Environmental analyses to inform transitions to sustainable diets in developing countries: case studies for Vietnam and Kenya

Citation:

Heller, Martin C.; Walchale, Abhijeet; Heard, Brent R.; Hoey, Lesli ; Khoury, Colin K.; Haan, Stef de; Burra, Dharani Dhar; Thi, Thanh Duong; Osiemo, Jamleck ; Thi Huong, Trinh & Jones, Andrew D. (2019). Environmental analyses to inform transitions to sustainable diets in developing countries: case studies for Vietnam and Kenya. The International Journal of Life Cycle Assessment, 1-14 p.

Publisher's DOI:
https://doi.org/10.1007/s11367-019-01656-0

Access through CIAT Research Online:
https://hdl.handle.net/10568/101990

Environmental Analyses to Inform Transitions to Sustainable Diets in Developing Countries: case studies for Vietnam and Kenya

Martin C. Heller[1*], Abhijeet Walchale[1], Brent R. Heard[1], Lesli Hoey[2], Colin K. Khoury[3], Stef De Haan[4], Dharani Dhar Burra[4], Thi Thanh Duong[4], Jamleck Osiemo[5], Thi Huong Trinh[4,6], Andrew D. Jones[7]

[1]*Center for Sustainable Systems, University of Michigan, Ann Arbor, MI U.S.A.*
[2]*Urban and Regional Planning Program, University of Michigan, Ann Arbor MI U.S.A*
[3]*International Center for Tropical Agriculture (CIAT), Cali, Colombia*
[4]*International Center for Tropical Agriculture, CIAT-Asia, Hanoi, Vietnam*
[5]*International Center for Tropical Agriculture, CIAT-Africa, Nairobi, Kenya*
[6]*Department of Mathematics and Statistics, Thuongmai University, Hanoi, Vietnam*
[7]*School of Public Health, University of Michigan, Ann Arbor, MI U.S.A.*

**Corresponding author. Tel.: +01-734-474-7166*
E-mail address: mcheller@umich.edu

Abstract

Purpose: Sustainable diets are an environmental, economic and public health imperative, but identifying clear intervention points is challenging. Decision-making will require descriptive analyses from a variety of perspectives, even under the inevitable uncertainty introduced by limited data. This study uses existing data to provide a diet-level perspective on environmental impact from food production in the case study countries of Vietnam and Kenya.

Methods: FAO food supply data at decadal time steps were used as a proxy for national average diets in Vietnam and Kenya. We combined these data with estimates of the greenhouse gas emissions (GHGE) and water use impact associated with producing food commodities. Generic GHGE factors were derived from a survey of the life cycle assessment literature. Country- and commodity-specific blue water use estimates were used, reflecting country-of-origin for import-dominated commodities. The AWARE characterization model was used to offer a diet-associated water scarcity footprint. Trends in diet-associated environmental impacts were interpreted in light of diet shifts, economic development trends, and other factors.

Results and discussion: Increasing per capita food supply in Vietnam, and in particular increases in meat, have led to rising diet-associated per capita GHGE. While supply of beef remains 5.2 times smaller than pork—the dominant meat—increases in beef demand in the past decade have resulted in it becoming second only to rice in contribution to diet GHGE. The water use and water scarcity footprint in Vietnam follow an increasing trend comparable to food supply. On the other hand, historically consistent levels of dairy and beef in Kenya dominate diet-level GHGE. Water use associated with the Kenyan diet shows marked increases between the 1990s and 2000s due to imports of wheat and rice from water-stressed regions. Environmental performance data for characteristic food production systems in these and other developing countries are needed to improve the representativeness and reliability of such assessments.

Conclusions: Despite data limitations, the methods and results presented here may offer a fresh perspective in sustainable development policy deliberations, as they offer an entry point to linking environmental impact and consumption behaviors and can elucidate otherwise obscure or unexpected outcomes. A clear need emerges for further environmental analysis of dominant production systems within both Vietnam and Kenya.

Keywords: *decision-making, sustainable development goals, greenhouse gas emissions, water use, diet, low- and middle-income countries*

1. Introduction

Human diets and the underlying food supply systems that support them are among the principal drivers of both human health and environmental change. There is an urgent environmental, economic, and public health imperative to reshape human diets in order to safeguard human health, mitigate climate change, and sustainably use the planet's natural resources, goals that are integral to the concept of "sustainable diets". Despite a growing number of ecologically-oriented dietary guidelines and initiatives, a major barrier in moving toward sustainable diets is defining clear intervention points that will provide net-positive systemic influence across sectors (Mason and Lang 2017). This challenge is exemplified by the ambiguous objectives and lack of clear policy guidance among the numerous diet-related Sustainable Development Goals (SDGs). The concept of "sustainable diets", for instance, is encompassed across SDG 1 (poverty eradication), SDG 2 (zero hunger), SDG 3 (health promotion), SDG 11 (sustainable cities), SDG 12 (sustainable consumption and food waste reduction), SDG 13 (climate action) and SDG 15 (sustainable forest management) (Blesh et al. 2019). Yet limited attention has been given to the interconnections – and potential tradeoffs – of attempting to address multiple goals that are directly or indirectly tied to food systems (Battersby 2017; Koehler 2015; Pradhan et al. 2017). Further, data informative to decision makers are commonly accessible only in disparate sources, limiting their effectiveness. Moreover, the segregation of data collection, analysis and related decision making within sectoral silos often prevents dissemination and application of information across sectors (Abson et al. 2017; MacRae et al. 2012). Differing disciplinary frames and narratives further introduce disparate interpretations and applications of "sustainable food systems," challenging common goal-setting and evaluation of trade-offs (Béné et al. 2019).

The Entry points to Advance Transitions towards Sustainable diets (EATS) Project aims to address these challenges in a low- and middle-income country (LMIC) context by considering the research question: how can existing data across various disciplines be leveraged to effectively shift multiple axes of food systems to enable sustainable diets? Focusing on two case study countries – Vietnam and Kenya – the EATS project aims to aggregate and analyze existing data and draw on insights from interviews with national and sub-national stakeholders to generate unique information packages aimed at informing decision-making related to sustainable diets. Vietnam and Kenya were chosen as case studies because both countries are on a path of rapid economic development, population expansion, urbanization, and agricultural intensification targeted for agricultural exports (see Supporting Information (SI) File 1, Figures S1 and S2 for relevant trends). All of these characteristics interact and influence diet patterns and food system attributes. However, key sociocultural and agro-ecological differences exist between the two countries, offering valuable contrasts in the trajectories of changes in their food systems. Differences in government structure are a cogent example: Vietnam is a communist, single-party government with a centralized bureaucracy (Vu 2016), whereas Kenya is a multi-party democratic republic in which the 2010 Constitution devolved policymaking power to municipalities (Kanyinga 2016). While framed within the broader interdisciplinary context of the EATS project, this paper focuses on analyses aimed at providing an often-overlooked perspective, particularly in a developing world context, of how consumer food choices drive environmental impact from food production. In this sense, we acknowledge that insights and conclusions drawn from this work must be further considered in relation to social, economic and additional environmental aspects of sustainable diets, but such a complete sustainability assessment is beyond the scope of this paper.

Assessments of the environmental impact of dietary patterns and dietary change via the application of results from individual food commodity life cycle assessments have been reported extensively (e.g., Aleksandrowicz et al. 2016; Blackstone et al. 2018; Hallstrom et al. 2015; Heller et al. 2018; Walker et al. 2018), albeit primarily for diets in developed, high-income countries. Examples of such assessments for low- and middle-income countries (LMIC) and/or developing economies are, to our knowledge, limited to Peru (Vázquez-Rowe et al. 2017), India (Green et al. 2018; Harris et al. 2017), and China (He et al. 2018; Song et al. 2017), as well as a handful of modeling studies with global coverage (e.g., (Chaudhary et al. 2018; Springmann et al. 2018; Tilman and Clark 2014). In all such assessments, researchers must contend with limited data by assigning proxy foods and/or generalizing production methods and regions. This is especially true in the LMIC context, where data on environmental impact of food production, as well as other parts of the food chain, are sparse. In addition, the availability of high-quality dietary recall survey data that may be used to describe average or typical diets in a country are often lacking in LMICs, especially over a consistent time series that allows examination of trends.

In addition to limited data availability, a variety of other challenges arise when considering sustainable diets in a LMIC context. In many developing economies, rapid transformations in food systems and diets have occurred, driven by agricultural developments, rising incomes, and increased urbanization. Often, these transformations have resulted in emerging divides – income divides, rural-urban divides, food access divides – which contribute to multiple burdens of malnutrition, including child stunting, micronutrient deficiencies across the human lifecycle, and overweight/obesity, along with increasing burdens of diet-related non-communicable diseases (Lim et al. 2012). Such developments are observable in both Vietnam and Kenya: average incomes and urbanization have risen steadily over the past two decades (although more dramatically in Vietnam than in Kenya) and childhood stunting and undernourishment have declined significantly (Kimani-Murage et al. 2015; Minh Do et al. 2018). At the same time, however, the prevalence of overweight among adults continues to rise (see SI File 1, Figures S1 and S2).

Addressing these challenges will require a food systems approach, and successful solutions are likely to be very context dependent. To contend with the vast interconnectivity of food systems problems and the likelihood of trade-offs across outcomes including economic growth, social equity, nutrition and health, and environmental performance, decision makers will require access to descriptive analyses from a variety of perspectives, even under the inevitable uncertainty introduced by limited data. In this paper, we combine available approximations of national average diets with estimates of the greenhouse gas emissions and water use impact associated with producing food commodities. The result is a diet-level perspective on environmental impact from food production that may be useful for informing transitions to sustainable diets. We then discuss the challenges and limitations of such an approach, identifying data needs and future research priorities to be pursued.

2. Methods

2.1. Approximating diets

Food Balance Sheet (FBS) "food supply" data from FAOSTAT were used as proxy for national average diets in Vietnam and Kenya, with 68 and 80 defined food commodities, respectively (FAO 2018a). To smooth inter-annual variability, data from consecutive years were averaged to provide decadal time steps. Five-year averages were centered around decadal time steps from 1971 to 2011 (all years of food supply data and resulting averages are provided in SI File 2). Water use impacts were limited to 1991, 2001 and 2011 due to limited historic data on import countries of origin (see section 2.3). FBS food commodities were matched with those available in environmental impact datasets, described below. Annual per capita food availability (in kg per capita per year) of each food commodity were multiplied by environmental impact factors per kg of food commodity. Impacts were summed across food groups and across the complete diet.

2.2. Greenhouse gas emissions of food production

Greenhouse gas emission (GHGE) factors associated with production of food commodities were collected and aggregated from life cycle assessment (LCA) literature as described in Heller et al. (2018). These aggregated data consisted of 1645 entries from 321 unique sources written in English and published in 2005-2016; all studies applied LCA methods to one or more food products and reported mid-point impact assessment of GHGE. Given the range of years, specific impact assessment methods varied, introducing a source of untraceable uncertainty, but all global warming potential factors were based on a 100-year time horizon, and the majority (more than 65%) utilized factors from Intergovernmental Panel on Climate Change (IPCC) 4^{th} Assessment Report (IPCC 2007). GHGE factors in kg CO_2eq. were adjusted to a functional unit of 'kg of food' with meat and fish/seafood adjusted to 'kg of edible boneless weight' as described in Heller et al. (2018). Boundary conditions were maintained at the farm gate (or, in the case of processed commodities such as flours and oils, at the processor gate), and therefore results represent emissions from production and do not include downstream stages of secondary processing, packaging, distribution, etc. The entries compiled in this dataset comprise case studies from diverse geographies, predominantly Europe and North America, and different production practices; with the exception of beef from dairy herds and heated greenhouse vegetable production (which were excluded from averages), arithmetic average GHGE factors were calculated using all entries for a given food commodity. As such, these emission factors do not explicitly represent the production practices in Vietnam and Kenya, but in the absence of country-specific data, they serve as a proxy to provide coarse guidance on diet-level impacts. Emission factors are assumed constant with time and therefore do not reflect changes in production likely to occur over the time series considered. Detailed descriptions of choices made in linking FBS commodities with GHGE data are provided in SI File 1.

Variability estimates based on the emission factors aggregated from the LCA literature were used to give an indication of uncertainty in the results. A 95% confidence interval around the mean value was calculated for each food commodity as described in Heller et al. (2018); if there were too few observations to calculate a standard deviation, related foods or groups of foods were used to establish a confidence interval. Lower and upper bounds of the confidence intervals were used in diet-level calculations to consider the influence of emission factor variability on diet-level results. It is important to recognize that the emission factor variability used here reflects data reported in the LCA literature, which is strongly biased towards high-income, industrialized contexts, and may not fully capture the variability possible in LMIC contexts.

2.3. *Water use impact*

The impact of fresh water consumption on other users (both human and ecosystems) is dependent on the relative availability of water in a given region. Based on the lifecycle-based approach framed in the ISO 14046 standard (ISO 2014), a consensus-based method for assessing water consumption impact, called AWARE (Available WAter REmaining), has recently been presented (Boulay et al. 2018). We use the AWARE method here to quantify a water scarcity footprint associated with the food production in national average diets. The AWARE method assesses the potential for water deprivation among humans and ecosystems by considering the difference between availability and demand in a given region (Boulay et al. 2018).

In this assessment, only "blue water" use – surface and ground water removed from the watershed through crop irrigation or consumed directly by livestock – is considered. Country average blue water use (m^3 per kg) for each of the FBS food commodities were taken from Mekonnen and Hoekstra (2010a, 2010b) and assumed constant across the time series considered. Beverages (with the exception of milk), and fish and seafood were excluded from the water use assessment due to incomplete data. Assessing blue water use for fish and seafood is methodologically challenging as it likely should only include water used in producing feed for aquaculture. Disaggregating aquaculture from wild caught fish and seafood and identifying appropriate aquaculture feed rations is beyond the scope of this paper. Annual, country average AWARE100 impact factors appropriate for agricultural production (that is, aggregated to better reflect the temporal and geographical patterns of consumption for agricultural activities) came from WULCA (2016). In this paper, we report both uncharacterized blue water use (without the water scarcity characterization factor applied), and water scarcity footprints (with AWARE factor applied).

The influence of "country of origin" water use for imported food commodities was considered for commodities where the import share of domestic availability (import plus production) was greater than 30% *and* per capita food supply was greater than 0.75 kg per capita per year. For Vietnam, this criteria was met by 'wheat and products' and 'milk' in 1991 and 2001 and by 'apples and products', 'maize and products', 'soybean oil', 'soybeans', 'wheat and products', 'milk', 'poultry meat', and 'bovine meat' in 2011. In Kenya, the criteria was met by 'palm oil', 'rice', 'sugar', and 'wheat and products' in 1991 and 2001, and by 'palm oil', 'peas', 'rice', 'sorghum and products', 'sugar', 'tea', and 'wheat and products' in 2011. For these commodities, the FAO Detailed Trade Matrix was used to identify country of origin and relative quantity of imports (averaged over 5 years) into Vietnam and Kenya. Country-of-origin average blue water use was again derived from Mekonnen and Hoekstra as above, and AWARE factors for each country-of-origin were used. An average blue water use as well as water scarcity footprint for imports of each commodity, weighted based on relative quantity of imports from each country, was calculated. Values used in final diet-level assessments were then derived by weighting domestic water use (and water scarcity footprint) and imported water use (and water scarcity footprint) by the share of domestic availability (import plus domestic production). Further detail of this water use impact method is provided in SI File 1.

3. Results

3.1. Diet-level environmental impacts for Vietnam

Figure 1a shows the changes in food supply in Vietnam – used here as a proxy for national average diets – over the five decades from 1971 – 2011. On a mass basis, cereals and grains (primarily rice) and vegetables dominate the Vietnamese food supply. Per capita supply of fruits, vegetables, seafood and sweeteners doubled between 1971 and 2011, quantities of dairy increased 3.5-fold, and legumes and meat increased more than 5-fold over the same time period. The growing trend in meat supply is amplified in terms of GHGE from food production. In Figure1b, meat contributes 26% of total diet GHGE in 1971 and 44% in 2011.

Figure 2 shows the trends in Vietnamese meat supply from 1971-2011. Pig meat clearly dominates and has continued to outpace the growth in supply of other meats since the 1970s, with pork supply in 2011 422% higher than beef and 178% higher than poultry. More recent increases in poultry and beef are notable. While still 5.2 times smaller than the pig meat supply, the 180% increase in beef supply from 2001 to 2011 has led to beef becoming the greatest contributor to GHGE from meat in the Vietnamese diet based on this estimate, and second only to rice in the overall diet. This emphasizes the effect that otherwise unremarkable changes in diet can have on food system environmental impacts such as GHGE.

Figure 3 gives the annual water use and water scarcity footprint per capita associated with the Vietnamese diet in the 1990s, 2000s and 2010s (see Figure 1a for corresponding annual food supply). Rice dominates water use at 60%, 49% and 33% in 1991, 2001, and 2011, respectively, with the decrease roughly proportional to the decreasing weight proportion of rice in the diet. Pig meat represents 11%, 13% and 19%, whereas all vegetables combined contribute 16%, 17% and 21% of water use in 1991, 2001, and 2011 respectively. Despite having a lower blue water use intensity, poultry contributes more to diet-level water use than beef in 2001 and 2011 due to a larger quantity in the food supply.

3.2. Diet-level environmental impacts for Kenya

Kenya offers an interesting contrast to the trends seen in Vietnam. The supply of dairy foods shows a general increasing trend in Kenya from 1971 to 2011; Bovine meat (beef) dominates total meat supply (60% of meat supply in 2011), with both bovine and total meat supply remaining relatively constant (coefficient of variation of 10% across *all* years in the 1969-2013 range) (Figure 4a). Maize is the predominant cereal/grain consumed in Kenya, but decreases in maize and increases in both wheat and rice are observable beginning in the 1990s (see SI File 1, Figure S5a). Vegetables show an approximately 30% increase through the 2000s, driven mostly by increases in potato supply. Beef is consistently the largest contributor to GHGE in the Kenyan diet, followed by other meats (predominantly mutton and goat and edible offal) and dairy. Contributions to GHGE in 2011 were 40%, 16% and 15% for beef, other meats, and dairy, respectively.

The water use and water scarcity footprint per capita associated with the Kenyan diet is shown in Figure 5 (see Figure 4a for corresponding annual food supply). Here, we see a disproportional increase in water use from cereals and grains between 1991 and 2001 and again in 2011, which can be traced to changes in imports of rice and wheat. While rice and wheat are secondary grains in the Kenyan diet, they dominate the water use in this group (see SI File 1, Figure S5). Between the 1990s and 2010s, wheat in the Kenyan diet increased by a factor of 1.8 and rice by 5.5; much of this increase was met by imports, and progressively from countries with high blue water demands for producing these crops as well as high water deprivation potential (see Table S2 in SI File 1 and further discussion in Section 4.1).

In contrast to the situation in Vietnam, results for Kenya show distinct differences in distribution between water use and water scarcity footprint, particularly for cereals and grains (see SI File 1, Figure S4 for the relative percentage version of Figure 5). Here, such a difference is indicative of large influences from imports, and can be traced almost entirely to increasing imports of wheat and rice from Pakistan. The AWARE characterization factor, a measure of water stress, for Kenya is 9.7, whereas Pakistan is 62.2 (note that the AWARE scale is normalized such that world average = 1).

3.3. Uncertainty analysis

Uncertainties due to lack of specific data are introduced into this assessment at a number of levels, and should be considered high. A number of these sources, including the appropriateness of FBS data as a representation of diet and the specificity of the GHGE factors used, are addressed as limitations and discussed in Section 4.4. Inter-annual variability in food supply data has been addressed by averaging five consecutive years to represent each decadal time point. The variation in these averages are shown in SI File 2; high coefficients of variation generally appear with food commodities characterized by small per capita supply and therefore do not influence environmental impact totals in a notable way. The variability in GHGE factors aggregated from the literature also introduces a range of uncertainty in the results. The influence of this variability on the diet-level GHGE across food groups is shown for each country in SI File 1 (Figures S7, S8 and Table S2). The range in emission factors introduces a range of ±17-19% to total diet GHGE for Kenya with no notable change in food group distribution. For Vietnam, the introduced range is ±28-41%. The higher variability for Vietnam is driven by a large confidence interval (54% of mean) for rice, and decreases as the quantity of rice in the diet decreases. The variability in emission factors shown here does offer a measure of uncertainty, but may not capture the range of variability possible in production systems in Vietnam or Kenya. As has been noted by Mekonnen and Hoekstra (2011, 2012), gaps in input data for the models used to estimate blue water use introduce high but unquantifiable uncertainties; these authors detail the sources of uncertainties.

4. Discussion

4.1. *Understanding observed trends*

The trends seen in the Vietnamese diet can be partly explained by economic development. Vietnam has experienced significant increases in both gross domestic product (GDP) and incomes per capita since the early 1990s, with GDP per capita quadrupling between 1990 and 2017[1] (The World Bank 2018; USDA Foreign Agricultural Service 2017). Increases in meat consumption are often seen as part of the 'nutrition transition', driven by increasing affluence and structural transformations in national and regional food systems (Popkin et al. 2012). However, more recent analyses suggest a confluence of factors contributing to the rise in meat consumption in Vietnam: increased availability (much through imports) and affordability of meat, an intensification of meat in traditional meals, increased frequency of eating outside the home, and the social significance of meat as a symbol of development and social status (Hansen 2018). Better understandings of the economic, social and cultural factors behind observed changes in diet can help direct policy responses. Regardless of the drivers, however, increases in meat consumption have a clear influence on the environmental impact attributable to diet, and this perspective offers a valuable balance to the positive effects that proportional increases in meat can have on nutrition. Based on the data used here, the GHGE intensity of beef is 6 times that of pork and 9.8 times that of poultry; therefore, the increasing demand for beef in Vietnam has a particularly strong effect on diet-related GHGE. All of these trends must be interpreted within the context of growing divides within the population: while national average undernourishment in Vietnam has fallen to 10%, childhood stunting still remains at 25% (FAO 2018a), and both measures are more severe in sub-populations such as ethnic minorities (Benny, Duc and Hang 2017).

The water use and water scarcity footprint in Vietnam follow a comparable increasing trend to food supply from the 1990s to 2010s. As with GHGE, rice remains a dominant contributor to water use impacts, as a significant portion of rice production in Vietnam is under irrigated conditions. Meat is also an important contributor to water use, although due to both the greater supply quantity and a higher blue water use intensity (according to the estimates used here), pork maintains the largest share of meat-related water use. Vegetables emerge as an important contributor to diet-level water use, although the FBS offer poor resolution of vegetable types, requiring proxies based on averages of not-otherwise-specified vegetables. Significant imports of wheat, dairy, beef and poultry influence the water footprint of the Vietnamese diet, and shifts in country-of-origin play a role, but such shifts are overshadowed by the dominance of rice and pork, and

[1] GDP per capita based on purchasing power parity (PPP). PPP GDP is gross domestic product converted to international dollars using purchasing power parity rates. An international dollar has the same purchasing power over GDP as the U.S. dollar has in the United States. Data are in constant 2011 international dollars.

therefore are not overtly apparent in the diet-level assessment.

Kenya's economic growth in recent decades, in comparison to Vietnam, has been more modest (see SI File 1, Figure S2), and despite a growing urban middle class, roughly 25% of the population remains undernourished (FAO 2018a). Pastoralism has a long history in Kenya and cattle milk and meat have remained a cornerstone of the diet, along with staples such as maize, potatoes and beans. Despite the dominance of meat in the carbon footprint of the Kenyan diet, average meat supply in Kenya (16.0 kg per capita per year) remains quite low relative to global averages (45.5 kg per capita) and supply in developed countries (U.S.=115.6 kg, Europe=80.8 kg) (FAO 2018a). In comparison, Vietnam's meat supply in 2013 was 62 kg per capita per year, which is high relative to other Southeast Asian countries at similar levels of GDP per capita (Hansen 2018).

The water use analysis with the Kenyan diet offers an interesting portrayal. Whereas the impacts of greenhouse gas emissions are felt globally in the form of climate change, the impacts of water use are primarily local to where the food is produced, and therefore invite a spatial component in assessment. Based on data from FAO, domestic production in Kenya in the early 1990s accounted for 50% and 41% of the wheat and rice supply, respectively; by 2011, this had decreased to 26% and 15%. Imported wheat in the 1990s came from North and South America and Western Europe whereas in 2011 imports from Eastern Europe and Pakistan become prominent (see SI File 1, Table S3). The jump in water use due to wheat (SI File 1, Figure S5b) can be predominantly attributed to imports from Pakistan, where blue water use intensity (m^3/kg) is high; the relatively high AWARE characterization factor for Pakistan, indicative of a water-stressed region, further amplifies this contribution in the water scarcity footprint. Note that, based on the dataset from Mekonnen and Hoekstra (2010a), wheat production in Kenya is not irrigated, and therefore does not contribute to this blue water assessment. A similar trend exists for rice with imports from a diversity of regions in the 1990s concentrating to Asia, and specifically Pakistan, through the 2000s and 2010s. For our 2011 data point, imports from Pakistan are responsible for 97% of Kenya's water scarcity footprint for both wheat and rice (SI File 1, Table S3). While it is unclear what policy incentives (if any) might arise from such an assessment, it is nonetheless a descriptive portrayal of an international food system where the environmental impacts of production may be experienced quite distantly from the consuming population.

4.2. *Comparisons between countries*

Vietnam and Kenya have experienced different development trajectories over the past three or four decades; comparisons of how these differences present in diet and associated environmental impacts may offer insights into paths toward sustainable diets. Such a comparison should not be interpreted as imposing a ranking between countries, however. As mentioned above, Vietnam has experienced a steep rise in GDP per capita since the 1990s, whereas Kenya's growth has been more modest (SI File 1, Figures S1b and S2b). Shifts in overall food supply in each country, while certainly not a sole result of this growth, appear fairly well correlated with it. Changes in environmental impact associated with the

Vietnamese diet are driven primarily by overall increases in per capita consumption and shifts to more meat in the diet; increased demand for beef in particular stands out as an important driver of GHGE. On the other hand, a consistent presence of meat and milk from cattle in the Kenyan diet dominate the carbon footprint, and it is only in the most recent decade that the per capita carbon footprint of the Vietnamese diet meets and exceeds that of Kenya. For perspective, however, the diet related carbon footprint for Vietnam and Kenya based on the 2011 time point are still about 32% and 49% lower, respectively, than estimates for the United States over approximately the same time period (Heller et al. 2018).

Blue water use associated with diet in Vietnam (64 m^3 $cap.^{-1}$ $yr.^{-1}$) and Kenya (42 m^3 $cap.^{-1}$ $yr.^{-1}$) are also lower than values reported for the global average (139 m^3 $cap.^{-1}$ $yr.^{-1}$), France (97 m^3 $cap.^{-1}$ $yr.^{-1}$)(Vanham et al. 2018) and the U.S. (~170 m^3 $cap.^{-1}$ $yr.^{-1}$)(Tom et al. 2016). Such comparisons reflect not only diet composition of a country, but also climate and prevalence of irrigation. An estimated 49% of agricultural land is serviced by an irrigation scheme in Vietnam (Nguyen et al. 2017) whereas less than 1% of arable land receives irrigation in Kenya (World Bank: CIAT 2015). To our knowledge, no comparable diet-associated water scarcity footprints based on the AWARE characterization method have been published.

Multiplying per capita GHGE by the population offers another interesting trend (Figure 6): total diet-associated GHGE very closely tracks population growth in Kenya from the 1970s to 2010s, whereas total GHGE in Vietnam increases much more rapidly than the population. This further emphasizes the effect that compositional changes in diet have on diet-associated emissions in Vietnam. Comparisons between the diet-associated emissions for the total population shown in Figure 6 and agricultural sector emissions reported in country GHGE inventories should be made with caution; diet-associated estimates include imported food and exclude exported agricultural production, whereas country-level inventories should be the opposite. Reported agricultural sector emissions in 2011 for Vietnam and Kenya were 59.7 Mt CO_2eq. and 38.8 Mt CO_2eq., respectively, and total country emissions (excluding land use change and forestry) were 247.3 Mt CO_2eq. and 60.3 Mt CO_2eq., respectively (WRI 2017). While agricultural emissions for Kenya correspond reasonably well with the 2011 value in Figure 6, discrepancies for Vietnam are greater than can be explained by adjustments for imports and exports, and need to be better understood. Parallel figures comparing population growth and diet associated water use impacts are offered in SI File 1 (Figures S8 and S9) and demonstrate similar differences: the rate of increase in diet-related impacts in Vietnam outpaces population growth.

4.3. *Key production systems in Vietnam and Kenya*

An appreciation of the dominant production practices for key commodities in each country may provide additional insight into interpretation and offer a path forward for characterizing and inventorying these production systems in future research. Evaluation of the environmental impact of these systems is complicated both by a lack of data but also by the difficulty of accounting for multi-functionality, especially for livestock. For example, cattle in LMIC contexts provide products and services beyond the production of milk and meat, including manure production for fertilizer or fuel, draught power, economic insurance, social status or cultural identity. Quantitatively accounting for these other functions, and allocating a portion of the environmental burden to them, is challenging (Weiler et al. 2014).

With a national coverage of 7.7 million ha, rice cultivation is recognized as the largest agricultural contributor in Vietnamese national GHGE profiles (USAID 2016). Rice is also the largest contributor to GHGE in the Vietnamese diet in our assessment. Much of the GHGE intensity of rice production is due to methane emissions under flooded conditions. The majority of rice production in Vietnam occurs in three distinct ecosystems: approximately 60% of production occurs under irrigated and intensive production in the Red River Delta and the Mekong River Delta; roughly 30% occurs in rain fed lowlands and flood-prone areas; the remaining production is primarily in predominantly rain fed upland ecosystems in the northern highlands (Bong 2000). In addition to differences in agro-ecology, there are also seasonal cycles with different management regimes for paddy rice: summer-autumn, autumn-winter, winter and winter-spring plantings. The summer-autumn and winter-spring plantings cover the largest areas. Whereas efforts are underway to understand the landscape-level variability in GHGE from rice production (Khai NH 2018; Torbick et al. 2017), this work does not readily translate into emission intensities per kg of rice produced. There are known methods to reduce methane emissions and therefore GHGE associated with rice production, including alternate wetting and drying (AWD) and midseason drainage techniques, both of which are key components of Vietnam's Nationally Determined Contribution plan in agriculture (Trung et al. 2017). Compared with continuous flooding, AWD in irrigated rice can reduce water use by up to 30% and reduce methane emissions by 48% without reducing yield. Adoption in the An Giang Province of the Mekong Delta has reached upwards of 50% of paddy area, yet irrigation conditions and water reliability limit implementation (Yamaguchi et al. 2019). Updated emission factors based on surveyed production practices would improve the current analysis.

The national herd of 5.8 million head of cattle in Vietnam has not been able to provide for the growing demand for beef and milk products in the country. Traditional smallholder production systems relying on local cattle breeds dominate (70-80% of cattle) (Asia Beef Network 2018). These systems are characterized by extensive cow-calf grazing with very little concentrated feed inputs. Slow growth rates, small animal size and low carcass percentage result in low beef productivity from native cattle; in turn, this likely means higher GHGE intensity. On the other hand, imports of beef and cattle into Vietnam from India, Australia and the U.S. have also increased to meet demand, now representing 30-40% of total beef consumption in Vietnam (Asia Beef Network 2018).

Characterization of the cattle (beef and dairy) production systems in Kenya is complex due to their diversity as well as the fact that producing a marketable product may not be the sole or primary reason for owning cattle, as is likely also true among smallholders in Vietnam (Herrero et al. 2013). A recent multi-stakeholder approach to such characterization identified semi-intensive (45% of farms) and small-scale intensive (35% of farms) as the predominant dairy production systems, with beef cattle production systems composed primarily of semi-intensive (54%) and extensive pastoralism (34%) (FAO 2018c). Semi-intensive dairies involve a combination of grazing and supplemental feed, typically small herd sizes (1-20 animals) and relatively low productivity, with much of the produced milk consumed at home. Small-scale intensive dairies are operations with 1-20 cows, typically exotic high-grade dairy breeds that use high quality feed and no grazing to maximize production, with milk primarily produced for market. Semi-intensive beef production systems are agro-pastoral, where animals are kept in complement with crop production (residues fed to animals, manure and drought power aid crop production) but animals are raised to be sold. Average herd sizes are between 10 and 12 cattle, with animals predominantly grazing on communal areas and supplemented with crop residues. Extensive pastoralism of beef cattle is a low input/low output system entirely reliant on transhumance grazing primarily in arid and semi-arid regions of the country and dominated by domestic cattle breeds. Despite low productivity, the meat produced is sold to consumers in urban markets, satisfying the bulk of domestic demand. These characteristics of Kenyan cattle production systems are expected to influence emission factors associated with the meat and milk produced, yet limited data is available to inform such an analysis. Recent efforts to consider the potential emission reduction from the Kenyan dairy sector reported a current baseline emission intensity of 2.4 kg CO_2eq. per kg milk (compared to 1.3 kg CO_2eq. / kg used in this study), with reduction scenarios involving improved forage quality and concentrate supplementation of 1.7-1.8 kg CO_2eq. / kg (Brandt et al. 2018). Interestingly, attempts to account for the multiple functions of cattle in carbon footprint estimates of smallholder dairying in Kenya have resulted in emission intensities for milk that are very much in line with the estimate used in this study (Udo et al. 2016; Weiler et al. 2014).

An additional methodological consideration that can affect emission intensities of both milk and beef is the chosen allocation between milk and meat production in LCAs of dairy production systems. Typically, meat from dairy systems has a lower carbon footprint than meat from dedicated beef systems because the production burdens are shared between milk and meat co-products. The emission factor for beef used in this analysis (33 kg CO_2eq. / kg boneless edible beef) was derived from studies reporting on dedicated beef systems; emission factors from studies identified in our literature review reporting beef from dairy systems averaged 19 kg CO_2 eq. / kg boneless edible beef. Bovine meat in Kenya is produced both through dairy production systems as well as dedicated beef cattle production systems but statistics on quantities from each system are poor. By one estimate, dairy system culls and fattened bulls represent 23% of the beef cattle population and 15% of the beef production value (derived from FAO (2018b), Table 6). Applying the "beef from dairy" emission factor to 15% of the beef supply in Kenya in 2011 would reduce the total diet-level emissions by 2.5%.

4.4. *Limitations and opportunities*

Food balance sheets, which represent top-down estimates of the availability of food for human consumption in a country, are a crude approximation of national diets. Inaccuracies in FAO Food Balance Sheet data in representing dietary intake are well known, thus presenting challenges for assessing nutritional adequacy (Del Gobbo et al. 2015). One aspect of this challenge is that the Food Balance Sheet data do not account for food waste in the household, which is problematic when attempting to assess nutrients *consumed* by the population. On the other hand, environmental impact assessments should include lost or wasted food as it carries the same production impacts as consumed food. Other challenges, such as not accounting for home production or food reaching households from non-retail markets, introduce error. Still, in cases where detailed diet intake data is unavailable or difficult to access, these proxies may provide sufficient information to guide national food system sustainability policy.

The GHGE assessment presented here relies on emission factors generalized from available LCA literature, which largely reflect crop and animal production methods in the industrialized north. LCA data from Southeast Asia and Africa remain sparse, and both sub-national and regionally appropriate data on the environmental impact of food production are a research priority. Others have gathered limited assessments into region-specific emission factors; using the regional emission factors (South & Southeast Asia for Vietnam and sub-Saharan Africa for Kenya) from one such global effort by Porter et al. (2016) showed 14-22% *increases* in the absolute value of per capita GHGE over our estimates but no observable changes in the trends discussed here. Regional/local LCA data should better reflect production practices, resource use, and yields, and may shift the relative importance of crops in these diet-level carbon footprint assessments. Similarly, emission factors for animal-based foods are strongly dependent on aspects such as animal feed rations, harvest yields, herd management and manure management practices.

Methane emissions from agriculture play an important role in the GHGE associated with diet in both Vietnam and Kenya. Data resolution in the approach taken here does not permit disaggregating contributions from individual greenhouse gases, but cattle and rice production are known to be dominant contributors to agricultural methane emissions. Further, 48% of Kenya's and 70% of Vietnam's reported agricultural GHGE in 2011 was from methane (UN FAO 2018). This is important to acknowledge because methane is a short-lived climate pollutant and its global warming potential relative to CO_2 is highly dependent on the defined time horizon of the assessment method (Fesenfeld et al. 2018). The dominant standard time horizon for comparing climate pollutants, and the basis for analysis and data presented here, is 100 years; over 100 years, the average radiative forcing of methane is 28-36 (depending on whether various indirect climate effects are included) times that of CO_2, but over a 20 year time horizon it is 84-87 times that of CO_2 (IPCC 2013). Some climate scientists and policy analysts call for greater emphasis on mitigation of short-lived climate pollutants in order to avoid self-accelerating climate change in the short term due to climate tipping points and amplifying feedbacks (Fesenfeld et al. 2018). Under such a scenario, contributions from rice, beef and dairy would become even more dominant.

The data on blue water use utilized here are based on the grid-based evapotranspiration, water balance and yield modeling of Mekonnen and Hoekstra (2010a,; 2010b). While providing a sound basis for global water use assessments, these modeling efforts require large sets of assumptions and may not always reflect local economic and technological realities of irrigation. In addition, there appear to be undocumented gaps in the blue water use dataset utilized here, which we interpret as a lack of irrigated production in the given country. Still, the overall trends seen here fit with expectations. The diet associated blue water footprint in Vietnam, where about half of the agricultural land is irrigated, is dominated by domestic production of water-demanding foods including rice, meats and vegetables. On the other hand, the blue water footprint in Kenya is heavily influenced by imported crops, as there is very little domestic irrigation.

In this exercise, GHGE factors and water use values are assumed to be constant over the time series presented. This is clearly a limiting assumption as changes in agricultural production since the 1970s have surely influenced emissions and resource use per unit of food produced. This simplification is justified, however, because the goal here is not to accurately characterize diets in Vietnam or Kenya per se, but instead to offer additional perspective in deliberating a transition to sustainable diets. Variability in emission factors are expected across global production systems (see, e.g., Poore and Nemecek (2018)), but differences between food types (e.g., beef and pork) stand out amongst this variability, providing a basis for deliberation on diet composition. Such bottom-up, consumption-oriented approaches to estimating the environmental impact of food production offer a valuable complement to top-down, production-oriented perspectives typical in country-level GHGE accounting.

The preliminary work presented here highlights a number of opportunities to improve data and scenario analysis in a LMIC context. First, improvements are needed in the representativeness of environmental impact factors for agricultural production systems. For example, it is well known that rice, vegetable and meat production practices in Vietnam are diverse in input and management intensity, with expected differences in GHGE and water footprints. Such data development serves not only diet-level assessments such as this one but also can aid in prioritizing interventions in production practices. However, because international climate action deliberations are characterized and quantified at national boundaries, resources may be prioritized to landscape-level estimates rather than building an understanding of production systems and environmental efficiencies (i.e., impacts per unit of output). While agricultural production typically dominates the life cycle for environmental impacts evaluated here, consideration of other parts of the value chain, including processing, transportation, refrigeration, retailing/marketing, and household consumption practices would offer additional insights into the influence of development trajectories. Second, alternative approaches to defining diets, including household living standard and/or nutrition survey instruments, would offer a different, and perhaps more accurate, perspective on food intake, and may allow differentiation across socio-economic (e.g., wealthy vs. poor) and geographic (e.g., rural vs. urban) strata. Such scenarios may aid in describing the drivers of observable transitions, thus offering further insight into potentially effective intervention points. Third, integration of socio-economic dimensions of sustainability could offer further insight into how changes in production practices and/or consumption behaviors influence livelihoods (jobs, disposable income, nutrition, health).

The demand-oriented perspective presented here – assigning emissions and water use based on the diet of a country – differs from, and should complement, typical national accounting inventories such as those for the United Nations Framework Convention on Climate Change (UNFCCC) that are oriented to national production activities. A recent review by Vázquez-Rowe et al. (2019) highlights the role that life-cycle methods have in complementing climate policy. One of the advantages of a life-cycle perspective is that it typically spans temporal and geographical boundaries that may be more rigid in national accounting. Such complementary perspectives may be useful in balancing policy goals including Sustainable Development Goals and UNFCCC targets. Further, a consumption-oriented perspective can aid in recognizing and addressing synergies and trade-offs across SDG goals. For example, SDG 12 (responsible consumption and production) has been identified as being linked to problematic trade-offs with other goals (Pradhan et al. 2017). Progress in goals associated with higher human development and socioeconomic standards have historically been in conflict with environmental protection goals; this can be seen in the patterns for Vietnam presented in this paper. Acknowledging the interactivity of SDGs may allow the emergence of policies that create new synergies among goals; consideration of dietary patterns offers an additional lens for such deliberations on the food system.

5. Conclusions

In this paper, we consider the greenhouse gas emissions and water use impacts associated with per capita food supply in Vietnam and Kenya within the constraints of non-ideal data availability common for low- and middle-income countries. Despite data limitations, the methods and results presented here may offer a fresh perspective in sustainable development policy deliberations, as they elucidate otherwise obscure or unexpected outcomes. Specifically, we demonstrate diet transitions and increased intake of meat in Vietnam leading to significant increases in diet-associated greenhouse gas emissions. Much of this increase is due to shifts in the type of meat: relatively low levels of beef have strong influence on the diet-associated carbon footprint. In Kenya, a historic preference for beef and milk dominates the GHGE associated with the diet, but relatively minor shifts in grains and increased dependence on imports from water-stressed regions greatly influence the water use impact associated with diet.

Inclusion of such consumption-oriented perspectives is vital to addressing the challenges of sustainable development. When it comes to diet and food systems, national priorities are likely to migrate toward critical considerations such as livelihoods, nutrition and equitable access, but maintaining an environmental perspective may assist in identifying cross-sectoral synergies and avoiding unintended setbacks. A consumption-oriented perspective also complements national emission accountings that consider contributions by sector, including agriculture: neither tell a complete story, but decision making will be more informed with both perspectives. The national-level time series inquiry presented here is informative as it demonstrates clear trends, but the same methodological approach could be used to compare current diets with projected or recommended patterns or to consider policy impact scenarios. Of course, any such analysis must be tempered by the quality and representativeness of the underlying data. We highlight a strong need for data on the environmental performance of divergent production and supply systems in LMICs, preferably based in and informed by LCA methods. Diet-level assessments would also benefit from application of alternative tools for defining per capita food intakes such as household living standard surveys or national nutrition surveys based on 24 hour recall. In general, investments in data across domains of food systems are strongly needed to monitor progress and inform decision-making within these critical yet complex systems.

6. Acknowledgement

This work is funded through a grant from the Graham Sustainability Institute at the University of Michigan.

7. References

Abson DJ et al. (2017) Leverage points for sustainability transformation. Ambio 46:30-39

Aleksandrowicz L, Green R, Joy EJ, Smith P, Haines A (2016) The impacts of dietary change on greenhouse gas emissions, land use, water use, and health: a systematic review. PLOS ONE 11:e0165797

Asia Beef Network (2018) Viet Nam Country Profile: Cattle Production Systems. http://www.asiabeefnetwork.com/country-profiles/viet-nam/cattle-production-systems/. Accessed January 2019

Battersby J (2017) MDGs to SDGs – new goals, same gaps: the continued absence of urban food security in the post-2015 global development agenda. Afr Geogr Rev 36:115-129 doi:10.1080/19376812.2016.1208769

Béné C et al. (2019) When food systems meet sustainability–Current narratives and implications for actions. World Dev 113:116-130

Blackstone NT, El-Abbadi NH, McCabe MS, Griffin TS, Nelson ME (2018) Linking sustainability to the healthy eating patterns of the Dietary Guidelines for Americans: a modelling study. Lancet Planet Health 2:e344-e352

Blesh J, Hoey L, Jones AD, Friedmann H, Perfecto I (2019) Development pathways toward "zero hunger". World Dev 118:1-14 doi:10.1016/j.worlddev.2019.02.004

Bong BB (2000) Bridging the Rice Yield Gap in Vietnam. FAO. http://www.fao.org/docrep/003/x6905e/x6905e0e.htm.

Boulay A-M et al. (2018) The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). Int J Life Cycle Assess 23:368-378

Brandt P, Herold M, Rufino MC (2018) The contribution of sectoral climate change mitigation options to national targets: a quantitative assessment of dairy production in Kenya. Environ Res Lett 13:034016 doi:10.1088/1748-9326/aaac84

Chaudhary A, Gustafson D, Mathys A (2018) Multi-indicator sustainability assessment of global food systems. Nat Commun 9:848 doi:10.1038/s41467-018-03308-7

Del Gobbo LC et al. (2015) Assessing global dietary habits: a comparison of national estimates from the FAO and the Global Dietary Database. Am J Clin Nutr 101:1038-1046 doi:10.3945/ajcn.114.087403

FAO (2018a) FAOSTAT Food Balance Sheets. http://www.fao.org/faostat/en/#data/FBS. Accessed June 8 2018

FAO (2018b) Kenya Cattle and poultry sectors: Livestock and livelihoods spotlight. I8978EN. http://www.fao.org/3/I8978EN/i8978en.pdf

FAO (2018c) Kenya Cattle and poultry sectors: Livestock production systems spotlight. i8270en. http://www.fao.org/3/i8270en/I8270EN.pdf

Fesenfeld LP, Schmidt TS, Schrode A (2018) Climate policy for short- and long-lived pollutants. Nat Clim Change 8:933-936 doi:10.1038/s41558-018-0328-1

Green RF et al. (2018) Greenhouse gas emissions and water footprints of typical dietary patterns in India. Sci Total Environ 643:1411-1418 doi:10.1016/j.scitotenv.2018.06.258

Hallstrom E, Carlsson-Kanyama A, Borjesson P (2015) Environmental impact of dietary change: a systematic review. J Cleaner Prod 91:1-11 doi:10.1016/j.jclepro.2014.12.008

Hansen A (2018) Meat consumption and capitalist development: The meatification of food provision and practice in Vietnam. Geoforum 93:57-68

Harris F, Green RF, Joy EJ, Kayatz B, Haines A, Dangour AD (2017) The water use of Indian diets and socio-demographic factors related to dietary blue water footprint. Sci Total Environ 587:128-136 doi:10.1016/j.scitotenv.2017.02.085

He P, Baiocchi G, Hubacek K, Feng K, Yu Y (2018) The environmental impacts of rapidly changing diets and their nutritional quality in China. Nat Sustain 1:122

Heller MC, Willits-Smith A, Meyer R, Keoleian GA, Rose D (2018) Greenhouse gas emissions and energy use associated with production of individual self-selected U.S. diets. Environ Res Lett 13 doi:10.1088/1748-9326/aab0ac

Herrero M, Grace D, Njuki J, Johnson N, Enahoro D, Silvestri S, Rufino MC (2013) The roles of livestock in developing countries. Animal 7:3-18 doi:10.1017/S1751731112001954

IPCC (2007) Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II, and III to the Fourth Assessment. International Panel on Climate Change, International Panel on Climate Change, Geneva, Switzerland.

IPCC (2013) Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

ISO (2014) Environmental management - Water footprint - Principles, requirements and guidelines. International Organization for Standardization. ISO 14046:2014.
Kanyinga K (2016) Devolution and the New Politics of Development in Kenya. Afr Stud Rev 59:155-167 doi:10.1017/asr.2016.85
Khai NH TT, Tin HC, Sanh NV (2018) Reducing Greenhouse Gas Emissions in Rice Grown in the Mekong Delta of Vietnam. Environ Pollut Climate Change 2:158 doi:10.4172/2573-458X.1000158
Kimani-Murage EW, Muthuri SK, Oti SO, Mutua MK, van de Vijver S, Kyobutungi C (2015) Evidence of a Double Burden of Malnutrition in Urban Poor Settings in Nairobi, Kenya. PLOS ONE 10:e0129943 doi:10.1371/journal.pone.0129943
Koehler G (2015) Seven Decades of 'Development', and Now What? J Int Dev 27:733-751 doi:10.1002/jid.3108
Lim SS et al. (2012) A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990–2010: a systematic analysis for the Global Burden of Disease Study 2010. The Lancet 380:2224-2260 doi:10.1016/S0140-6736(12)61766-8
MacRae R, Langer J, Cuddeford V (2012) Lessons from 20 years of CSO advocacy to advance sustainable pest management in Canada. In: MacRae R, Abergel E (eds) Health and sustainability in the Canadian food system: advocacy and opportunity for civil society. UBC Press, Vancouver, BC, pp 127-152
Mason P, Lang T (2017) Sustainable diets: How ecological nutrition can transform consumption and the food system. Routledge, London
Mekonnen MM, Hoekstra AY (2010a) The green, blue and grey water footprint of crops and derived crop products. UNESCO-IHE, Delft, The Netherlands. No. 47.
Mekonnen MM, Hoekstra AY (2010b) The green, blue and grey water footprint of farm animals and animal products. UNESCO-IHE, Delft, The Netherlands. No. 48.
Mekonnen MM, Hoekstra AY (2011) The green, blue and grey water footprint of crops and derived crop products. Hydrol Earth Syst Sci 15:1577-1600 doi:10.5194/hess-15-1577-2011
Mekonnen MM, Hoekstra AY (2012) A Global Assessment of the Water Footprint of Farm Animal Products. Ecosystems 15:401-415 doi:10.1007/s10021-011-9517-8
Minh Do L, Lissner L, Ascher H (2018) Overweight, stunting, and concurrent overweight and stunting observed over 3 years in Vietnamese children. Glob Health Action 11:1517932
Nguyen T, Roehrig F, Grosjean G, Tran D, Vu T (2017) Climate Smart Agriculture in Vietnam. International Center for Tropical Agriculture (CIAT), Hanoi, Vietnam. http://cgspace.cgiar.org/rest/bitstreams/155807/retrieve
Poore J, Nemecek T (2018) Reducing food's environmental impacts through producers and consumers. Science 360:987-992
Popkin BM, Adair LS, Ng SW (2012) Global nutrition transition and the pandemic of obesity in developing countries. Nutr Rev 70:3-21
Porter SD, Reay DS, Higgins P, Bomberg E (2016) A half-century of production-phase greenhouse gas emissions from food loss & waste in the global food supply chain. Sci Total Environ 571:721-729 doi:10.1016/j.scitotenv.2016.07.041
Pradhan P, Costa L, Rybski D, Lucht W, Kropp JP (2017) A systematic study of Sustainable Development Goal (SDG) interactions. Earth's Future 5:1169-1179
Song G, Li M, Fullana-i-Palmer P, Williamson D, Wang Y (2017) Dietary changes to mitigate climate change and benefit public health in China. Sci Total Environ 577:289-298 doi:10.1016/j.scitotenv.2016.10.184
Springmann M, Wiebe K, Mason-D'Croz D, Sulser TB, Rayner M, Scarborough P (2018) Health and nutritional aspects of sustainable diet strategies and their association with environmental impacts: a global modelling analysis with country-level detail. Lancet Planet Health 2:e451-e461
The World Bank (2018) GDP per capita (current US$). https://data.worldbank.org/indicator/NY.GDP.PCAP.CD. Accessed Sept 7 2018
Tilman D, Clark M (2014) Global diets link environmental sustainability and human health. Nature 515:518-522
Tom MS, Fischbeck PS, Hendrickson CT (2016) Energy use, blue water footprint, and greenhouse gas emissions for current food consumption patterns and dietary recommendations in the US. Environ Syst Decis 36:92-103 doi:10.1007/s10669-015-9577-y
Torbick N, Salas W, Chowdhury D, Ingraham P, Trinh M (2017) Mapping rice greenhouse gas emissions in the Red River Delta, Vietnam. Carbon Management 8:99-108
Trung ND, Anh LH, Thang NT, Sebastian LS (2017) The Challenges in Implementing Vietnam's Nationally-Determined Contribution (NDC) in the Agriculture Sector under the Current Supporting Laws, Regulations, and Policies. CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS), Wageningen, the Netherlands. CCAFS Working Paper No. 217. www.ccafs.cgiar.org
Udo H, Weiler V, Modupeore O, Viets T, Oosting S (2016) Intensification to reduce the carbon footprint of smallholder milk production: fact or fiction? Outlook Agric 45:33-38
UN FAO (2018) FAOSTAT Data portal. http://www.fao.org/faostat/en/#data. Accessed January 2019
USAID (2016) Greenhouse Gas Emissions in Vietnam. https://www.climatelinks.org/resources/greenhouse-gas-emissions-factsheet-vietnam
USDA Foreign Agricultural Service (2017) Vietnam Retail Foods - Sector Report 2016.
Vanham D, Comero S, Gawlik BM, Bidoglio G (2018) The water footprint of different diets within European sub-national

geographical entities. Nat Sustain 1:518
Vázquez-Rowe I, Kahhat R, Larrea-Gallegos G, Ziegler K (2019) Peru's road to climate action: Are we on the right path? The role of life cycle methods to improve Peruvian national contributions. Sci Total Environ 659:249-266
Vázquez-Rowe I, Larrea-Gallegos G, Villanueva-Rey P, Gilardino A (2017) Climate change mitigation opportunities based on carbon footprint estimates of dietary patterns in Peru. PloS one 12:e0188182 doi:10.1371/journal.pone.0188182
Vu T (2016) The Revolutionary Path to State Formation in Vietnam: Opportunities, Conundrums, and Legacies. J Vietnam Stud 11:267-297 doi:10.1525/vs.2016.11.3-4.267
Walker C, Gibney ER, Hellweg S (2018) Comparison of Environmental Impact and Nutritional Quality among a European Sample Population–findings from the Food4Me study. Sci Rep 8:2330 doi:10.1038/s41598-018-20391-4
Weiler V, Udo HMJ, Viets T, Crane TA, De Boer IJM (2014) Handling multi-functionality of livestock in a life cycle assessment: the case of smallholder dairying in Kenya. Curr Opin Environ Sustain 8:29-38 doi:10.1016/j.cosust.2014.07.009
World Bank: CIAT (2015) Climate-Smart Agriculture in Kenya. The World Bank Group, Washington D.C.
WRI (2017) CAIT Climate Data Explorer. World Resources Institute, Washington, DC. https://cait.wri.org
WULCA (2016) AWARE factors: sub-watershed and country level. http://www.wulca-waterlca.org/aware.html. Accessed February 2018
Yamaguchi T, Tuan LM, Minamikawa K, Yokoyama S (2019) Assessment of the relationship between adoption of a knowledge-intensive water-saving technique and irrigation conditions in the Mekong Delta of Vietnam. Agric Water Manage 212:162-171 doi:https://doi.org/10.1016/j.agwat.2018.08.041

Figure captions

Figure 1. Carbon footprint of diet in Vietnam. Figures show decadal trends in (a) annual per capita food supply for Vietnam according to FAO FBS and (b) associated greenhouse gas emissions (GHGE). See Figure 2 for an expansion of the "meat" category.

Figure 2. Trends in meat supply in Vietnam (a), alongside associated greenhouse gas emissions (b). While pig meat is dominant, small increases in bovine meat in the most recent decade outpace pig in terms of associated greenhouse gas emissions. Note that this figure is an expansion of the meat category in Figure 1. "Meat, other" is an FAO category representing other meats not specified.

Figure 3. Water use impacts of diet in Vietnam. Figures show decadal trends in (a) food supply associated annual water use per capita and (b) associated water scarcity footprint per capita. Annual food supply corresponds to Figure 1a, with the exception that beverages and fish and seafood are not included.

Figure 4. Carbon Footprint of diet in Kenya. Figures show decadal trend in (a) annual per capita food supply for Kenya and (b) associated greenhouse gas emissions (GHGE). "Other meat" is the sum of all meat supply except bovine meat.

Figure 5. Water use impacts of diet in Kenya. Figures show decadal trends in (a) food supply associated annual water use per capita and (b) associated water scarcity footprint per capita. Annual food supply corresponds to Figure 4a, with the exception that beverages and fish and seafood are not included.

Figure 6. Comparison of population growth and diet-associated GHGE for the total population of Vietnam and Kenya. Population data from (FAO 2018a). Mt = megatonnes = 1 million metric tons.

Figure 1.

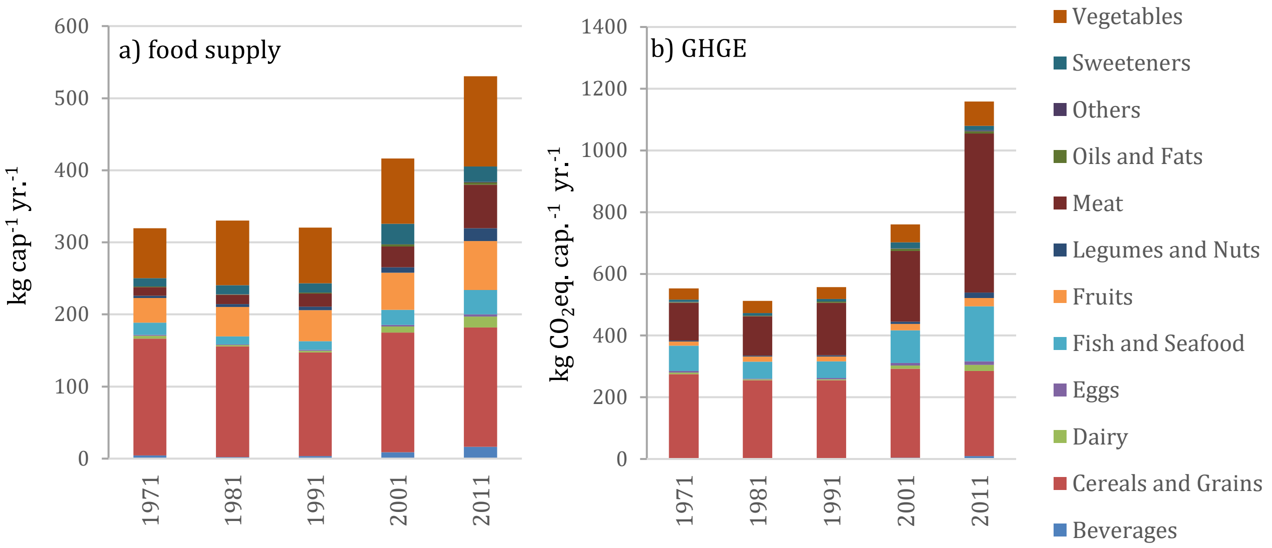

Figure 2.

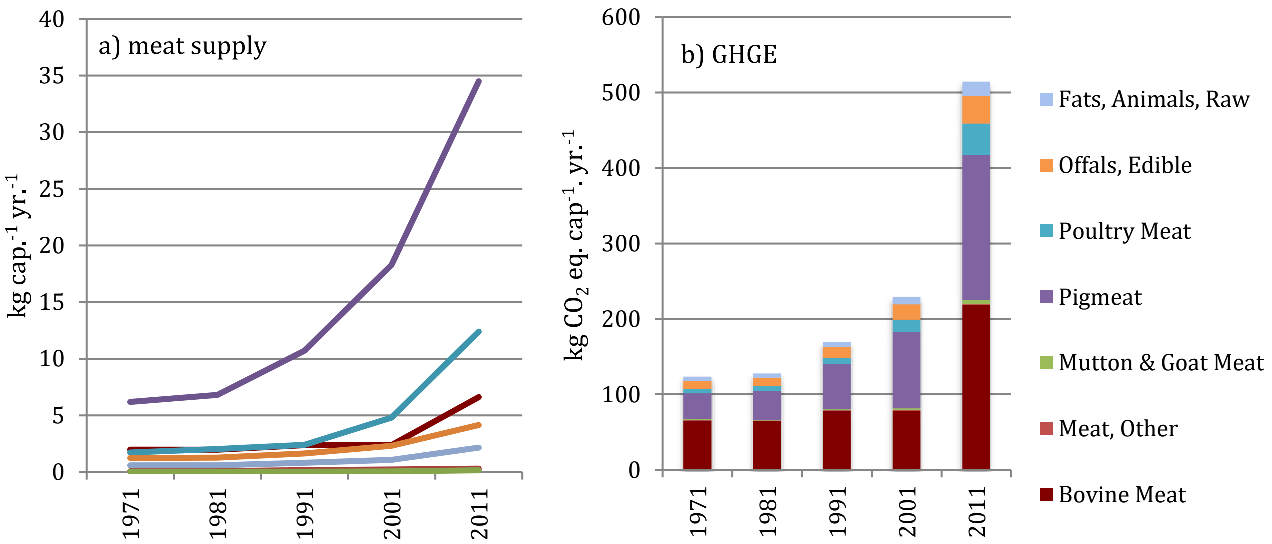

Figure 3.

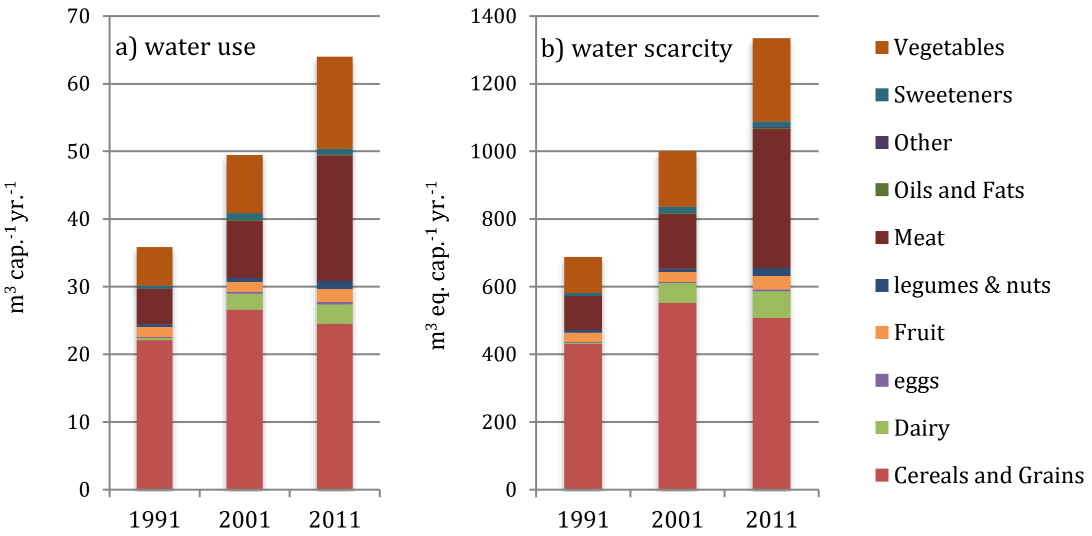

Figure 4.

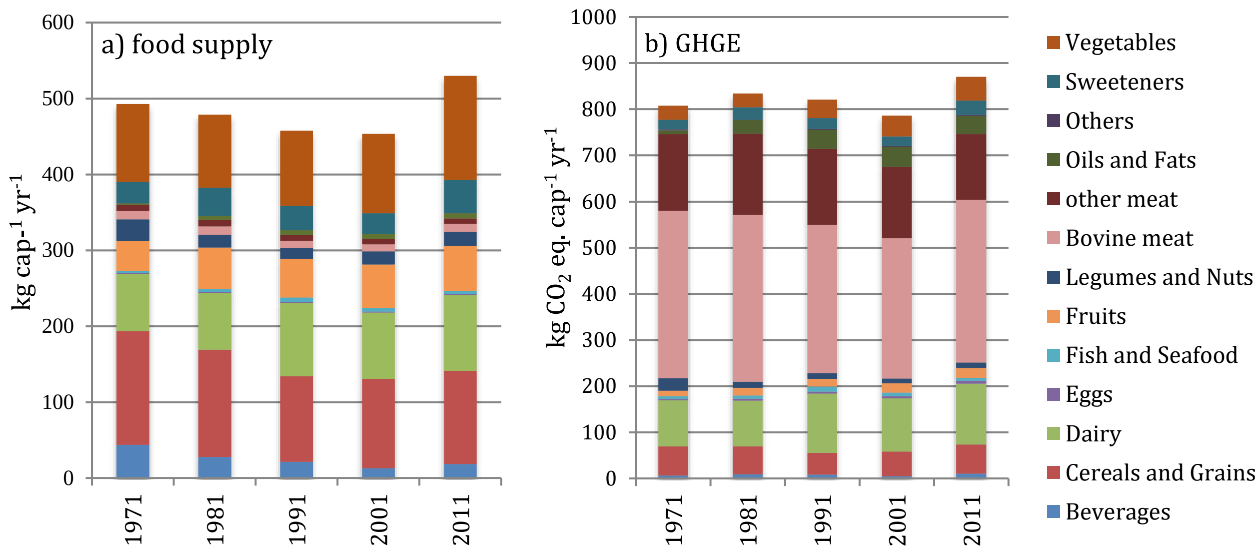

Figure 5.

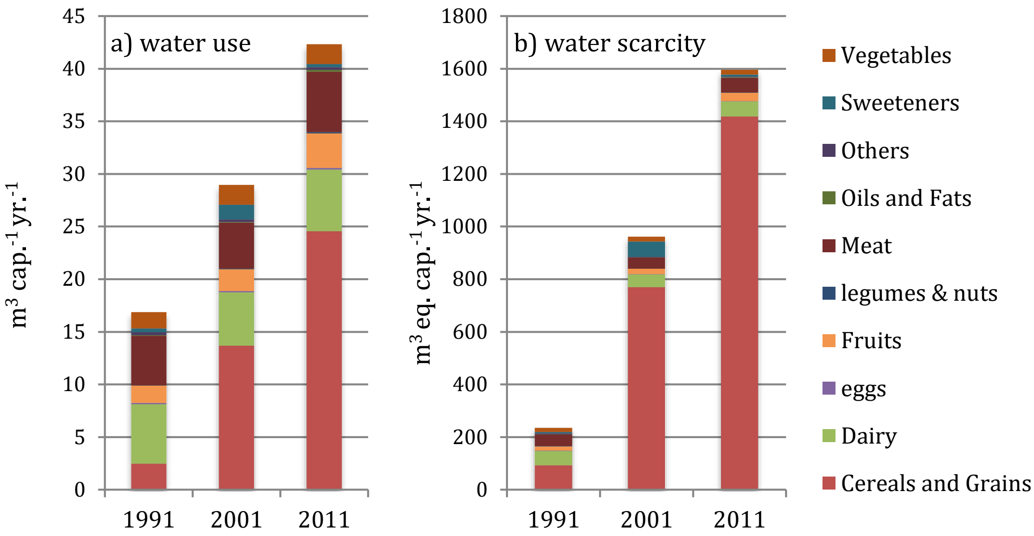

Figure 6.