weighted standard deviation (σ_w):

$$\sigma_w = \sqrt{\frac{\sum_{i=1}^{N} p_i(\eta_i - \overline{\eta_w})^2}{\sum_{i=1}^{N} p_i}} \tag{2}$$

where p_i is the production of a plant commodity group in country i, η_i is the land footprint intensity of a plant commodity group in country i, and $\overline{\eta_w}$ is the global production-weighted average of land footprint intensity for a plant commodity group. The full list of products considered for land and all other environmental metrics is presented in Supplementary Table 3. I also note the differences in product coverage between environmental metrics – due largely to data limitations and varied naming schemes – which should be kept in mind when considering the findings of this study.

Based on feed conversion ratios (FCRs) and feed rations reported by FAO[36,37] (Supplementary Table 4a-d), the feed component of the global land footprint intensity of animal product k, η_k, was then calculated as follows:

$$\eta_k = f_k \sum \left(\frac{r_{pc,k}\eta_{pc}}{100}\right) \tag{3}$$

where f_k is the FCR (i.e. plant kcal: animal kcal) for animal product k, $r_{pc,k}$ is the feed ration (%) of a given plant commodity for animal product k and η_{pc} is the land footprint intensity of that plant commodity. These FCR values agree well with those presented in Davis and D'Odorico[38]. Further information of how FCRs and dietary rations (originally reported at the sub-regional scale) were converted to global scale can be found in the Supplementary Table 4a-d. Pasture land was split between beef and milk production (92% and 8%, respectively) following the methodology of Eshel and colleagues[39]. The variability of land footprint intensity for each animal commodity was calculated using error propagation through Equation 3.

The land footprint intensity for seafood was calculated separately from other animal products. I only calculate the global seafood land footprint intensity based on the direct land requirements to produce the ingredients of aquaculture feeds. Aquaculture feed compositions vary, but generally include a combination of fishmeal, fish oil, and meals, cakes, protein concentrates, and oils of crops (e.g. soybeans, canola, sunflower, etc.). Feed composition and use varies by species and production method. I use data from Tacon et al.[40] to calculate a weighted average of the land footprint intensity for seafood. First, I determined the land footprint intensity of terrestrial feeds for aquaculture. The land footprint intensity for feed meal derived from crop i ($\eta_{fm,i}$) was calculated as:

$$\eta_{fm,i} = \eta_{rc,i}\left(\frac{p_{roc,i}}{p_{oc,i}}\right) \tag{4}$$

where $\eta_{rc,i}$ is the land footprint intensity of raw crop i, $p_{oc,i}$ is the oilcake production of crop i in 2010[24] and $p_{roc,i}$ is the oilcake production of crop i in the year 2010 in raw equivalents (i.e., the total amount of raw crop i required to produce $p_{oc,i}$). This calculation was used for cottonseed meal, mustard seed cake, peanut meal, rapeseed meal, soybean meal, and sunflower seed meal. The value for rapeseed meal was used for canola protein concentrate. The average value of soybean meal and peanut meal was used for lupin kernel meal, faba bean meal and field pea meal. Because gluten products are the protein concentrate of a crop, I assume that only the weight of the protein remain after processing for gluten. Thus, the land footprint intensity of a (wheat or corn meal) gluten product ($\eta_{g,i}$) was calculated as:

$$\eta_{g,i} = \eta_{rc,i}\left(\frac{pc_{food,i}}{pc_{prot,i}}\right) \tag{5}$$

where $pc_{food,i}$ is the daily per capita food supply of wheat or maize and $pc_{prot,i}$ is the daily per capita protein supply of wheat or maize. The land footprint intensities for soybean oil and rapeseed oil in aquaculture feed, $\eta_{vo,i}$, were calculated in the same way as described above for vegetable oils. Values are reported in Supplementary Table 5a.

Next, I calculated the total land use (ha) of these terrestrial feeds for seafood group j (L_j) as:

$$L_j = \sum\left(\frac{p_{i,j}r_{i,j}\eta_{i,j}}{100}\right) \tag{6}$$

where $p_{i,j}$ is the production of terrestrial feed i used for seafood group j[40], $r_{i,j}$ is the feed ration (%) of terrestrial feed i for seafood group j[40], and $\eta_{i,j}$ is the land footprint of terrestrial feed i. Following this, the overall land footprint intensity of seafood from aquaculture ($\eta_{aq,i}$) was found by:

$$\eta_{aq,i} = \frac{\sum(c_j L_j)}{\sum(p_{s,j})} \tag{7}$$

where c_j is the conversion factor from live weight to product weight for seafood group j[41] and $p_{s,j}$ is the live weight production of seafood group j. The total land use for aquaculture was estimated at 36.6 Mha (the product of total aquaculture production and the land footprint intensity of aquaculture). Because a reliable value does not exist, the area physically occupied by aquaculture ponds – reported up to 8.2 Mha[42] – was not included in our estimate. Also, due to a lack of country-specific data for seafood, I could not assess its inter-country variability for land footprint intensity and for all other environmental metrics considered in this study.

Water

The methodology described above for the land footprint intensity of seafood was also used to calculate the water footprint of aquaculture seafood. I assume the water, nitrogen and land footprints of capture seafood to be zero (e.g., ref. 43) and that this type of seafood production comprises 60.1% of total current seafood production[44]. Values used in the calculations for the land and water footprints of seafood are presented in Supplementary Tables 5a-c.

The global water footprint for each non-seafood product was calculated as the production-weighted average of the country-level water footprints (Supplementary Table 3). The global water footprint for each commodity group was then determined as the production-weighted average of these global water footprints of individual products. The variability of each commodity group was calculated using Equation 2.

Nitrogen

Country- and crop-specific nitrogen footprint intensities for plant products (i.e. kg of applied N per kg of crop) were calculated as the amount of synthetic nitrogen fertilizer applied in 2010[30] divided by the amount of crop production (kg crop). Production-weighted averages were again used to combine the nitrogen footprint intensities of individual crops into the larger commodity groupings. Because pulses were included with ‘other crops’ in the IFA data, the nitrogen footprint intensity calculated for soybeans (also a nitrogen-fixing crop) was used for pulses. The nitrogen footprint intensity for vegetable oils, the nitrogen footprint intensity for the feed component of non-seafood animal products, and the standard deviation for each commodity group were all calculated in the same way as for the land footprint intensity. Data on nitrogen use for pastures came from Lassaletta and colleagues[45] and was split between beef and milk production (92% and 8%, respectively)[39]. The nitrogen footprint intensity for seafood was an updated value taken from Leach et al.[35]. While most N used in agriculture in developed countries is from synthetic fertilizer, other sources (e.g., biological N fixation, manure, compost) play a more important role in developing countries; as such, the N footprint intensities calculated here are likely conservative estimates.

Carbon

Global carbon footprints (i.e. kg CO_2eq emissions per kg of crop) for different food commodity groups were used as reported by Tilman and Clark[23]. For cereals and fruits, a production-weighted average was used to combine the values for specific products into commodity groups. The carbon footprint of seafood was taken from Tilman and Clark[23], with an average of non-trawling capture and non-recirculating aquaculture, weighted by production. All current (circa 2009) footprints are reported in Supplementary Table 1b. Because the Tilman and Clark values only reported the standard error between carbon footprint studies that they considered, I do not include an estimate of the variability between countries for carbon footprint.

In summary, the global footprint intensities for land, water, and nitrogen were calculated as production-weighted averages of individual countries while the global footprint for GHGs was only available at the global scale. These global footprint intensities were then used directly for future projections and scenarios.

Projections of diet, demand and efficiencies

Changes in annual per capita demand for each commodity group were calculated as linear trends from 2009 values (from ref. 24) to the 2050 projected values from Tilman and colleagues[23]. The percent changes in per capita demand for 'empty calories', 'fruits/vegetables' and 'pulses/nuts' – as reported by Tilman and Clark[23] – were used in this study for sugar crops, vegetable oils and oil crops, respectively. For a given year (*x*) and environmental metric (*EM*), the total global environmental burden of food production ($g_{EM,x}$) assuming a constant footprint intensity was calculated as:

$$g_{EM,x} = p_x \sum \left(d_{g,x}\eta_{g,2009}\right) \tag{8}$$

where p_x is the projected population in year *x*, $d_{g,x}$ is the projected per capita demand for commodity group *g* in year *x*, and $\eta_{g,2009}$ is the current global footprint intensity of commodity group *g* corresponding to the environmental metric of interest. I assume that any future growth in seafood demand – for GDP-based, Mediterranean and pescetarian diets – will be met by aquaculture, as production from global capture fisheries has already leveled off[46]. For global demand for seafood under a vegetarian diet (which decreases to zero by 2050), I assume a constant percentage (39.9%) of seafood production contributed by aquaculture through time.

Historical changes in production efficiency for 1985 through 2011 were estimated using data from FAO[24]: total agricultural land ('arable land plus permanent crops' + 'permanent meadows and pastures'), nitrogen applied to agricultural land, greenhouse gas emissions from agriculture (including from livestock) and area equipped for irrigation. Each of these was used to divide total crop and animal production (in tonnes) to calculate historical resource use efficiency. Linear regressions fit to these historical changes in production efficiency (PE; e.g., tonnes of applied N per tonne of food produced) were then extrapolated to the year 2050 (Supplementary Table 6). Finally, the percent change in overall environmental burden required to support food production (ΔEB) in year x was calculated as:

$$\Delta EB = 100\left[\left(\frac{g_{EM,x} - g_{EM,2009}}{g_{EM,2009}}\right) + \left(\frac{PE_{EM,x} - PE_{EM,2009}}{PE_{EM,2009}}\right)\right] \quad (9)$$

where $PE_{EM,x}$ is the production efficiency in year x estimated from the linear extrapolation of historical PE. If this sum is positive for a particular environmental metric, then its overall environmental burden will likely need to increase – because efficiency changes cannot keep pace – in order to sustain that particular diet.

Lastly, the process of 'sustainable intensification' aims to increase food production through yield improvements while minimizing humanity's pressure on the environment. This approach requires an enhancement in production efficiency (i.e., the amount of food produced per unit amount of resource used). However, when commodities are produced more efficiently, their consumption rates also tend to increase, a phenomenon known as Jevons' Paradox[47]. Because this phenomenon would be inconsistent with the notion of 'sustainable intensification', such interactions between production efficiency and consumption rates have not been addressed in this study. Rather, I investigated scenarios of reduced per capita consumption rates associated with changes in diet.

Results

We estimate that 776 m^3 H_2O, 15.3 kg N, 299 kg CO_2eq and 0.85 ha are required annually to support an average global diet; where available, these estimates agree well with published values in the literature[48,49,50]**.** Not surprisingly, animal products account for much of this required water

(43%), nitrogen (58%), GHG (74%) and land (87%) (Figure 1). By comparison, these products provide 18% of an individual's caloric intake and 39% of protein intake[24]. As expected, I also observe large variation within each footprint intensity of the current diet (Supplementary Table 1a), reflecting the different efficiencies with which food products can be produced in different climates, soil regimes and production systems. While this variation was modest for land use (7% of the mean), it was more substantial for nitrogen (18%) and water (21%).

We also find that substantial changes can occur in the environmental burden of potential future diets. For land use, changes in beef consumption had the most important influence, contributing to a large increase under a GDP-based future and to substantial reductions in land use for other diet scenarios. For other metrics, the changes in environmental burden were distributed more diffusely across commodity groups (Figure 2). For instance, the absence of pork in pescetarian and vegetarian diets contributed to a substantial reduction in per capita GHG emissions. Conversely, the increased consumption of aquaculture seafood in the GDP-based diet led to a sizeable increase in required nitrogen. Interestingly, fruits contribute the largest increase in water demand for the Mediterranean diet. Relative to the current diet, the GDP-based diet required increases in all four environmental burdens, the Mediterranean diet produced apparent tradeoffs (increases in nitrogen and water demand and decreased land and GHG requirements per capita), and pescetarian and vegetarian choices led to consistent and marked decreases.

Finally, in examining the increase in overall human demand, I estimate that average footprint intensities will need to improve substantially (H_2O: 65%, N: 85%, GHG: 72%, Land: 97%) in order to prevent further increases in environmental burdens (Figure 3 (upper panels); Supplementary Table 7). GDP-based growth in food demand likely cannot be met without substantially increasing total resource demand and GHG emissions (Figure 3). With existing technology and production systems, efficiency improvements alone cannot be relied upon – if affluence continues to dictate dietary choices – to minimize the environmental burden of population growth and dietary change. Transitioning to alternative – and generally less impactful – diets would in many cases allow enhancements in footprint intensities to keep pace with growth in human demand and, in turn, prevent growth in overall resource demand and GHG emissions. For instance, the composition of the Mediterranean diet (i.e., increased fruits/vegetables/milk and decreased cereals/beef) minimizes additional land requirements but requires growth in GHG emissions and water and nitrogen demands comparable to the GDP-based diet. Shifting to pescetarian or vegetarian diets reduces environmental burdens relative to other diets and may even decrease all environmental burdens below current levels. Moreover, the similar reductions observed in these two scenarios support our assumptions about seafood footprint intensities and provide further evidence that a transition away from terrestrial animal products – especially ruminants – is an important strategy for reducing the environmental impacts of the food system.

Discussion

Agriculture's growing environmental footprint – Consumption, production, and trade

Sustainable intensification involves enhancing agricultural yields while simultaneously minimizing environmental impacts. Yet, the focus of most recent studies has been on whether and how increases in food production can keep pace with growth in demand (e.g., refs. 16, 22, 51). In light of this, our study attempts to fill an important knowledge gap by providing a much needed assessment of the potential environmental consequences of future food demand. Our findings make apparent that continued improvements in footprint intensities will be insufficient to prevent further increases in the environmental burden of agriculture should dietary trends continue. Altering consumption patterns can yield – in most cases – improvements in resource use and emissions relative to an affluence-based diet and has the potential to contribute to resource savings and emissions reductions when combined with improved production efficiencies (Figure 3). Indeed, shifts in historical demand demonstrate that such changes are possible. For example, the on-going transition in livestock production away from ruminants (e.g., cattle) and towards non-ruminants (e.g., pigs and chickens) has reduced the land and GHG requirements per animal unit and led to an overall plateauing in the sector's land requirements[24,52] – though this has also been accompanied by an increase in nitrogen per animal unit[53]. Achieving continued demand-side changes is the real issue, as historical shifts in diets have been influenced more by accessibility, cost and technology than by government programs or environmental concerns (e.g., refs. 12, 39, 54, 55).

Consumption

Combining economic, nutritional and environmental considerations, several new studies have also shed light on how better to connect dietary changes with improved environmental stewardship. For instance, Jalava et al.[20] showed that – by modifying diets to: 1) reflect nutrient recommendations from the World Health Organization and 2) reduce animal-source proteins – countries could realize substantial water savings from food production. Tilman and Clark[23] also linked healthier diets to improved environmental sustainability, showing that environmentally burdensome diets also have higher incidence of heart disease, diabetes and cancer. In addition, it has been speculated that as societies become more affluent their health and environmental concerns should draw down the rates of meat consumption, according to a Kuznet-like inverted U curve[56]. However, because these changes are expected to take place at (high) income levels that most countries will not attain for the next several decades, it is likely that per capita consumption of animal products will increase globally in the near future. Even without altering diets, reducing consumer food waste – as well as minimizing losses throughout the food supply chain – can decrease environmental impacts and contribute substantially to food security[17,57]. This is particularly true for animal products, with recent studies demonstrating that large crop areas are required to support consumer waste of beef, pork and poultry[58] and that the crops lost via consumer waste of animal foods could feed 235 million people[38]. While this growing body of knowledge shows that healthy diets and responsible food use are also beneficial for the environment, further research is required to identify mechanisms that might effect such changes in consumption patterns.

Production

With regard to production, overall agricultural inputs will likely need to increase, but a continuation of historical gains in major crop yields may be insufficient to meet demand by mid-century[22]. For this reason, certain production increases required to support aspects of the alternative diets (e.g., fruit/vegetable demand of Mediterranean scenario; pulse/oilcrop demand for vegetarian scenario) may therefore be unrealistic to achieve and, in turn, limit the options for modifying diets (Supplementary Table 8). In addition, historical trends in improving yields and production efficiencies may falter in the coming years. For example, crop yields have plateaued or stagnated in many agricultural areas[57] and increases in fertilizer application have resulted in diminishing returns from cereal production over the past several decades[24,59]. Also, large volumes of additional irrigation water (i.e., blue water) will likely be required to further improve crop yields[16,60]. Furthermore, high-yielding cereals – in particular, wheat, rice, and maize – have replaced more nutrient-rich varieties, contributing to diminished nutrient content in the world's cereal supply[61]. These trends based on various studies therefore likely mean that our estimations of additional resource requirements are conservative, as I assumed a linear continuation of improving production efficiencies.

Trade

While it is clear there are obstacles for 'sustainable intensification' of the global food system, the variation within the footprint of each commodity group indicates that there still exists considerable scope for improving the environmental burden of agriculture. Much of this can be explained by three factors: climate, technology and composition. Climate extremes (e.g., heat waves, droughts) can lead to crop failures and animal heat stress. Limited access to advanced techniques, farming equipment, irrigation infrastructure, high-yielding varieties or other agricultural technologies can prevent high yields. And certain products within a commodity group can be more resource-demanding than others. To cope with these stressors, limitations and uncertainties, countries have increasingly turned to international food trade to meet domestic demands. Indeed, food trade has contributed to important resource savings (e.g., ref. 62) and allowed the populations of many countries to exceed what could be supported by locally available resources[18,63,64]. Yet this virtual trade of natural resources appears to have created a disconnect between where food production occurs and where that food is consumed, effectively separating consumers from the environmental impacts of their dietary choices[33,65,66,67]. There is also concern that the global food system has lost resilience and become too rigid and homogeneous to respond to unanticipated climatic and economic shocks[64,68,69]. For example, water-rich countries may soon reduce their virtual water exports in order to preserve domestic food supplies and water resources[70]. Thus while a globalizing food trade system may have allowed for more efficient use of natural resources for food production, these improvements have likely come at the expense of system resilience and nations' long-term food self-sufficiency.

A new food revolution? Beyond changes in efficiency and consumption

These various lines of evidence – unsustainable dietary changes, faltering yield trends and greater reliance on food trade – all point toward the need for a new food revolution combining existing technologies and approaches with a new generation of innovations. While the Green Revolution focused on increasing supply, how those changes would affect the environment was not a primary consideration. Over the past several decades however, the environmental impacts of a rapidly increasing food production have contributed substantially in pushing humankind's footprint to the brink of (or beyond) numerous planetary thresholds[8,9,11,71]. Therefore, a new food revolution should not aim at increased human appropriation of natural resources but at changes in consumer habits and improved efficiencies in the production system. As our projections show, an integrated approach combining efficiency improvements with shifted consumption patterns can simultaneously meet future demand and minimize agriculture's environmental impacts.

Population growth, globalization and urbanization, and climate change make future sustainable agriculture an unprecedented challenge. Yet, there is hope for real improvement in agricultural resource demand, some examples of which I highlight in this final section. For instance, while food trade remains a necessary feature of the global food system, accompanying trade flows with technology transfers can improve the food security outlook for both the importer and exporter. By facilitating such diffusions of technologies from the most efficient countries into under-performing areas, decision-makers can better ensure that projections of resource demand tend towards the lower side of their variabilities, thereby closing the 'technology gap'. Investments in technology, however, are often associated with important shifts between systems of production (e.g., from subsistence farming to large-scale commercial agriculture) that will likely require new policies to protect rural livelihoods and ecosystems. Through technological innovation, import-reliant nations could improve their food self-sufficiency, decrease their dependence on food imports and minimize local environmental impacts. As another example, genetically engineered organisms (GEOs) or transgenic products have received increased attention as a possible avenue for raising yield ceilings, but not without their share of controversy. To be sure, the 'organic movement' is in large part a response to the growing prevalence of GE crops available to consumers. What is less understood is the introduction of GE animals for food. As animal products are generally more environmentally burdensome, intervening to improve their yields and feed conversion efficiencies – while addressing ethical concerns related to animal welfare – could substantially reduce competition for crop use and resource demand. The recent approval of the GE-Atlantic salmon may be that threshold event that presents both great uncertainty and opportunity for more efficient animal products. However, a number of uncertainties remain regarding their related ethics, their potential long-term health and environmental impacts as well as their cultural acceptance and incorporation into diets. Other approaches include land sparing, wildlife-friendly farming[72], vertical farming[73], incorporating insects into feeds/food[74], nutrient capture and recycling (e.g., ref. 75) and better integrated nutrient and energy cycles of crop and animal production[76].

There also exist a host of more speculative – but potentially promising – ways to meet future demand and minimize environmental impacts. One such approach is the large-scale implementation of precision agriculture that utilizes remote sensing and responds in real-time to crop resource requirements and to weather and climatic conditions. Also, with cost being such an important factor in consumer choices, policy-makers can seek a market-based solution for modifying consumption patterns by better incorporating the true environmental costs to produce a food item. While this approach would require the approval of various vested interests, development of valuation criteria, and programs to support access to food and agricultural resources for low-income communities, it could effectively and impartially transition diets towards minimized environmental burdens. This solution could also be combined with internationally defined 'sustainable targets' or caps[9], for which each country would then be allowed to implement the solutions most suitable to its economic, social and environmental landscapes.

Conclusion

The need for both demand- and supply-side solutions to achieve 'sustainable intensification' of the global food system is apparent. Our study quantified the extent to which changes in consumption patterns and efficiency can play a role in improving the environmental footprint of the global food system. If dietary trends continue to grow based on GDP, improvements in efficiency likely will not be sufficient to prevent further increases in agriculture's environmental burden, and additional solutions will be urgently needed. Land use and GHG emissions are the most responsive to changes in diet – in large part due to the reduction/elimination of beef demand – while improvements in nitrogen and water uses were more modest. This indicates that changes to efficiency and consumption patterns are not a panacea for comprehensive reductions in the environmental burden of agriculture but are still essential mechanisms towards realizing environmental sustainability of the global food system. This study provides a useful approach for evaluating the attainability of sustainable targets and for better integrating food security and environmental impacts.

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Tables and Figures

Table 1. Global average demand of current diet and selected diet scenarios. Current diet composition was calculated as the population-weighted average of each country's diet (FAO, 2015a). As a result, an individual country's diet may differ substantially from this average global diet (e.g., no pork consumption in many Middle Eastern countries). For diet scenarios, per capita demand for each commodity group was calculated as the product of current per capita demand and the ratio, r_{kcal}, of 2050 per capita calorie demand to current (circa 2009) per capita calorie demand, as reported by Tilman and Clark[23] (Supplementary Table 1). The r_{kcal} values derived from Tilman and Clark: 1) for 'Fruits/Vegetables' were used for fruits, vegetables and oils, 2) for 'Nuts/Pulses' were used for oilcrops and pulses, and 3) 'Dairy/Eggs' were used for milk and eggs. The composition of the future diet scenarios is therefore determined by a combination of the current diet composition and the r_{kcal} values.

Diet (kg cap^{-1} yr^{-1})	**Current**	**GDP-based**	**Mediterranean**	**Pescetarian**	**Vegetarian**
Cereals	146	147	86	99	106
Fruits	72	53	350	75	75
Oilcrops	7	3	2	10	11
Pulses	7	3	2	10	10
Roots/Tubers	61	74	32	54	58
Sugar crops	24	37	20	20	20
Oils	12	9	28	12	12
Vegetables	131	100	314	136	136
Beef	10	14	5	0	0
Milk	88	135	162	112	159
Pig meat	15	19	2	0	0
Poultry meat	14	14	5	0	0
Eggs	9	13	16	11	16
Seafood	18	30	21	38	0
Total	*613*	*650*	*1044*	*576*	*602*

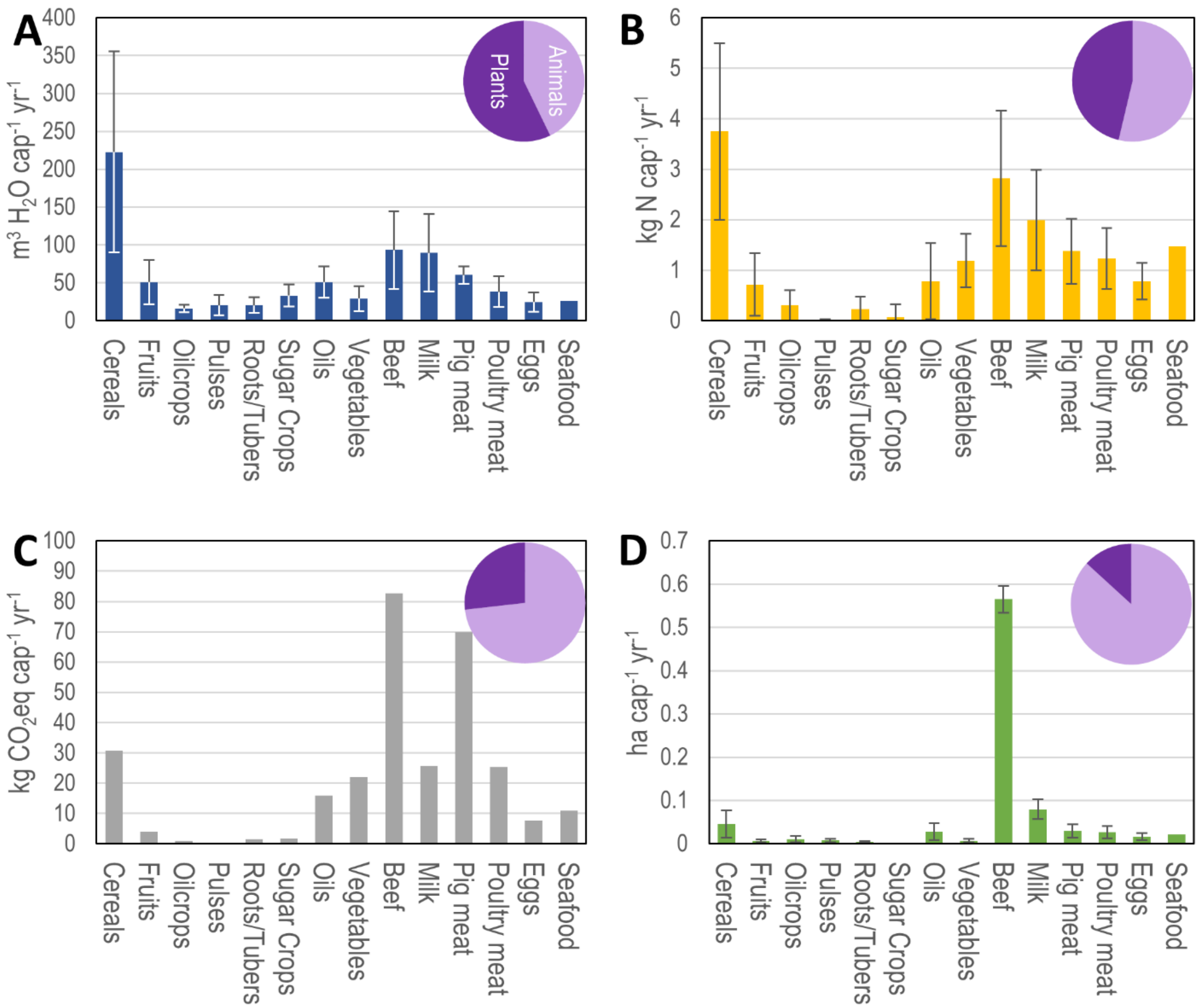


Figure 1. Per capita environmental burdens (EBs) of current diets. Water (A), nitrogen (B), carbon (C), and land (D) footprints associated with the food commodities comprising the average global diet in the year 2009. For N use, the standard deviations of sugar crops and starchy roots were larger than their means. The same was true for carbon use values of starchy roots and vegetables. Uncertainty for beef and milk production only accounts for land use for feed production. Values can be found in Supplementary Table 1a. Pie diagrams (inset) show the relative contribution of plant and animal products to the footprint of current diets.

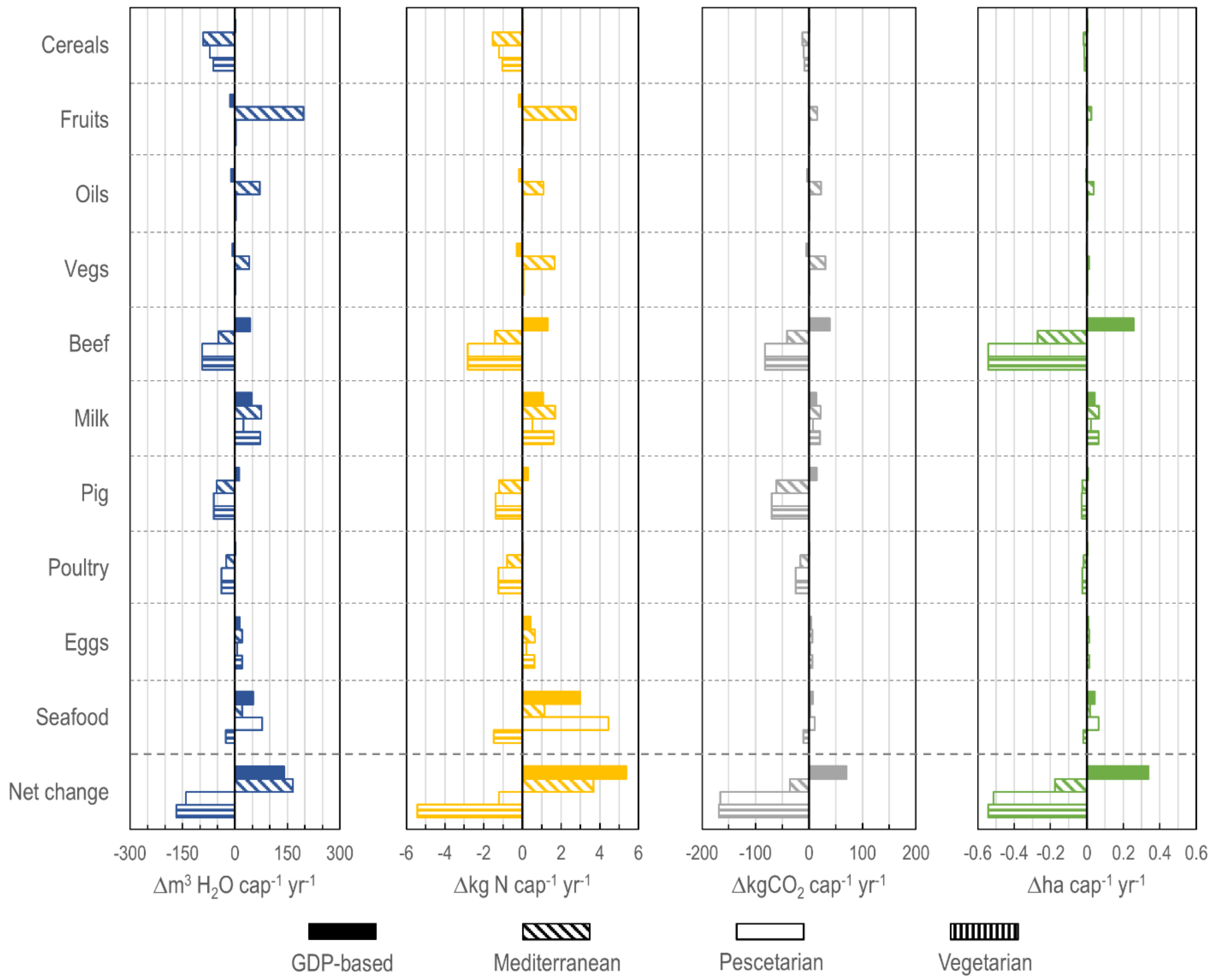


Figure 2. Change in per capita EBs of future diet scenarios. Using year 2009 footprints, bars show the difference in per capita environmental burden between the 2050 scenario diets (GDP-based, Mediterranean, pescetarian, vegetarian) and the 2009 dietary composition. Several commodity groups (oil crops, pulses, roots/tubers, and sugar crops) were not included in this figure because their changes in footprint intensity between diets was generally small in comparison to the groups shown. Detailed information on all commodity groups can be found in the Supplementary Table 1a.

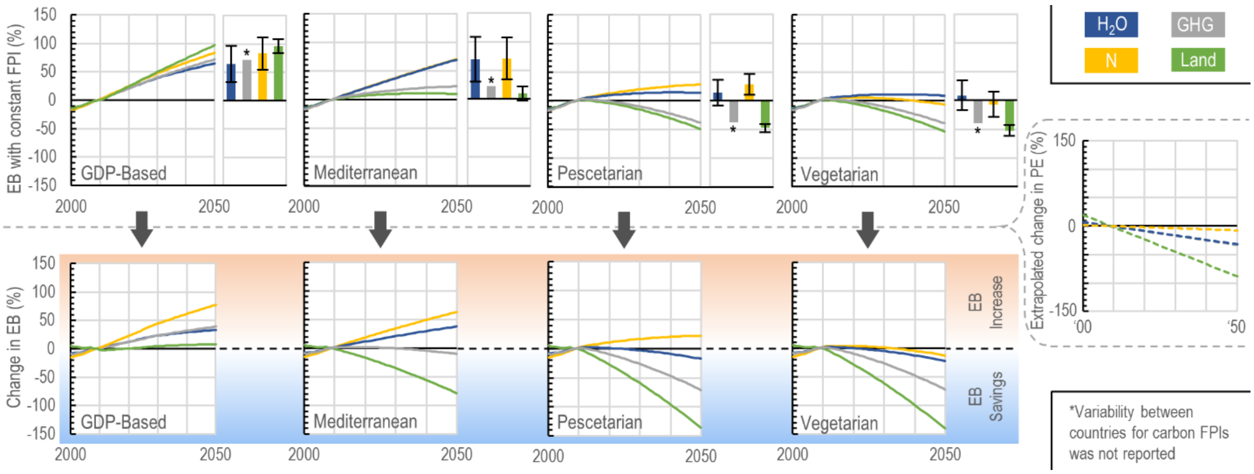


Figure 3. Relative change in overall EBs for different diet scenarios from 2009 to 2050. Upper panels show the product of changing population, changing diets and constant (year 2009) footprint intensities (FPIs), relative to year 2009 environmental burdens. Bar plots represent the variability among countries of relative change in the year 2050. This variability is due to differences in available technologies and agricultural practices as well as to climate, soil texture and other geographic constraints. Because the Tilman and Clark[23] values only reported the standard error between carbon footprint studies that they considered, I do not include an estimate of the variability between countries for carbon footprint. Dashed lines are extrapolations of historical trends (1985-2011) in production efficiency (PE; e.g., tonnes of applied N per tonne of food produced); projected change in H_2O and GHG PEs are nearly identical. Lower panels show the sum of percent change in EB under constant footprints and the percent change in production efficiency. If this sum is positive (i.e., above the x-axis) for a particular environmental metric, then its overall EB will likely need to increase – because efficiency changes cannot keep pace – in order to sustain that diet. Values are presented in Supplementary Tables 6-7.

CONCLUSION

Growth in food production has increased substantially over the past 50 years. The more than tripling of crop supply over this period is often referred to as the Green Revolution, in which higher yielding crop varieties were combined with increases in irrigation, the application of synthetic fertilizers and more extensive mechanized agricultural practices. From a food security perspective, this steady increase in food production provided food for billions more people and reduced incidence of undernourishment and malnutrition globally. At the same time, the various demands on this food production – population growth, dietary changes, crop-based biofuels and intensifying livestock sector – have increased competition for crop use as food, feed and fuel. The greater reliance of many nations on food imports as well as overall growth in the interconnectedness of countries via food trade has also left the system exposed to exogenous shocks that can more easily cascade from country to country. While some countries have made concerted efforts to increase their food self-sufficiency, this vulnerability has also led many investors to acquire agricultural land in the global South, with important impacts on communities and the environment in targeted areas. All of these factors listed above have combined to influence the development of the global food system and made it, on one hand, essential to humanity and on the other hand, pervasive and environmentally impactful. Thus improvements in food security have been made possible through ever accumulating environmental and human impacts as well as possible reductions in the overall resilience of the system.

Humanity's efforts to meet future food demand are, therefore, presented with an apparent and formidable dilemma that has been largely unresolved to date – the need to substantially increase food supply while at the same time minimizing the human and environmental impacts of the food system. My dissertation work has advanced understanding of how the food system has evolved, using the example of the livestock sector to highlight two sets of tradeoffs that have occurred within the global food system at large. The first has occurred within aspects of food security where, on one hand, rapid growth in the demand for animal products has been supported by a greater reliance on crop-based feeds and increased competition for crop use. On the other hand, the increase in livestock production supported by non-feed biomass (e.g., grasses, crop residues) has been a positive for food security, as these animals are able to convert biomass which humans cannot directly consume into usable animal calories and proteins. The second of these tradeoffs has been between food supply and the environment, where growth in animal production has also resulted in increased greenhouse gas emissions, more extensive land requirements, and large amounts of irrigation water and fertilizers required to support feed production.

In addition to steady increases in production – and its associated environmental impacts – increased competition for the use of food production as well as greater vulnerability to shocks (e.g., climate disturbances, changes in trade policies) also characterize some of the important changes in the global food system. However, in many places across the planet, it is possible to substantially increase crop productivity under currently available technologies (e.g., irrigation and fertilizer use). My dissertation builds on previous work to assess this potential to increase crop yields globally, and how combining this intensification with other solutions (e.g., dietary changes, reduced crop-based biofuel production) can greatly increase the number of people fed by crop production. In doing so, this work shows that it is possible to not only feed billions more people but also improve the food self-sufficiency of countries. In this way, countries can reduce their susceptibility to shocks occurring beyond their borders and beyond their immediate control, though the resources required to support this additional crop production may be substantial.

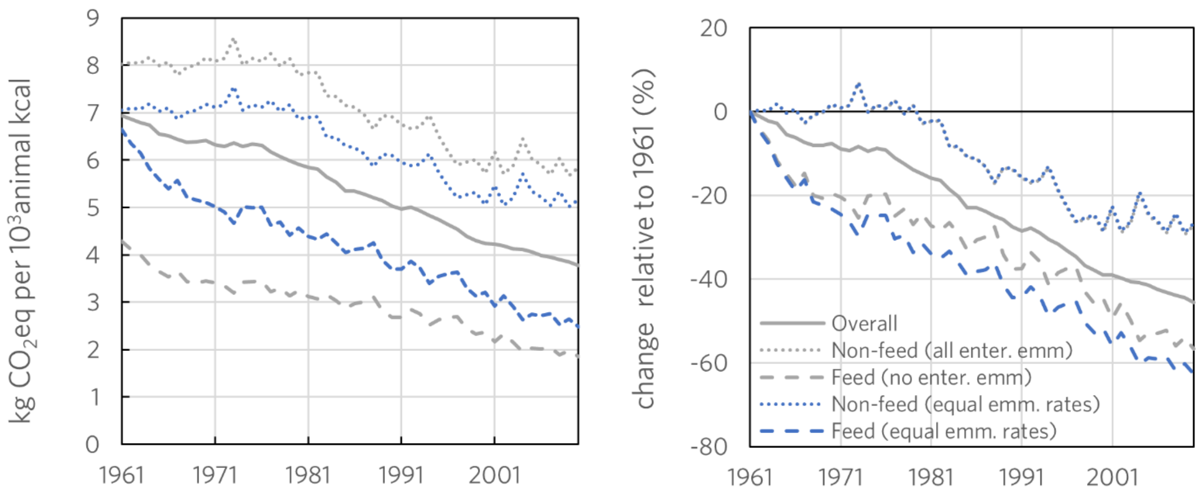
As an alternative to closing yield gaps locally, many countries and investors have also begun acquiring agricultural land in the global South in order to increase the amount of agricultural resources under their control. While these types of investments have been promoted as a shortcut for developing countries to rapidly increase crop yields, to facilitate the influx of more advanced agricultural technologies and to create job opportunities, land acquisitions are many times associated with significant impacts on the communities and environment in targeted areas. Two chapters of my dissertation focused on realized and potential impacts at the global and local scales. My work demonstrated for the first time that the livelihoods of millions of people are potentially vulnerable if all of the acquired land is put under commercial production. My dissertation also utilized a case study of Cambodia, the first ever quantitative assessment to show how land concessions can enhance forest loss and land use change. These studies demonstrate that there is a persistent disconnect between global efforts to increase food supply and the local impacts of those decisions. There remains a need to better involve all stakeholders in these investment decisions as well as to provide more open access to information on land deals, in order to candidly assess their human and environmental impacts.

Ultimately there are various strategies which can be adopted in order to achieve ‘sustainable’ intensification of agriculture. The potential efficacy of improving efficiencies, reducing food waste and moderating consumption patterns was addressed in the final study of my dissertation, which examined how a combination of these solutions can help to integrate considerations of food security, livelihoods and the environment from global to local scales. By considering food supply, future human demand, and the resources associated with the food production required to meet that demand, this study provided a novel evaluation of potential future environmental tradeoffs, showing that adopting these strategies together can offer great promise for meeting future food demand while realizing resource savings relative to current levels of use and impact. And while the multitudinous benefits of such pathways are apparent, two substantial uncertainties remain. The first is how to make such practices the norm, and the second is how quickly such practices can be adopted. While the answers to these questions are unknown, my dissertation work provides a clear path forward, showing that a truly sustainable food system is one which brings food security, livelihoods and the environment into synergy.

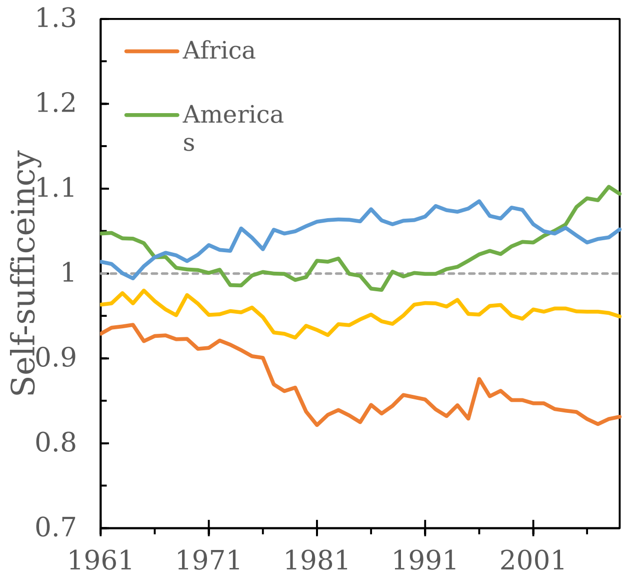
APPENDIX 1 – HISTORICAL TRADEOFFS OF LIVESTOCK'S ENVIRONMENTAL IMPACTS

<u>Supplementary Information</u>

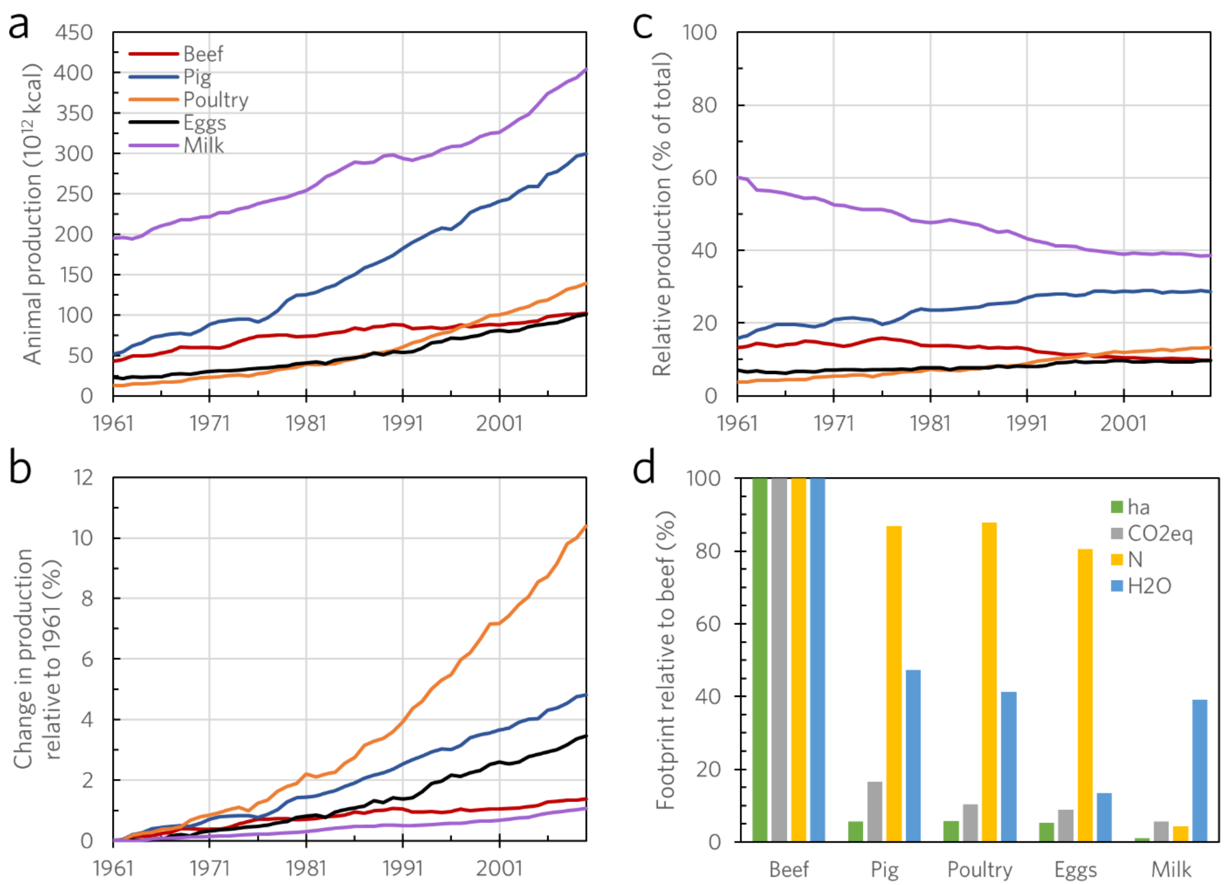
Supplementary tables are provided online through the University of Virginia's LIBRA service. These tables can also be found at: http://iopscience.iop.org/article/10.1088/1748-9326/10/12/125013



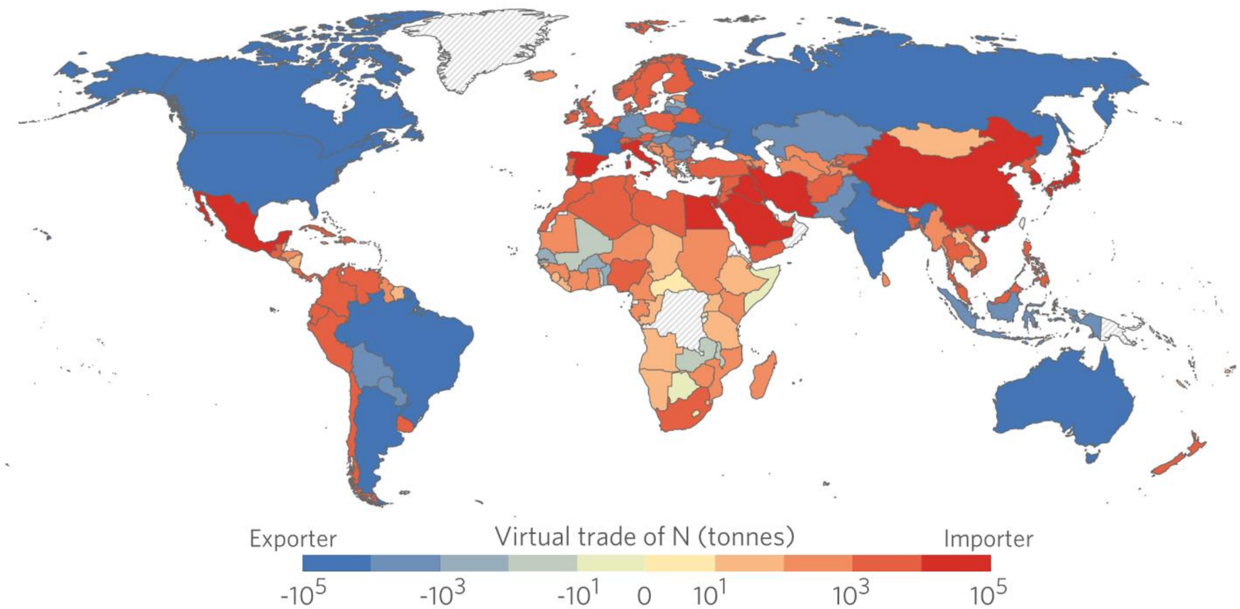
Supplementary Figure 1. Sensitivity of GHG emission intensities to attribution of enteric methane emissions. Attributing all enteric emissions to non-feed sources produces a greater (and more realistic) difference between feed-fed and non-feed GHG emissions per animal calorie than if I assume that enteric emission rates are equal for feed-fed and non-feed sources (blue lines). However this had little effect on the temporal trend and relative changes that I observe.



Supplementary Figure 2. Regional self-sufficiency of animal calorie production. Self-sufficiency was calculated as the local animal production (with waste and feed accounted for) divided by local demand. Any values above 1 thus mean that a region produces more animal calories than were needed to meet domestic human demand. Oceania is not shown because its self-sufficiency value was 1.5 or greater for the entire time period.



Supplementary Figure 3. Comparison of production trends and resource use efficiencies for major animal products. a) Time-series of calorie production for each major animal product. b) Change in calorie production relative to 1961. c) Relative contribution of each animal product to overall calorie production. d) Resource use efficiencies of major animal products relative to beef. Water footprint values came from Mekonnen and Hoekstra (2010).



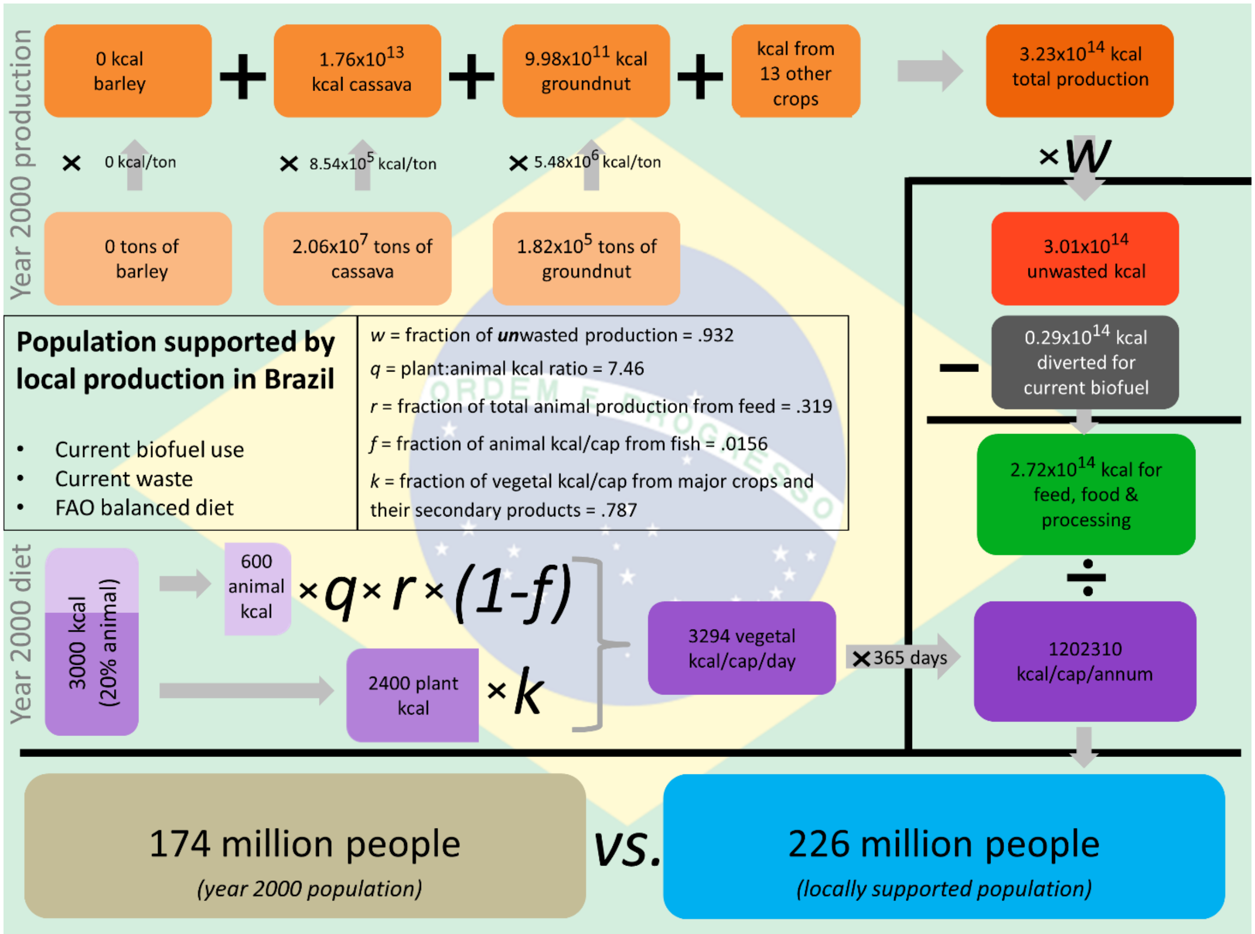
Supplementary Figure 4. Virtual trade of nitrogen via feed. Values are reported in Supplementary Table 10.

References

Mekonnen M M and Hoekstra A Y 2010 *The green, blue and grey water footprint of farm animals and animal products*, Value of Water Research Report Series No. 48 (Delft, the Netherlands: UNESCO-IHE).

APPENDIX 2 – MODERATING DIETS TO FEED THE FUTURE

Supplementary Information



Box A1. Sample calculation of population potentially supported by local production for the year 2000. Calculation was made using a 3000 kcal diet in Brazil with current waste, current biofuel use and current production.

Supplementary Table A1. List of major crops considered.

Barley
Cassava
Groundnuts
Maize
Millet
Oil palm
Potatoes
Rapeseed
Rice
Rye
Sorghum
Soybeans
Sugar beet
Sugarcane
Sunflower
Wheat

Supplementary Table A2. List of conversion factors for each animal product. Values were taken from Pimentel & Pimentel [2008]. Duck meat, goose/guinea fowl meat and other bird meat were assigned the average value for turkey and chicken. Buffalo and cattle meat were assigned the average value of fodder- and grain-fed beef cattle. Each value was divided by 2.5 (unit: kcal fossil fuel input/kcal plant protein), the average input:output ratio for plant calories; this left the units as plant kcal/ animal kcal. Values with (*) represent the average ratio for all animal production, as determined by Pimentel & Pimentel [2008].

Animal product	**kcal Fossil Fuel Input/kcal Animal Protein**
Bird meat (other)	7:1
Buffalo meat	30:1
Cattle meat	30:1
Chicken meat	4:1
Duck meat	7:1
Goat meat	57:1
Goose and guinea fowl meat	7:1
Pig meat	14:1
Sheep meat	57:1
Turkey meat	10:1
Eggs	39:1
Milk	14:1
Offals	25:1*
Animal fat	25:1*
Other meats	25:1*

Supplementary Table A3. List of countries, sub-region, weighted-average plant-to-animal conversion factors, and percent of total animal production from feed. The abbreviations for sub-regions are as follows: EAS – East Asia; IND – Industrial countries; LAC – Latin America and Caribbean; NENA – Near East/ North Africa; SAS – South Asia; SSAF – sub-Saharan Africa; TRAN – Transition countries.

Country	**Sub-region**	**Plant:animal kcal conversion (*q*)**	**Fraction of total animal production from feed (*r*)**
Afghanistan	NENA	9.57	0.26
Albania	TRAN	7.64	0.11
Algeria	NENA	8.95	0.22
Angola	SSAF	9.59	0.32
Argentina	LAC	8.46	0.06
Armenia	TRAN	8.19	0.05
Australia	IND	9.08	0.06
Austria	IND	7.05	0.32
Azerbaijan	TRAN	8.58	0.12
Bahrain	NENA	10.65	0.00
Bangladesh	SAS	9.09	0.09
Barbados	LAC	4.45	0.81
Belarus	TRAN	7.27	0.50
Belize	LAC	4.48	0.00
Benin	SSAF	9.65	0.57
Bhutan	SAS	7.29	0.00
Bolivia	LAC	8.43	0.09
Bosnia and Herzegovina	TRAN	6.89	0.61
Brazil	LAC	7.46	0.32
Brunei Darussalam	EAS	7.30	0.08
Bulgaria	TRAN	7.73	0.30
Burkina Faso	SSAF	10.56	0.00
Burundi	SSAF	9.44	0.00
Cambodia	EAS	8.09	0.04
Cameroon	SSAF	10.14	0.12
Canada	IND	7.33	0.63
Chile	LAC	6.79	0.25
China	EAS	7.92	0.19
Colombia	LAC	7.40	0.11
Congo	SSAF	8.92	0.00
Costa Rica	LAC	7.09	0.30
Cote d'Ivoire	SSAF	10.29	0.16
Croatia	TRAN	7.86	0.84
Cuba	LAC	7.78	0.35

Cyprus	IND	6.93	0.28
Czech Republic	TRAN	7.04	0.53
Denmark	IND	6.50	0.31
Dominica	LAC	6.77	0.01
DRC	SSAF	10.54	0.00
Ecuador	LAC	6.93	0.10
Egypt	NENA	7.74	0.70
El Salvador	LAC	7.35	0.46
Eritrea	SSAF	10.70	0.00
Estonia	TRAN	6.69	0.55
Ethiopia	SSAF	9.87	0.01
Fiji Islands	EAS	7.58	0.01
Finland	IND	6.71	0.22
France	IND	7.05	0.26
French Polynesia	EAS	8.96	0.02
Gabon	SSAF	9.34	0.27
Gambia	SSAF	9.71	0.21
Georgia	TRAN	7.73	0.16
Germany	IND	7.16	0.30
Ghana	SSAF	10.50	0.85
Greece	IND	8.33	0.49
Grenada	IND	8.72	0.62
Guatemala	LAC	7.46	0.33
Guinea	SSAF	10.23	0.42
Guyana	LAC	5.88	0.49
Honduras	LAC	6.94	0.11
Hungary	TRAN	6.48	0.71
Iceland	IND	9.60	0.01
India	SAS	7.01	0.06
Indonesia	EAS	8.63	0.27
Iran	NENA	8.62	0.45
Iraq	NENA	7.60	0.00
Ireland	IND	7.75	0.13
Israel	IND	6.63	0.45
Italy	IND	7.48	0.32
Jamaica	LAC	5.31	0.39
Japan	IND	8.64	0.28
Jordan	NENA	6.64	0.50
Kazakhstan	TRAN	8.24	0.23
Kenya	SSAF	8.58	0.01
Kuwait	NENA	11.52	0.10

Kyrgyzstan	TRAN	8.69	0.16
Latvia	TRAN	6.97	0.37
Lebanon	NENA	7.52	0.09
Libya	NENA	8.13	0.54
Lithuania	TRAN	7.17	0.47
Luxembourg	IND	6.54	0.17
Macedonia	TRAN	8.43	0.37
Madagascar	SSAF	8.49	0.05
Malawi	SSAF	9.39	0.90
Malaysia	EAS	6.83	0.43
Maldives	SAS	10.00	0.00
Mali	SSAF	9.46	0.09
Malta	IND	7.25	0.84
Mauritius	SSAF	6.83	0.54
Mexico	LAC	7.88	0.14
Moldova	TRAN	7.14	0.58
Mongolia	EAS	11.73	0.00
Morocco	NENA	9.47	0.15
Mozambique	SSAF	8.08	0.20
Myanmar	EAS	6.83	0.64
Namibia	SSAF	10.80	0.05
Nepal	SAS	8.33	0.14
Netherlands	IND	7.00	0.17
New Caledonia	EAS	10.14	0.13
New Zealand	IND	8.43	0.06
Nicaragua	LAC	6.94	0.09
Niger	SSAF	10.12	0.08
Nigeria	SSAF	11.86	0.57
Norway	IND	7.68	0.12
Oman	NENA	10.43	0.00
Pakistan	SAS	7.34	0.02
Panama	LAC	7.18	0.23
Papua New Guinea	EAS	9.21	0.00
Paraguay	LAC	9.07	0.18
Peru	LAC	7.76	0.23
Philippines	EAS	6.92	0.22
Poland	TRAN	6.61	0.42
Portugal	IND	7.05	0.36
Qatar	NENA	9.79	0.00
Romania	TRAN	7.11	0.58
Russian Federation	TRAN	7.50	0.37

Rwanda	SSAF	8.60	0.00
Saint Kitts and Nevis	LAC	11.62	0.00
Samoa	EAS	6.76	0.00
Saudi Arabia	NENA	7.53	0.68
Senegal	SSAF	10.30	0.02
Serbia and Montenegro	TRAN	7.24	0.38
Seychelles	SSAF	8.51	0.17
Singapore	EAS	3.93	0.00
Slovak Republic	TRAN	7.12	0.34
Slovenia	TRAN	6.85	0.36
South Africa	IND	8.22	0.26
South Korea	EAS	7.87	0.48
Spain	IND	7.39	0.30
Sri Lanka	SAS	8.28	0.14
Sudan	SSAF	8.88	0.01
Suriname	LAC	7.43	0.82
Sweden	IND	7.13	0.29
Switzerland	IND	6.80	0.11
Syrian Arab Republic	NENA	9.81	0.16
Tajikistan	TRAN	8.47	0.03
Tanzania	SSAF	9.41	0.03
Thailand	EAS	6.60	0.15
Togo	SSAF	9.69	0.35
Tonga	EAS	6.99	0.08
Trinidad and Tobago	LAC	3.26	0.94
Tunisia	NENA	8.47	0.19
Turkey	NENA	8.12	0.17
Uganda	SSAF	8.67	0.29
Ukraine	TRAN	7.40	0.40
United Arab Emirates	NENA	11.13	0.16
United Kingdom	IND	7.15	0.23
United States	IND	7.12	0.64
Uruguay	LAC	9.55	0.02
Venezuela	LAC	7.44	0.10
Vietnam	EAS	6.74	0.14
Yemen	NENA	9.94	0.03
Zambia	SSAF	9.50	0.03
Zimbabwe	SSAF	8.70	0.19

Supplementary Table A4. Population sizes supported under different global diet, biofuel, and yield scenarios.

	Current yield	**50% closure**	**75% closure**	**90% closure**	**100% closure**
Current scenario					
Current diet	5,830,683,543	6,498,310,626	7,800,615,244	8,906,605,750	9,770,533,821
Adequate diet	6,654,212,473	7,280,579,266	8,583,409,262	9,734,811,619	10,661,044,741
2030 diet	4,842,029,350	5,384,828,744	6,454,169,907	7,363,471,177	8,075,271,750
2050 diet	4,550,321,897	5,055,161,322	6,052,638,671	6,902,500,319	7,568,755,275
Projected biofuel					
Current diet	5,157,664,547	5,766,236,455	6,942,207,666	7,932,766,859	8,703,049,534
Adequate diet	5,764,432,807	6,330,287,023	7,493,866,403	8,510,776,180	9,322,811,304
2030 diet	4,281,704,584	4,776,479,082	5,744,242,051	6,559,918,717	7,194,984,483
2050 diet	..	..	..	..	..
No waste/ No biofuel					
Current diet	6,914,036,077	7,690,440,636	9,211,158,062	10,507,308,090	11,522,633,874
Adequate diet	7,838,051,493	8,572,576,418	10,100,996,502	11,452,135,169	12,539,698,958
2030 diet	5,739,379,600	6,371,753,411	7,621,616,690	8,687,913,380	9,524,831,600
2050 diet	5,396,827,377	5,985,006,661	7,151,167,320	8,148,131,328	8,931,861,498
No waste/ No biofuel + Cottonland					
Current diet	7,155,458,203	7,961,163,366	9,541,145,402	10,885,188,353	11,937,567,370
Adequate diet	8,097,072,737	8,855,798,192	10,436,404,109	11,832,479,105	12,956,330,031
2030 diet	5,938,086,203	6,594,261,595	7,892,609,202	8,998,071,806	9,865,330,629
2050 diet	5,582,300,549	6,192,486,186	7,403,624,832	8,436,984,017	9,248,943,538

Supplementary Table A5. List of substituted country diets. Shows the countries for whom the diet of a neighboring country was substituted where FAO Food Balance sheets were not available. The diet used for Serbia and Montenegro was the year 2000 population-weighted average diet and composition.

Country diet missing	**Country diet used**
Afghanistan	Pakistan
Bahrain	Saudi Arabia
Bhutan	India
DRC	Congo
Iraq	Iran
Oman	Saudi Arabia
Papua New Guinea	Indonesia
Qatar	Saudi Arabia
Serbia and Montenegro	Serbia + Montenegro
Singapore	Malaysia
Tonga	Fiji

APPENDIX 3 – LAND GRABBING: A PRELIMINARY QUANTIFICATION OF ECONOMIC IMPACTS ON RURAL LIVELIHOODS

Supplementary Information

Table S1. List of countries and specific crops. For each crop, the grabbed area, yield, net unit price (given as the rounded whole number of the value used) and gross production value (and conversion factor for oil palm) are given. The yield value for bananas grown in Cameroon was used for bananas in Nigeria. The average African oil palm conversion efficiency and unit price were used for Ethiopia, South Sudan/Sudan and Uganda. The unit price for citrus fruit in Morocco was calculated as the production-weighted average of clementines, lemons, limes, oranges, mandarins and tangerines.

Country	Crop	Area (ha)	Yield (t/ha)	Unit Price (USD/t)	Conversion factor	Gross production value lost (USD)
Angola	Oil Palm	81,500	4.7	484	0.196	36,332,951
	Rice	101,500	1.4	309		43,004,861
Argentina	Maize	40,331	5.6	157		35,703,396
	Rapeseed	146,998	1.3	310		58,709,970
	Rice	57,331	5.3	310		94,537,539
	Sorghum	42,650	4.6	171		33,710,629
	Soybeans	156,501	2.2	304		105,774,351
	Wheat	42,555	2.4	175		17,513,319
Benin	Cassava	1,000	11.0	111		1,222,950
	Maize	1,667	1.1	141		267,083
	Oil Palm	1,667	10.8	484	0.169	1,469,594
	Rice	16,000	2.0	231		7,406,623
	Soybeans	1,667	0.5	304		238,518
	Sugar Cane	4,800	35.3	36		6,178,350
Brazil	Maize	21,500	2.9	157		9,939,023
	Oil Palm	59,550	8.7	482	0.214	53,740,609
	Soybeans	21,500	2.4	304		15,709,482
	Sugar Cane	148,550	69.4	36		375,580,725
Cameroon	Bananas	24,930	7.3	313		56,575,778
	Cassava	12,465	11.5	116		16,663,818
	Maize	12,465	2.0	157		3,959,105
	Oil Palm	71,504	15.5	630	0.149	104,109,260
	Rice	12,465	3.2	309		12,298,471

	Sugar Cane	29,141	9.5	36		10,068,689
Colombia	Oil palm	157,876	11.9	482	0.251	227,863,360
	Sugar Cane	125,000	38.5	36		175,445,549
Congo	Cassava	1,000	9.0	116		1,043,522
	Maize	1,667	0.8	157		209,583
	Oil Palm	1,667	12.4	483	0.18	1,794,875
	Rice	16,000	0.7	309		3,515,182
	Sugar Cane	4,800	37.5	36		6,563,902
DRC	Maize	1,016	0.8	157		127,623
	Oil Palm	388,180	3.4	483	0.161	102,531,227
	Rice	7,046	0.7	309		1,612,736
	Soybeans	8,062	0.5	304		1,300,896
Ethiopia	Maize	134,994	1.9	157		39,905,399
	Oil Palm	238,326	3.3	483	0.171	64,565,033
	Rice	257,482	0.7	309		51,764,249
	Soybeans	130,721	2.1	297		81,799,019
	Sugar Cane	186,496	80.1	36		544,318,670
	Wheat	132,571	1.2	175		27,627,929
Gabon	Oil Palm	1,325,016	7.9	483	0.2	1,013,487,025
	Sugar Cane	226,952	51.6	36		426,659,115
Ghana	Bananas	17,570	2.9	339		17,047,018
	Cassava	3,000	11.7	116		4,062,770
	Maize	162,774	1.5	157		37,111,676
	Oil Palm	201,945	9.1	481	0.062	54,660,382
	Rice	171,580	1.9	302		98,602,107
	Sugar Cane	133,333	24.9	36		121,188,373
Guatemala	Oil palm	66,718	12.7	483	0.15	61,324,724
	Sugar Cane	2,500	79.2	37		7,248,923
Indonesia	Oil Palm	3,597,597	16.4	482	0.215	6,102,248,153
	Sugar Cane	826,293	54.3	36		1,633,776,512
Liberia	Oil Palm	588,948	3.3	484	0.241	225,161,293
Madagas.	Maize	26,667	0.9	157		3,731,800
	Oil Palm	9,100	8.6	484	0.191	7,195,111
	Potatoes	150	5.8	187		161,858
	Rice	10,000	2.0	309		6,154,925

	Sugar Cane	130,000	28.8	38		141,054,645
Malaysia	Oil palm	5,192,468	17.9	483	0.2	8,956,266,573
Morocco	Citrus	350,000	9.2	216		692,594,280
	Olives	350,000	0.7	902		233,742,412
Mozam.	Bananas	18,800	6.4	872		105,357,132
	Maize	27,134	0.9	132		3,287,705
	Pineapples	174,552	6.8	1821		2,174,558,887
	Rice	31,800	1.1	443		15,760,238
	Sugar Cane	279,393	14.6	35		144,049,511
Nigeria	Cassava	198,264	10.1	116		232,178,816
	Maize	6,335	1.1	157		1,115,695
	Oil Palm	26,787	2.6	348	0.162	3,896,213
	Rice	224,183	1.3	309		92,219,983
	Sorghum	7,000	1.1	171		1,338,380
	Soybeans	4,085	0.8	304		1,032,334
Papua New Guinea	Oil Palm	2,140,539	11.0	483	0.276	3,130,086,466
	Sugar cane	315,005	54.7	36		628,098,318
Peru	Oil Palm	16,834	2.8	845	0.138	5,434,684
	Sugar Cane	104,831	29.2	37		113,689,948
Philippines	Bananas	32,266	12.9	313		130,010,558
	Cassava	20,000	5.3	116		12,359,922
	Maize	153,500	0.9	157		20,998,371
	Oil Palm	70,000	12.1	484	0.174	71,321,717
	Pineapples	21,833	36.3	316		250,743,882
	Rice	253,671	2.8	309		219,684,572
	Sugar Cane	50,000	65.9	30		98,899,386
Russia	Sugar Beet	30,000	19.3	48		27,585,683
Sierra Leone	Cassava	144,051	5.2	116		86,517,319
	Maize	31,823	0.9	157		4,653,636
	Oil Palm	701,057	3.3	483	0.239	265,634,518
	Rice	146,727	1.1	309		49,465,898
	Sugar Cane	37,085	70.4	36		95,195,819

S. Sudan & Sudan	Maize	681,169	0.7	157		79,253,919
	Oil Palm	438,101	3.3	483	0.171	118,686,153
	Rice	113,346	1.5	309		51,536,481
	Sorghum	534,622	0.6	171		53,847,119
	Sugar Cane	667,078	78.4	36		1,906,371,006
	Sunflower	213,323	0.7	70		10,941,060
	Wheat	2,107,995	2.3	273		1,340,624,635
Tanzania	Barley	2,940	0.1	132		46,595
	Maize	18,570	1.5	157		4,350,670
	Oil Palm	55,118	5.3	485	0.098	13,757,985
	Rice	40,932	1.4	309		18,230,367
	Sorghum	70,000	1.9	171		23,182,712
	Sugar Cane	74,000	91.0	36		245,487,123
Uganda	Oil Palm	71,012	3.3	483	0.171	19,237,881
Uruguay	Barley	71,345	0.9	132		8,102,809
	Maize	71,345	3.5	157		38,702,819
	Soybeans	71,345	1.9	304		41,921,226
	Wheat	71,345	2.1	175		26,363,342
Total		**27,107,998**				**34,262,003,020**

APPENDIX 4 – ACCELERATED DEFORESTATION DRIVEN BY LARGE-SCALE LAND ACQUISITIONS IN CAMBODIA

Supplementary Information

Supplementary Table 1. Summary of changes to forest cover in Cambodia and within ELCs.

	Cambodia	Acquired land	% of Cambodian total
Total area (10^6 ha)	17.91	2.05	11.4%
Total forested area in 2000 (10^6 ha)	7.49	0.93	12.4%
Forested area in 2000 (% of total land area)	41.9%	45.2%	-
Total forest loss during 2000-2012 (10^6 ha)	1.31	0.26	19.8%
Forest loss during 2000-2012 (%)	17.5%	28.0%	-

Supplementary Table 2. Summary of deforestation rates for matched plots under different selection criteria. r_{ELC} is the average percent annual forest loss in ELCs for the year 2010 – 2012. r_{NonELC} is the average percent annual forest loss in Non-ELC areas for the year 2010 – 2012. Percent difference between 2010-2012 average deforestation rates of ELC and non-ELC areas, calculated as 100 * (r_{ELC} - r_{NonELC}) / r_{NonELC}. 'P.A. buffer' excludes from consideration any plots that are within 2 km of a protected area. 'ELC buffer' excludes from consideration any non-ELC plots that are within 2 km of an ELC boundary.

	N matched pairs	r_{ELC} (%)	r_{NonELC} (%)	% difference from r_{NonELC}
All	28439	4.29	3.26	31.8
All (P.A. buffer)	26784	4.37	3.38	29.2
All (ELC buffer)	28439	4.29	3.13	37.0
All (P.A. buffer + ELC buffer)	26784	4.37	3.37	29.8
2001-2006	5711	5.04	3.53	42.6
2001-2006 (P.A. buffer)	5395	5.09	3.60	41.4
2001-2006 (ELC buffer)	5711	5.04	3.08	63.8
2001-2006 (P.A. buffer + ELC buffer)	5395	5.09	3.46	47.1
2007-2012	8724	5.17	2.52	104.9
2007-2012 (P.A. buffer)	8418	5.23	2.60	101.3
2007-2012 (ELC buffer)	8724	5.17	3.04	70.2
2007-2012 (P.A. buffer + ELC buffer)	8418	5.23	3.32	57.8

Supplementary Table 3. Covariate matching results for all matched pairs (no buffers around ELCs or protected areas). ‘Mean eQQ diff’, ‘median eQQ diff’ and ‘max eQQ diff’ are the mean, median, and maximum differences in the empirical quantile-quantile plot of treatment and control groups. The eQQ values for each variable were measured on the scale of that variable. ‘Mean eCDF diff’ is the mean difference in the cumulative distribution functions. Improved covariate balance is evidenced when the difference in mean values, the mean eQQ difference and the mean difference in cumulative distribution functions move towards zero.

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	6.1814	6.836	-0.6546	0.6866	0.3805	20.795	0.0242
Matched	6.1814	6.0454	0.136	0.1506	0.0754	4.0642	0.0078
Distance from river							
Unmatched	5.2543	4.792	0.4623	0.4627	0.4401	2.1146	0.0341
Matched	5.2543	5.1546	0.0997	0.123	0.1039	0.7297	0.0084
Distance from railroad							
Unmatched	190.96	153.11	37.85	40.548	43.924	72.54	0.1158
Matched	190.96	191.79	-0.83	2.9752	1.7338	13.346	0.009
Distance from city							
Unmatched	48.753	42.641	6.112	6.1183	6.1951	28.609	0.0544
Matched	48.753	48.204	0.549	0.9525	0.5844	6.0205	0.0083
Distance from forest edge							
Unmatched	9.7298	13.208	-3.4782	3.4789	3.2934	27.907	0.0664
Matched	9.7298	9.7918	-0.062	0.3745	0.2153	5.8024	0.0077
Slope class							
Unmatched	3.9929	4.0316	-0.0387	0.2433	0	2	0.0303
Matched	3.9929	4.002	-0.0091	0.0144	0	1	0.0024
Soil suitability class							
Unmatched	5.9688	6.0659	-0.0971	0.0971	0	1	0.0194
Matched	5.9688	5.9567	0.0121	0.0143	0	1	0.0029
District area							
Unmatched	2505.5	2219.8	285.7	285.78	187.04	1595.6	0.0324
Matched	2505.5	2502	3.5	18.852	0	755.74	0.0058

Supplementary Table 4. Covariate matching results for matched pairs with an ELC contract date between 2001 and 2006 (no buffers around ELCs or protected areas). For a definition of column labels

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	4.0488	6.836	-2.7872	2.7901	1.874	23.89	0.1402
Matched	4.0488	4.0587	-0.0099	0.1031	0.0739	1.0568	0.0096
Distance from river							
Unmatched	5.0231	4.792	0.2311	0.2583	0.0443	2.7482	0.0078
Matched	5.0231	4.9677	0.0554	0.0948	0.0621	1.7139	0.0056
Distance from railroad							
Unmatched	193.57	153.11	40.46	43.024	40.155	86.496	0.1322
Matched	193.57	193.65	-0.08	1.9077	1.3826	29.726	0.001
Distance from city							
Unmatched	64.254	42.641	21.613	21.626	22.079	43.758	0.1782
Matched	64.254	64.26	-0.006	1.0757	0.7246	8.5471	0.0082
Distance from forest edge							
Unmatched	12.842	13.208	-0.366	3.5182	3.2106	36.067	0.0731
Matched	12.842	12.546	0.296	0.4955	0	3.1171	0.0117
Slope class							
Unmatched	4.1014	4.0316	0.0698	0.2324	0	3	0.0289
Matched	4.1014	4.1122	-0.0108	0.0109	0	1	0.0027
Soil suitability class							
Unmatched	5.8601	6.0659	-0.2058	0.2057	0	2	0.0412
Matched	5.8601	5.8625	-0.0024	0.0053	0	1	0.0011
District area							
Unmatched	2646.3	2219.8	426.5	444.21	341.19	1599.5	0.0605
Matched	2646.3	2635.8	10.5	25.345	0	533.64	0.0128

Supplementary Table 5. Covariate matching results for matched pairs with an ELC contract date between 2007 and 2012 (no buffers around ELCs or protected areas).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	7.5771	6.836	0.7411	0.9937	0.8708	22.268	0.0546
Matched	7.5771	7.4181	0.159	0.2325	0.1705	1.6252	0.011
Distance from river							
Unmatched	4.489	4.792	-0.303	0.5015	0.2308	5.6174	0.0284
Matched	4.489	4.428	0.061	0.0956	0.0751	1.0066	0.0086
Distance from railroad							
Unmatched	229.33	153.11	76.22	76.246	77.826	114.3	0.2161
Matched	229.33	227.99	1.34	2.6236	1.9129	24.999	0.0101
Distance from city							
Unmatched	45.649	42.641	3.008	5.9265	4.7101	29.449	0.064
Matched	45.649	45.494	0.155	0.9256	0.8859	4.0494	0.0144
Distance from forest edge							
Unmatched	9.6012	13.208	-3.6068	3.8323	0.728	63.484	0.051
Matched	9.6012	9.4066	0.1946	0.4039	0.3034	1.6438	0.0142
Slope class							
Unmatched	3.9956	4.0316	-0.036	0.3045	0	3	0.038
Matched	3.9956	3.9966	-0.001	0.0023	0	1	0.0005
Soil suitability class							
Unmatched	5.7901	6.0659	-0.2758	0.2758	0	2	0.0552
Matched	5.7901	5.7909	-0.0008	0.0008	0	1	0.0002
District area							
Unmatched	2782.8	2219.8	563	571.92	289.57	2580.5	0.044
Matched	2782.8	2770.3	12.5	17.147	0	295.32	0.0061

Supplementary Table 6. Covariate matching results for all matched pairs (P.A. buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	6.2207	6.8031	-0.5824	0.6073	0.4074	15.529	0.0241
Matched	6.2207	6.0431	0.1776	0.1824	0.0939	1.1242	0.0092
Distance from river							
Unmatched	5.4055	4.9308	0.4747	0.4751	0.4549	2.1146	0.035
Matched	5.4055	5.3131	0.0924	0.125	0.1058	0.7332	0.0086
Distance from railroad							
Unmatched	194.05	155.21	38.84	41.097	43.47	71.19	0.1198
Matched	194.05	194.79	-0.74	3.176	1.7911	15.907	0.0093
Distance from city							
Unmatched	48.594	42.575	6.019	6.0207	6.2216	28.829	0.0563
Matched	48.594	47.873	0.721	1.0466	0.6047	7.7571	0.009
Distance from forest edge							
Unmatched	9.4304	12.509	-3.0786	3.0796	2.8232	37.146	0.0634
Matched	9.4304	9.4447	-0.0143	0.3732	0.1201	3.967	0.008
Slope class							
Unmatched	3.9511	3.9874	-0.0363	0.2359	0	3	0.0295
Matched	3.9511	3.9598	-0.0087	0.015	0	1	0.003
Soil suitability class							
Unmatched	5.955	6.0514	-0.0964	0.0964	0	1	0.0193
Matched	5.955	5.9411	0.0139	0.0169	0	1	0.0034
District area							
Unmatched	2533.8	2209.2	324.6	324.66	201.32	1595.6	0.0355
Matched	2533.8	2527.7	6.1	22.306	0	755.74	0.0076

Supplementary Table 7. Covariate matching results for matched pairs with an ELC contract date between 2001 and 2006 (P.A. buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	4.0696	6.8031	-2.7335	2.7358	1.8661	18.624	0.1439
Matched	4.0696	4.05	0.0196	0.0848	0.0617	1.0129	0.0074
Distance from river							
Unmatched	5.2099	4.9308	0.2791	0.2829	0.0606	2.7173	0.0093
Matched	5.2099	5.1426	0.0673	0.1033	0.0706	1.7139	0.0062
Distance from railroad							
Unmatched	194.76	155.21	39.55	42.682	42.465	83.186	0.1352
Matched	194.76	194.79	-0.03	2.3109	1.8862	29.561	0.0124
Distance from city							
Unmatched	64.39	42.575	21.815	21.817	21.313	42.991	0.1852
Matched	64.39	64.021	0.369	1.42	0.7739	11.304	0.0099
Distance from forest edge							
Unmatched	11.708	12.509	-0.801	3.7784	3.8752	37.146	0.0863
Matched	11.708	11.36	0.348	0.5456	0	3.1974	0.0128
Slope class							
Unmatched	4.0539	3.9874	0.0665	0.2002	0	3	0.0249
Matched	4.0539	4.0654	-0.0115	0.0115	0	1	0.0029
Soil suitability class							
Unmatched	5.8328	6.0514	-0.2186	0.2185	0	2	0.0437
Matched	5.8328	5.8343	-0.0015	0.0082	0	1	0.0016
District area							
Unmatched	2707.5	2209.2	498.3	507.09	399.4	1952.1	0.0674
Matched	2707.5	2681.3	26.2	42.337	0	592.08	0.0201

Supplementary Table 8. Covariate matching results for matched pairs with an ELC contract date between 2007 and 2012 (P.A. buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	7.6176	6.8031	0.8145	0.9779	0.8766	17.002	0.0551
Matched	7.6176	7.4022	0.2154	0.2713	0.1948	2.1691	0.0123
Distance from river							
Unmatched	4.5731	4.9308	-0.3577	0.528	0.2083	5.6174	0.0292
Matched	4.5731	4.521	0.0521	0.0927	0.0757	1.0468	0.008
Distance from railroad							
Unmatched	228.84	155.21	73.63	73.64	74.468	117.07	0.2097
Matched	228.84	227.05	1.79	2.7403	1.8675	24.832	0.0106
Distance from city							
Unmatched	44.642	42.575	2.067	5.9702	5.2138	24.523	0.0669
Matched	44.642	44.51	0.132	1.0142	0.9119	3.7275	0.016
Distance from forest edge							
Unmatched	9.6712	12.509	-2.8378	3.1059	0.7805	63.484	0.0423
Matched	9.6712	9.3948	0.2764	0.4403	0.2601	1.7277	0.0161
Slope class							
Unmatched	3.9797	3.9874	-0.0077	0.2521	0	3	0.0314
Matched	3.9797	3.9804	-0.0007	0.0007	0	1	0.0001
Soil suitability class							
Unmatched	5.766	6.0514	-0.2854	0.2855	0	2	0.0571
Matched	5.766	5.7672	-0.0012	0.0012	0	1	0.0002
District area							
Unmatched	2787.4	2209.2	578.2	588.65	291.7	2580.5	0.046
Matched	2787.4	2775	12.4	18.725	0	435.23	0.0064

Supplementary Table 9. Covariate matching results for all matched pairs (ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	6.1814	6.836	-0.6546	0.6866	0.3805	20.795	0.0242
Matched	6.1814	6.0454	0.136	0.1506	0.0754	4.0642	0.0078
Distance from river							
Unmatched	5.2543	4.792	0.4623	0.4627	0.4401	2.1146	0.0341
Matched	5.2543	5.1546	0.0997	0.123	0.1039	0.7297	0.0084
Distance from railroad							
Unmatched	190.96	153.11	37.85	40.548	43.924	72.54	0.1158
Matched	190.96	191.79	-0.83	2.9752	1.7338	13.346	0.009
Distance from city							
Unmatched	48.753	42.641	6.112	6.1183	6.1951	28.609	0.0544
Matched	48.753	48.204	0.549	0.9525	0.5844	6.0205	0.0083
Distance from forest edge							
Unmatched	9.7298	13.208	-3.4782	3.4789	3.2934	27.907	0.0664
Matched	9.7298	9.7918	-0.062	0.3745	0.2153	5.8024	0.0077
Slope class							
Unmatched	3.9929	4.0316	-0.0387	0.2433	0	2	0.0303
Matched	3.9929	4.002	-0.0091	0.0144	0	1	0.0024
Soil suitability class							
Unmatched	5.9688	6.0659	-0.0971	0.0971	0	1	0.0194
Matched	5.9688	5.9567	0.0121	0.0143	0	1	0.0029
District area							
Unmatched	2505.5	2219.8	285.7	285.78	187.04	1595.6	0.0324
Matched	2505.5	2502	3.5	18.852	0	755.74	0.0058

Supplementary Table 10. Covariate matching results for matched pairs with an ELC contract date between 2001 and 2006 (ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	4.0488	6.9583	-2.9095	2.9123	1.9938	23.89	0.1479
Matched	4.0488	4.153	-0.1042	0.2328	0.181	5.3692	0.0194
Distance from river							
Unmatched	5.0231	4.8163	0.2068	0.2652	0.0816	2.854	0.0085
Matched	5.0231	4.9336	0.0895	0.1759	0.1071	2.348	0.0322
Distance from railroad							
Unmatched	193.57	148.1	45.47	47.138	46.669	91.384	0.1456
Matched	193.57	190.74	2.83	5.3698	3.4746	43.212	0.0204
Distance from city							
Unmatched	64.254	42.061	22.193	22.202	22.399	44.377	0.1829
Matched	64.254	62.837	1.417	2.9218	1.5699	13.352	0.0207
Distance from forest edge							
Unmatched	12.842	13.494	-0.652	3.6664	3.6865	36.067	0.0747
Matched	12.842	12.642	0.2	0.8684	0.1007	4.8822	0.0167
Slope class							
Unmatched	4.1014	4.0339	0.0675	0.2609	0	3	0.0324
Matched	4.1014	4.0783	0.0231	0.0399	0	1	0.001
Soil suitability class							
Unmatched	5.8601	6.0698	-0.2097	0.2096	0	2	0.0419
Matched	5.8601	5.8328	0.0273	0.0315	0	1	0.0063
District area							
Unmatched	2646.3	2211.5	434.8	455.1	344.58	1599.5	0.0634
Matched	2646.3	2599	47.3	71.224	0	984.59	0.029

Supplementary Table 11. Covariate matching results for matched pairs with an ELC contract date between 2007 and 2012 (ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	7.5771	6.9583	0.6188	0.9079	0.7832	22.268	0.0481
Matched	7.5771	7.2995	0.2776	0.4187	0.3732	2.6585	0.0199
Distance from river							
Unmatched	4.489	4.8163	-0.3273	0.4879	0.193	5.6174	0.0269
Matched	4.489	4.4152	0.0738	0.1975	0.1708	2.4804	0.0174
Distance from railroad							
Unmatched	229.33	148.1	81.23	81.246	82.817	123.04	0.2301
Matched	229.33	226.19	3.14	5.7297	4.87	24.999	0.02
Distance from city							
Unmatched	45.649	42.061	3.588	6.3167	5.0132	35.346	0.0675
Matched	45.649	45.634	0.015	2.2076	2.2043	5.9371	0.0328
Distance from forest edge							
Unmatched	9.6012	13.494	-3.8928	4.0751	0.6672	63.484	0.0552
Matched	9.6012	8.9576	0.6436	0.8454	0.414	3.5271	0.0319
Slope class							
Unmatched	3.9956	4.0339	-0.0383	0.3329	0	3	0.0415
Matched	3.9956	4.0076	-0.012	0.0122	0	1	0.0024
Soil suitability class							
Unmatched	5.7901	6.0698	-0.2797	0.2796	0	2	0.0559
Matched	5.7901	5.8223	-0.0322	0.0324	0	1	0.0065
District area							
Unmatched	2782.8	2211.5	571.3	579.36	291.7	2580.5	0.0466
Matched	2782.8	2782.4	0.4	32.473	0	435.23	0.0111

Supplementary Table 12. Covariate matching results for all matched pairs (P.A. buffer + ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	6.2207	6.9195	-0.6988	0.7069	0.5307	15.529	0.0297
Matched	6.2207	5.9656	0.2551	0.3102	0.2828	1.2518	0.0179
Distance from river							
Unmatched	5.4055	4.9482	0.4573	0.4578	0.4264	2.1146	0.0332
Matched	5.4055	5.2874	0.1181	0.2434	0.2405	1.4587	0.018
Distance from railroad							
Unmatched	194.05	150.19	43.86	45.866	49.421	75.571	0.1328
Matched	194.05	192.54	1.51	5.7567	4.8889	19.7	0.0168
Distance from city							
Unmatched	48.594	42.129	6.465	6.4693	6.6639	31.314	0.0599
Matched	48.594	47.434	1.16	1.9988	1.6654	10.061	0.0191
Distance from forest edge							
Unmatched	9.4304	12.761	-3.3306	3.3316	2.9188	37.461	0.0674
Matched	9.4304	9.5095	-0.0791	0.8358	0.239	9.1536	0.0168
Slope class							
Unmatched	3.9511	3.9889	-0.0378	0.2628	0	3	0.0328
Matched	3.9511	3.9635	-0.0124	0.02	0	1	0.004
Soil suitability class							
Unmatched	5.955	6.0571	-0.1021	0.265	0	1	0.0204
Matched	5.955	5.9517	0.0033	0.0175	0	1	0.0036
District area							
Unmatched	2533.8	2198.7	335.1	335.11	211.95	1595.6	0.0384
Matched	2533.8	2528.8	5	42.016	0	985.33	0.0144

Supplementary Table 13. Covariate matching results for matched pairs with an ELC contract date between 2001 and 2006 (P.A. buffer + ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	4.0696	6.9195	-2.8499	2.8522	1.9917	18.624	0.1518
Matched	4.0696	4.1512	-0.0816	0.2288	0.1869	5.3692	0.0201
Distance from river							
Unmatched	5.2099	4.9482	0.2617	0.2734	0.0425	2.7947	0.0083
Matched	5.2099	5.1201	0.0898	0.2167	0.1476	2.4077	0.0135
Distance from railroad							
Unmatched	194.76	150.19	44.57	46.676	49.104	87.644	0.1489
Matched	194.76	192.75	2.01	5.0996	3.7366	43.718	0.0201
Distance from city							
Unmatched	64.39	42.129	22.261	22.262	21.618	43.537	0.1887
Matched	64.39	63.345	1.045	2.7524	1.5772	13.536	0.0197
Distance from forest edge							
Unmatched	11.708	12.761	-1.053	3.9314	4.1121	37.146	0.0878
Matched	11.708	11.462	0.246	0.7708	0.0481	6.168	0.016
Slope class							
Unmatched	4.0539	3.9889	0.065	0.223	0	3	0.0278
Matched	4.0539	4.0254	0.0285	0.0308	0	1	0.0077
Soil suitability class							
Unmatched	5.8328	6.0571	-0.2243	0.2245	0	2	0.0449
Matched	5.8328	5.8117	0.0211	0.0445	0	1	0.0089
District area							
Unmatched	2707.5	2198.7	508.8	519.91	399.4	1952.1	0.0704
Matched	2707.5	2659.5	48	69.081	0	592.08	0.0312

Supplementary Table 14. Covariate matching results for matched pairs with an ELC contract date between 2007 and 2012 (P.A. buffer + ELC buffer).

	Mean ELC plots	Mean control plots	Diff in mean value	Mean eQQ diff	Median eQQ diff	Max eQQ diff	Mean eCDF diff
Distance from road							
Unmatched	7.6176	6.9195	0.6981	0.8817	0.7749	17.002	0.0485
Matched	7.6176	7.2566	0.361	0.4857	0.406	2.6193	0.0219
Distance from river							
Unmatched	4.5731	4.9482	-0.3751	0.5225	0.1856	5.6174	0.0284
Matched	4.5731	4.5081	0.065	0.2038	0.1604	2.4804	0.0179
Distance from railroad							
Unmatched	228.84	150.19	78.65	78.654	77.162	125.94	0.2237
Matched	228.84	224.55	4.29	6.5783	4.9943	24.999	0.023
Distance from city							
Unmatched	44.642	42.129	2.513	6.308	5.565	30.549	0.0702
Matched	44.642	44.804	-0.162	2.6511	2.2873	7.1738	0.0378
Distance from forest edge							
Unmatched	9.6712	12.761	-3.0898	3.3062	0.7179	63.484	0.0455
Matched	9.6712	9.0448	0.6264	0.8366	0.3854	3.4749	0.0312
Slope class							
Unmatched	3.9797	3.9889	-0.0092	0.2751	0	3	0.0343
Matched	3.9797	3.99	-0.0103	0.0125	0	1	0.0025
Soil suitability class							
Unmatched	5.766	6.0571	-0.2911	0.2912	0	2	0.0582
Matched	5.766	5.8004	-0.0344	0.0425	0	1	0.0085
District area							
Unmatched	2787.4	2198.7	588.7	598.06	319.08	2580.5	0.0486
Matched	2787.4	2781.2	6.2	38.471	0	435.23	0.0121

Supplementary Table 15. Rosenbaum's test of sensitivity to hidden bias for matched pairs (no buffers around ELCs or protected areas).

Γ	All	2001 - 2006	2007 - 2012
1.00	<0.001	<0.001	<0.001
1.25	0.313	<0.001	<0.001
1.50	~1	<0.001	<0.001
1.75	~1	0.026	0.598
2.00	~1	0.719	~1

Supplementary Table 16. Test of sensitivity to hidden bias for matched pairs (excluding plots within 2 km of protected areas).

Γ	All	2001 - 2006	2007 - 2012
1.00	<0.001	<0.001	<0.001
1.25	0.944	<0.001	<0.001
1.50	~1	<0.001	0.007
1.75	~1	0.094	0.799
2.00	~1	0.873	~1

Supplementary Table 17. Test of sensitivity to hidden bias for matched pairs (excluding non-ELC plots within 2 km of ELCs).

Γ	All	2001 - 2006	2007 - 2012
1.00	<0.001	<0.001	<0.001
1.25	0.313	<0.001	0.792
1.50	~1	<0.001	~1
1.75	~1	<0.001	~1
2.00	~1	<0.001	~1

Supplementary Table 18. Test of sensitivity to hidden bias for matched pairs (excluding plots within 2 km of protected areas *and* non-ELC plots within 2 km of ELCs).

Γ	All	2001 - 2006	2007 - 2012
1.00	<0.001	<0.001	0.007
1.25	~1	<0.001	0.988
1.50	~1	<0.001	~1
1.75	~1	0.006	~1
2.00	~1	0.461	~1

Supplementary Table 19. Classes for median terrain slope and agro-ecological suitability for rain-fed, high input oil palm. For a detailed description of how slope gradient and suitability index (SI) were calculated, see ref. 32.

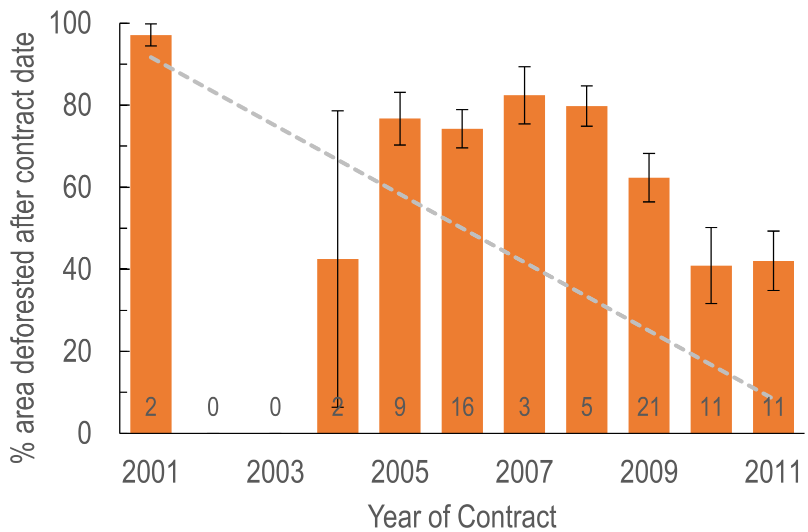
Class	Slope gradient	Soil suitability
1	0-0.5%	Very high (SI > 85)
2	0.5-2%	High (SI > 70)
3	2-5%	Good (SI > 55)
4	5-8%	Medium (SI > 40)
5	8-16%	Moderate (SI > 25)
6	16-30%	Marginal (SI > 10)
7	30-45%	Very marginal (SI > 0)
8	>45%	Not suitable (SI = 0)

Supplementary Table 20. Summary of tree cover and 'native' forest misidentification before and after accounting for tree plantations on 29 randomly selected ELCs. ELCs in bold are those shown in Supplementary Figure 2.

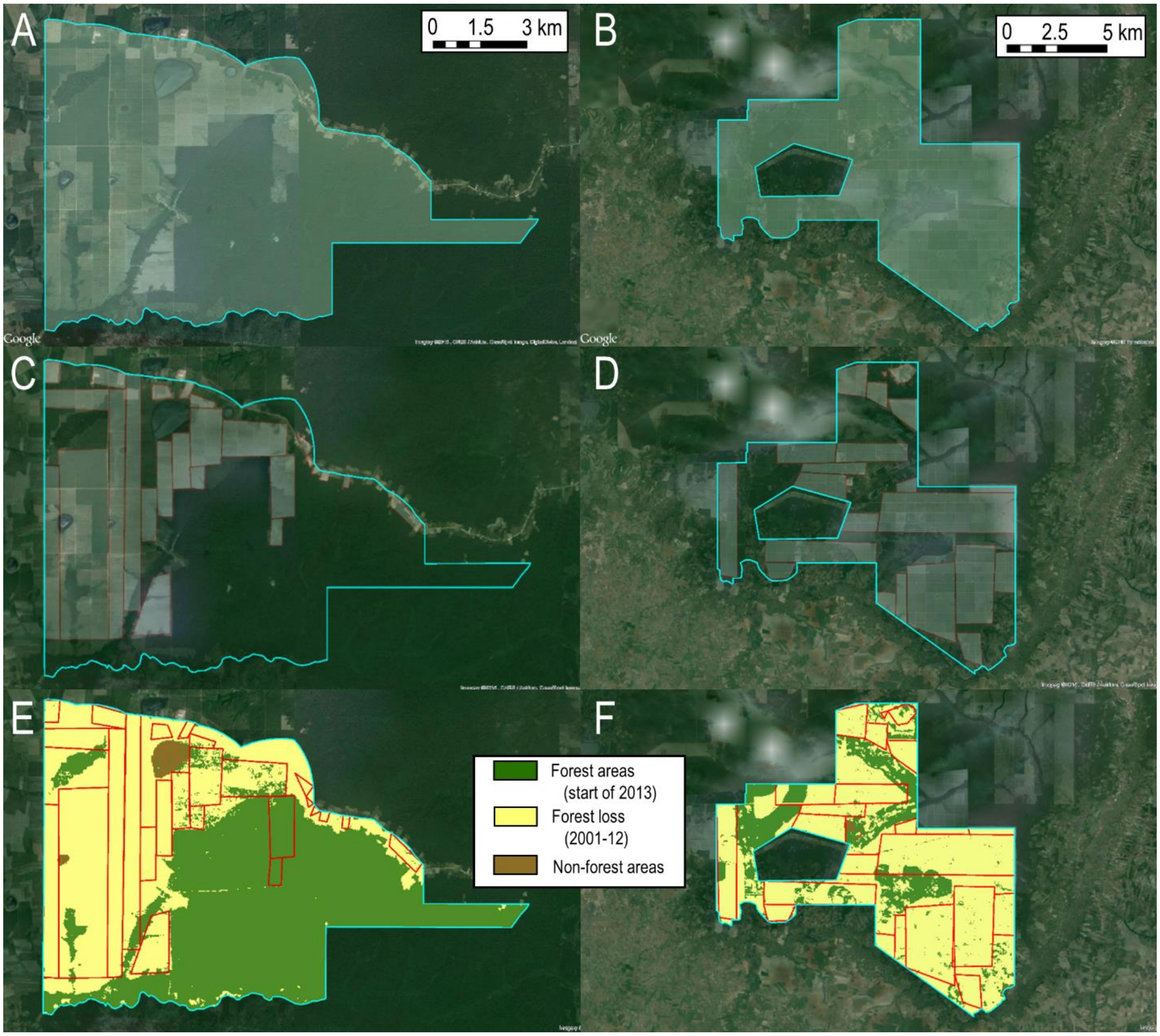
Open Dev. ID[4]	Average tree cover w/ tree plantations (%)	Average tree cover excl. tree plantations (%)	Forested area in 2012 (ha)	Forested area in 2013 tree plantations (ha)	% of 2012 forested area in tree plantations
2	49.0	49.0	7910	0	0.0
5	48.0	48.0	414	0	0.0
9	55.6	55.6	4573	0	0.0
23	41.7	41.7	8389	0	0.0
33	59.8	59.8	4215	0	0.0
38	60.5	60.5	3467	0	0.0
45	87.6	86.8	4733	213	4.5
46	**95.0**	**94.3**	**4299**	**398**	**9.3**
66	**91.7**	**90.1**	**2302**	**449**	**19.5**
75	35.1	33.0	4466	61	1.4
90	45.3	44.9	6352	135	2.1
104	51.2	51.2	7248	0	0.0
107	46.8	46.8	6687	0	0.0
137	72.2	72.2	4501	0	0.0
155	7.6	7.6	276	0	0.0
167	86.4	86.4	752	0	0.0
175	51.7	51.7	337	0	0.0
210	86.8	85.3	4409	1209	27.4
211	88.5	88.1	589	15	2.6
212	44.9	44.9	44	0	0.0
213	90.5	87.9	2895	420	14.5

214	87.2	87.2	971	0	0.0
217	92.7	92.7	9229	0	0.0
221	50.7	50.7	6274	0	0.0
223	47.3	47.3	5474	0	0.0
224	62.5	62.5	7266	0	0.0
281	40.4	40.4	5521	0	0.0
282	43.0	43.0	2548	0	0.0
283	60.3	59.9	5348	124	2.3
Overall	…	…	*121489*	*3024*	*2.5*

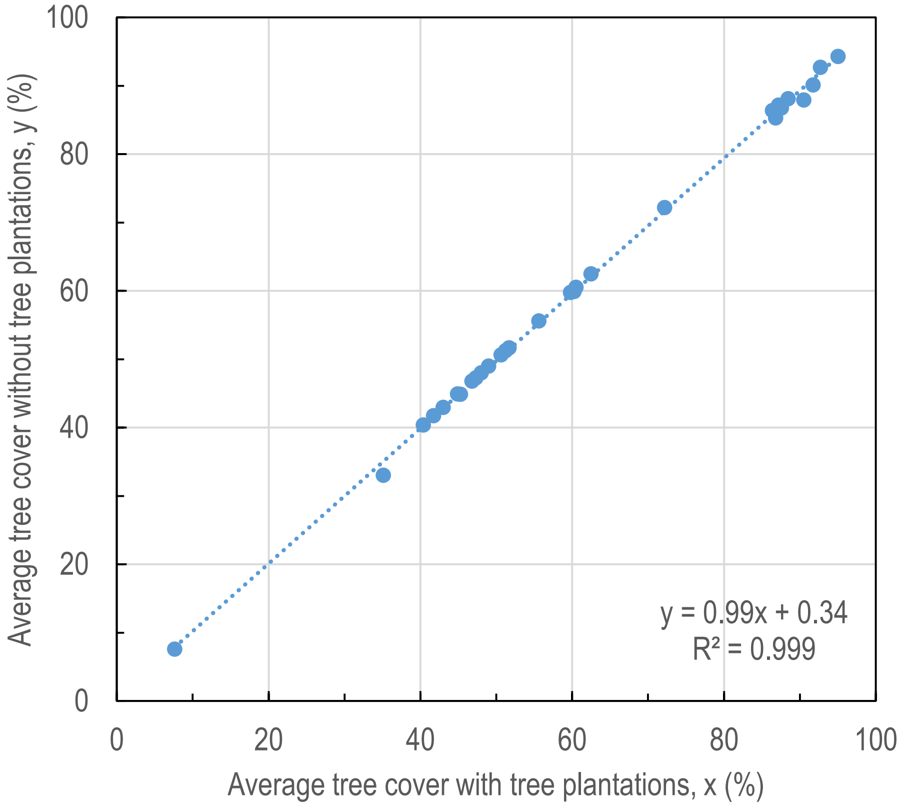
Figures



Supplementary Figure 1. Deforestation in Economic Land Concessions (ELCs) in Cambodia. Bars represent the percentage of total forest loss within an ELC after the contract date. Value inside the base of each bar represents the number of deals with contract dates in that year. Dashed line represents a null hypothesis i.e. the percentage of deforestation would be expected to occur after the contract date if land acquisitions have no effect on deforestation. It is important to note however that deforestation rarely occurs at constant rates. Error bars represent the standard error of the mean.



Supplementary Figure 2. A/B) Representative examples of ELCs (deal #'s 46 and 66, respectively, as reported by Open Development Cambodia[4]) viewed in Google Earth Pro, C/D) Manual digitization of tree plantations, E/F) Areas identified as forested areas, forest loss areas and non-forest-areas according to the Hansen dataset[7].



Supplementary Figure 3. Comparison of average tree cover before and after accounting for tree plantations in 29 random selected ELCs (for details, see Supplementary Table 20).

APPENDIX 5 – MEETING FUTURE CROP DEMAND WITH CURRENT AGRICULTURAL RESOURCES: REQUIRED CHANGES IN DIETARY TRENDS AND PRODUCTION EFFICIENCIES

Supplementary Information

Supplementary tables are provided online through the University of Virginia's LIBRA service.

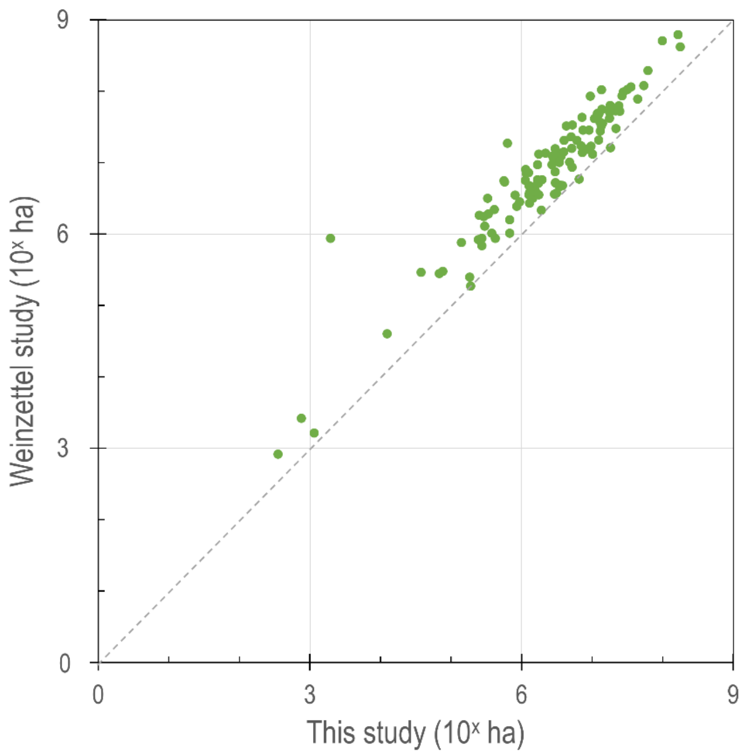


Figure S1. Comparison of estimates of land for crop production. Dashed line represents the 1-to-1 line.