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Ethical aspects of life cycle assessments of diets

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Abstract: Since the turn of the century a growing chorus of researchers has been espousing reduced meat and dairy intake as a partial strategy to transition towards a sustainable food system. Many of these studies have been predicated on a life-cycle assessment (LCA) methodology and though transparent in communicating their work within that framework, it has largely gone unmentioned that LCA involves a number of choices by the assessor and LCA methodology developers that are ultimately subjective. This study uses a consequential LCA of the average Danish diet in comparison to model vegetarian and vegan diets, leveraging the cultural perspectives afforded by the ReCiPe methodology, as starting point to explore the ways that subjectivity influences the LCA process and to test the robustness of the results against these different viewpoints. Mirroring earlier studies, we find vegetarian and vegan diets generally perform better environmentally compared to a standard Danish diet, but that there was minimal difference between the two no-meat options. Results were resilient to varying cultural perspectives applied in the model. LCA methodology, though loaded with value judgments, remains a dependable tool for assessing environmental dietary performance, but is less suited for estimating environmental pressures that are highly dependent on local conditions (e.g. chemical toxicity).

- **Keywords:** Life-cycle-assessment; Ethics; Diets; Vegetarian; Vegan; Sustainable consumption

Introduction

At the global level, food production is estimated to be responsible for between 20% and 50% of anthropogenic environmental impacts (McLaren, 2010, Notarnicola et al., 2012 and Roy et al., 2012). Irrespective of this pressure's true value, it is clear that global food consumption affects the performance of ecosystems negatively (locally and globally) through contributions to a variety of environmental issues including: climate change, water stress, toxic chemical release, air quality degradation, eutrophication of water bodies, soil erosion, and biodiversity losses (Cribb, 2010 and Foley et al., 2011). Ecosystem damages aside, current intensive agricultural systems rely on non-renewable resources (fossil fuels, land, and minerals) that are being exhausted and inefficiently employed (IBID). A projected 33% population growth – from 7 billion today to 12 billion by 2100 (Gerland et al., 2014) – with concurrently increased global economic activity (Price Waterhouse Cooper, 2010) will challenge the global agriculture system to produce more food with less resources while minimizing environmental impacts synchronously. Recent trends have been discordant with these ambitions, showing reduced growth in yields per unit production factor (land, fertilizer, etc.) in a number of countries as well as increased gross, non-renewable resource consumption from 1985–2005 (Foley et al., 2011 and Tilman and Clark, 2014).

Meat and dairy products are central to food-related impacts, having large environmental burdens including agricultural land degradation due to overgrazing, surface and groundwater contamination from uncontrolled waste management, biodiversity loss through the proliferation of grazing land (and land for feed production), and greenhouse gas (GHG) emissions related to livestock digestion (particularly ruminants) (Asner et al., 2004, Cribb, 2010, FAO, 2006, Modernel et al., 2013 and Nijdam et al., 2012). Due to the inherent inefficiencies of producing biomass at higher trophic levels (McMichael et al., 2007 and Pimentel and Pimentel, 2003), livestock production also requires calorific inputs amounting to 40% of global grain production (IBID; Foley et al., 2011). These feed requirements have environmental impacts embodied within their production, exacerbating the direct environmental disturbances of animal husbandry. Accounting for pastures and animal feed, livestock production is estimated to commandeer nearly one third of global, ice-free surface area (McMichael et al., 2007). These environmental pressures and land constraints are key issues if the predicted global animal product demand doubles from year 2000 levels by 2050 in response to population and economic drivers (FAO, 2006, Feeley and Machovina, 2014, McMichael et al., 2007 and Tilman and Clark, 2014).

Technological improvements to livestock production can mitigate some environmental harm, but eco-efficiency gains have failed thus far to mitigate net environmental impacts. Conversely, tackling this challenge on the demand side by reducing meat and dairy consumption has been championed as a way to improve the environmental integrity of nourishing humanity (FAO, 2006, Foley et al., 2011 and Tilman and Clark, 2014). This approach has been most salient in the United States Department of Agriculture's (USDA) 2015 dietary guidelines (2015). Indeed, environmental audits using life-cycle assessment (LCA) have shown that, low meat, vegetarian (no meat), and vegan (no meat or dairy) diets can have significant environmental benefits in comparison to prevailing dietary trends in wealthy countries (see Table 1). LCA estimates the resultant environmental impacts in a number of pertinent indicators from the supply chain (raw material extraction, processing, use, disposal, and related transport) required to deliver a product or service. These studies have shown univocally that vegetarian and vegan diets have reduced GHG emissions over standard omnivorous diets in a wealthy context. For other environmental impacts, LCA conclusions vary, showing that reduced animal product consumption reduces all accounted environmental impacts (Baroni et al., 2007), reduces particulate matter formation and land occupation (Saxe, 2014) or, conversely, exacerbates water consumption (Meier and Christen, 2013).

Table 1 - Previous environmental life cycle assessments of dietary habits

Reference	Country	Impacts Included				GHG Reduction (% change relative to omnivorous diet)	Other Comments
		Non-toxic	Toxic	H_2O Use	Land Use		
Heller and Kaoleian (2014)	United States	X				Vegetarian: 33% Vegan: 53%	
Saxe (2014)	Denmark	X		X	X	New Nordic Diet: 30% w/ reduced transport: 35% w/ organics: 32%	- land occupation reduction with reduced meat diet. - organic content of diet raised particulate matter and land occupation impacts.
Scarborough et al. (2014)	United Kingdom	X				Medium Meat: 21% Low Meat: 35% Pescatarian: 46% Vegetarian: 47% Vegan: 60%	- comprehensive diet survey used
van Dooren et al. (2014)	Netherlands	X			X	Vegetarian: 21% Vegan: 37%	
Meier et al. (2013)	Germany	X		X	X	Vegetarian: 25% Vegan: 50%	- water use inversely proportional to meat intake
Berners-Lee et al. (2012)	United Kingdom	X				Vegetarian: 22% Vegan: 36%	
Roy et al. (2012)	Japan	X				Not Applicable	
Saxe (2012)	Denmark	X				New Nordic Diet: 6% w/ optimization: 27% Vegetarian: 27%	- select local, organic and meat consumption performed equal to vegetarian
Macdiarmid et al. (2012)	United Kingdom	X				Reduced meat: 36%	- unrealistic sustainable diet achieved 90% reduction in GHGs
Tukker et al. (2011)	Europe	X	X			Reduced red meat: 8% Mediterranean: 5%	
Baroni et al. (2007)	Italy	X	X	X	X	Vegetarian: 74% w/ organic: 87% Vegan: 90% w/ organic: 97%	- ubiquitous superior performance across all impact categories with reduced meat
Wallén et al. (2004)	Sweden	X				Reduced meat: 5%	

Though compelling, the veracity of environmental benefits from reducing meat consumption has shortcomings. The common application of single issue indicators, chiefly the GHG burdens, dominates relevant literature (Berners-Lee et al., 2012, Heller et al., 2013, Roy et al., 2012, Saxe et al., 2012 and Wallén et al., 2004), running the risk that reduced meat diets may increase other environmental impacts (i.e. environmental burden shifting). Moreover, where expanded indicator sets covering more types of environmental pressures have been applied, paucities exist in illuminating the latent assumptions within the LCA framework and their potential consequences. Baroni et al. (2007) explored this theme with their analysis of the robustness of LCA results of dietary shifts to changes in assessor concern for different environmental impacts, both in terms of impact type and time-frame, finding that in general little change was seen with shifting assessor perspective. Aside from nascent investigation, there has been sparse discussion surrounding how the choice of indicators included in and LCA or the way that chemicals are modeled in the environment might affect dietary study results. Moreover, environmental efficacy has been ascribed to dietary choices even when the compared diets perform within the margins of error typically applied to LCA assessments.Herrmann et al. (2014) note that the margin of error can be significantly larger than the 10% uncertainty used in some of the reviewed studies. Lastly, with the exception of Saxe's work, studies have utilized attributional LCA models which are not representative of production systems at play with market forces (Plevin et al., 2014). Clearly, even within the LCA framework which strives for scientific objectivity, subjective values influence assessments, although this is only one aspect of the power of personal preferences in the discussion of the sustainability of diets.

A number of food related ethical discussions have gained momentum the past 20 years (Mepham, 1996) such as livestock welfare, food waste, food safety, food security, rural development, agricultural practices related to conventional, organic, and transpersonal agroecology, crops as biofeuls and the use of biotechnology as breeding tools on both animals and plants. "Sustainability" can mean many things in regard to food (Gamborg and Sandsøe, 2005) and various aspects of sustainability can easily conflict leaving one to choose between different values (e.g. land use efficiency and animal welfare) (Gjerris et al., 2011). Consumer-driven sustainability on food thus faces serious challenges, since it can be confusing as a consumer to determine the more sustainable choice. This is both because of scientific uncertainty, but also because of different and value-driven definitions of what "sustainability" actually is (Gjerris et al., 2016). From an environmental perspective, sustainability is roughly equated with humanity's stewardship of the environment in a way to not undermine its long- and short-term ability to provide natural resources, pollution assimilation and other ecosystem services to mankind, whilst concurrently supporting a meaningful proportion of the planet's wildlife and biodiversity. However, since sustainability is multi-faceted in nature (encompassing economic, environmental, social and institutional traits), the environmental performance concerning different diets is interconnected to discussions of food, culture, animals, humans relationship to nature, economics and values. Therefore, even though environment assessments (LCA, ecological footprint, emergy, etc.) are important to understanding and communicating environmental impacts related to diets, no assessment strategy completely covers all quantifiable (e.g. environmental and economic impacts) nor less-quantifiable (e.g. social issues) aspects of sustainability. In relation to this article the task therefore becomes to show what values drive different LCA methodologies to clarify the extent to which they affect the conclusion. Considering LCAs as value-neutral decision tools is precarious, as the values informing the political process used to develop LCA methodologies thus become hidden. Leaving decisions about sustainability to LCA experts does not make the decisions value-free, but merely ensures that it is the values of the experts that inform the decisions.

This paper presents an LCA comparing predicted environmental performance of average omnivorous Danish and conceptual vegetarian and vegan diets. Denmark provides an interesting case, because it has high per capita meat consumption (97 kg/a, 11th globally) (The Economist, 2012), produces a significant portion of its consumed meat and dairy (FAO, 2014), and enforces stringent energy and environmental controls on agricultural production. Moreover, this paper utilizes the full suite of LCA indicators, consequential LCA modelling methodology, and supporting databases not yet used in literature for dietary assessments at the time of writing. This study also explores the normative nature of environmental assessments and deduces the tractability of LCA as tool for comparing diets, with a discussion of the axiological ethical positions implicit in modeling environmental impacts using LCA.

Materials and Methods

LCA framework

LCA attempts to quantify the materials and energy consumed, and chemicals emitted to the environment during resource extraction, manufacturing, distribution, use, and end-of-life stages of a product/service (Guinée et al., 2002). LCA utilizes the *functional unit*concept in comparing different food products. In essence the functional unit strives to provide a common basis of comparison between different means of achieving the same end, strictly defined as a service that the assessed system(s) must fulfill (e.g. provide containment for a certain volume of liquid). The amounts of mass and/or energy required to meet that functional unit (e.g. the amount of ceramic or polystyrene needed to hold the amount of fluid defined by the functional unit) are called the reference flows.

Through the entire system life cycle, the LCA accounts exchanges (resource consumptions, energy, pollutant emissions) between different, well-defined environmental compartments (water, land and air in their different permutations) and the system (herein the ‘product-system’) providing the functional unit. Summing like flows of these resource inputs or pollutant outputs along the entire supply chain, a system inventory is generated for the total resource needs and pollution loading related to the functional unit. Lastly, the chemical and energy exchanges between society and environment are converted to environmental pressure potentials pertaining to the environment and human health. Resources used by the system and pollutants leaving the system are assessed for contributions to specific environmental problems (e.g. climate change, freshwater ecotoxicity, etc.) or resource scarcity issues (e.g. metal depletion). These scores represent *estimated* contributions to environmental and resource challenges imparted by the product system to fulfill the functional unit, called impact potentials (IPs).

IPs can be characterized at midpoint or at endpoint. Endpoint indicators model the entire impact pathway up to damages to 3 areas of protection (ecosystem quality, human health, and natural resources). Midpoint indicators stop earlier than endpoints along the cause-effect chain. For example, climate change impacts at the midpoint level are measured as the equivalent amount of carbon dioxide emitted to the atmosphere by the product system, while the product systems contribution to the damage to human health endpoint accounts for the estimated rise in temperatures and resultant loss of healthy years of living from disease, sea level rise and other factors. An assessor starts with the midpoint IPs and uses conversion factors which weight the contributions of that midpoint to a given endpoint category in order to create common unit that can be summed.

Though endpoint indicators are more meaningful from a decision-making perspective, they are less certain than midpoint (Hauschild, 2005). Lastly, the endpoint IPs themselves can be further weighted and summed to generate a single score, though this is hindered by subjectivity regarding the how the weighting should be done. LCA has seen increased harmonization in recent years, with the basic requirements outlined by the ISO 14040 series of standards, and detailed best practices guidance in the International Reference Life Cycle Data System (ILCD) handbook (EC, 2010, Finkbeiner et al., 2006 and Owsianiak et al., 2014). Lastly, LCA has seen increased application to food in recent years, viewed as an effective assessment method for environmental impacts food products.

Functional unit and scoping of the assessed diets

Different functional units for food LCAs have been proposed in the past: they can relate to agricultural areas, entire farms, a single livestock unit, quantities of food produced or consumed, nutritional values of meals (Haas et al., 2000). In this study, the primary function is considered to be the supply of adequate energy and nutrient levels to an adult person. The functional unit in this study will be taken as the provision of 2000 kcal per day of food excluding beverages aside from dairy. The United States Department of Agriculture recommends a daily calorific intake of 2000 kilo calories (kcal) per average adult (weighted for gender and age) (Venti and Johnston, 2002), with this standard adopted throughout Europe (Meier and Christen, 2013 and Van Dooren et al., 2014). It should be stressed that consuming 2000 kcal per day does not automatically equate to a nutritionally adequate diet. The inclusion of other nutritional metrics to ensure compatibility of the compared systems would improve the study (Heller et al., 2013), but as a rough guide for nutritional equivalency, calories suffices for the study at hand.

Three dietary patterns are assessed: the average Danish diet, and two recommended diets – an ovo-lacto vegetarian diet (no meat consumed, herein ‘vegetarian’) and a vegan diet (no meat or dairy products consumed). The scope of the assessment will stretch from the extraction of the raw materials necessary for the system up to the manufacture and production of the food products, with all processes beyond agricultural production excluded. Though this will underestimate total environmental impact by excluding processes downstream from the farm, it has been shown that food production is the dominant contributor to food-related environmental burdens (Davis et al., 2010, Meier and Christen, 2013 and Roy et al., 2012). For the use stage it has been shown that the processes of refrigeration and transport are typically the most important activities. Food miles tend to contribute marginally towards final environmental burdens, excepting cases involving air transport or long-distance refrigeration (Born and Purcell, 2006). Refrigeration itself, both in-store and at-home, can also be important contributors to life-cycle energy consumption and environmental impacts, though this is uncommon and not a priority in affecting food system sustainability. Furthermore, the impacts from cold storage speak more about the supporting energy system than the dietary choices themselves (Garnett, 2011). With regard to the disposal stage, impacts related to the incineration or composting of organic waste, both representative for Denmark, are not deemed to vary significantly between the three diet systems analyzed in this study. Thus, truncating their life cycles should not impact their comparative environmental performance. Finally, packaging is excluded from the assessments. The variety of possible packaging and cooking methods precludes sensible modelling, their inclusion adding marginal completeness in terms of impacts at the price of model robustness (Muñoz et al., 2010).

Data sources and inventory settings

The assessed diets were constructed from two sources. The standard Danish omnivore diet was taken from Danish consumption surveys for 2003 to 2008 and scaled from 10 MJ supplied energy to the functional unit (DTU Fødevareinstitutet, 2010). For the vegetarian and vegan diets, where actual consumption data was lacking, the recommended vegetarian and vegan diets were based on the 'Vegetarian food guide pyramid' (Loma Linda University – School of Public Health, 2008), which in turn relied on the US Department of Agriculture's nutritional guidelines (Haddad et al., 1999 and Venti and Johnston, 2002). The recommended diets list the required servings of broad food groups (e.g. whole grains, legumes, and soy, etc.) to meet the nutritional requirements of a balanced 2000 kcal/day diet. The broad food groups were disaggregated into the individual food components found in the Danish diet (e.g. the food group 'fruits and vegetables' is broken down into the food items like: 'tomatoes, cucumbers, and peppers'). The ratios of different food products available to the average Danish consumer according to Statistics Denmark food balance sheets (2014) were maintained for the vegetarian and vegan diets, but scaled to the amount required to meet recommendations for 'vegetables' in the food pyramid (this was done for all food groups). As such, the conceptual vegan and vegetarian diets reflect Danish consumer habits assuming that the food balance sheet expresses consumer demand. Moreover, in keeping with the system boundary of the farm, certain foods had to be dissected to their base agricultural constituents (e.g. bread was converted to grains), with the exception of vegetable oils from complex bio-refineries.

Food losses occur due to pests, damage during harvesting, processing losses from aesthetic or functional quality control, rough handling and spoilage during distribution, and at the retail and consumer due to further spoilage (FAO, 2011). Farm losses were internalized within individual modelling processes, since these scale total inputs and outputs at the farm to mass of product delivered at farm gate. Post farm-gate loss factors (in an OECD context) of 8%, 19%, 31%, 26%, and 32% for meat, dairy products, cereals, fruits and vegetables, and roots and tubers, respectively were taken from FAO (IBID) and applied to the diets. As such, the reference flows in Table 2 are inflated above actual consumption, representing demand at farm-gate necessary to supply the 2000 kcal/day for the given diets. Calculations are outlined in Supplementary Material S1 and S2.

Table 2 - Food demands at farm gate to meet a functional unit of 2000 kcal/day for the three considered diets and associated processes used in modelling. Diets do not include drinks (barring dairy) and vegan and vegetarian have high water content in foods consumed.

Food Item	**Omnivorous (g/day)**	**Vegetarian (g/day)**	**Vegan (g/day)**	**Process (All ecoinvent 3 unless other sources are listed)**
Dairy and Eggs				
Milk	278.4	449.8	-	(LCA Food, 2007)
Cream	31.3	-	-	(Weidema and Schmidt, 2014)
Creme fraiche	7.8	-	-	IBID
Butter	6.0	-	-	IBID
Cheese	28.2	19.1	-	IBID
Eggs	14.5	31.0	-	(Nielsen et al., 2013)
Meat				
Beef and veal	47.7	-	-	(Nguyen et al., 2010)
Edible offals of cattle	1.4	-	-	IBID
Pig meat	54.2	-	-	(Reckmann et al., 2013)
Edible offals of pigs	1.9	-	-	IBID
Poultry meat	22.7	-	-	Chicken for slaughtering
Mutton and lamb	2.0	-	-	Goat for slaughtering
Grains				
Wheat flour	141.6	271.0	316.2	Wheat
Durum wheat e.g.	15.1	0.0	0.0	Wheat
Rye flour	33.8	21.7	25.4	Rye grain, rye production
Oat-meal	24.7	0.0	0.0	Wheat
Rice and rice flour	15.8	144.9	169.1	Rice, production
Potato flour etc.	2.0	0.0	0.0	Potato
Other flour and groats, etc.	24.0	0.0	0.0	Wheat
Fruits and Vegetables				
Potatoes	115.6	238.4	238.4	Potato
Cucumbers	25.3	47.1	47.1	Cucumber
Spring-white cabbage	9.0	16.7	16.7	Cabbage white
Spring-red cabbage	9.0	16.7	16.7	Cabbage red
Brussels sprouts	0.6	1.1	1.1	Radish
Broccoli	11.8	11.0	11.0	Broccoli
Cauliflower	11.8	11.0	11.0	Cauliflower
Chinese cabbage	6.8	11.0	11.0	Cabbage red
Leeks	6.8	12.6	12.6	Celery
Beetroots	5.4	10.1	10.1	Radish
Celery	2.6	4.8	4.8	Celery
Carrots	44.5	82.8	82.8	Carrot
Onions	31.2	58.1	58.1	Onion
Lettuce	25.0	46.6	46.6	Lettuce
Tomatoes	106.2	272.3	272.3	Tomato
Cherries sour and sweet	6.6	16.9	16.9	(Carlsson-Kanyama and Emmenegger, 2000)
Strawberries	9.7	24.8	24.8	(Gunady et al., 2012)

Apples	169.9	435.6	435.6	Apple
Pears	23.9	61.2	61.2	Pear
Protein Substitutes				
Beans	0	135.1	135.1	Fava bean
Tofu	0	94.6	94.6	(Ercin et al., 2012)
Soy Beverage	0	32.4	32.4	(Ercin et al., 2012)
Peanuts	0	20.3	40.5	(University of Arkansas, 2012)
Cashews	0	20.3	40.5	(Figueiredo et al., 2014)
Oils and Sugar				
Vegetable Oil	0.0	7.6	7.6	(Stevenson, 2014)
Margarine	30.9	7.6	7.6	(Nilsson et al., 2010)
Sugar	30.1	0.0	0.0	Sugar from beet

System modelling

Two types of LCA modelling frameworks exist, namely the attributional and the consequential modelling, the choice of which has been a continuous source of polemic in the LCA community (e.g. Ekvall and Weidema, 2004 and Weidema, 2003). Consequential LCA differs from attributional LCA in two main ways: (1) the processes encompassed in the study are those which are most likely to respond to a change in demand, and (2) the co-product allocation is avoided by system expansion (Schmidt and Weidema, 2008). In this study, we have opted for a consequential modelling to reflect the environmental consequences that the change in diets may imply on the systems within and outside the primary agricultural processes, e.g. market reactions to proposed future consumption (ex-ante modelling). This approach is also in compliance with the ISO14044 requirements (ISO, 2006). For instance, in Denmark, butter (a by-product of milk) requires milk fat, which is re-allocated from high fat cheese and powdered milk production. Thus, when butter is demanded, powdered milk manufacturers substitute palm oil for milk fat, while high fat cheese production decreases forcing consumers to purchase other comparable fats (low fat cheese). Thus, in a consequential model butter is modeled as the amount of palm oil and low fat cheese produced in response to market demand for butter which are then translated into estimated environmental impacts (Weidema and Schmidt, 2014).

The implementation of the consequential modelling was facilitated by the use of the ecoinvent 3.1 database, which exist in 2 versions dedicated to attributional and consequential modelling, respectively (Weidema et al., 2013). The consequential database, containing inventories of resource consumption and pollutant releases for the different foodstuffs, was therefore utilized in the study. In conjection with LCA software, the database can model interactions with other systems by use of marginal data, which model supplies of products by taking a mix of all unconstrained suppliers in the market, i.e. those suppliers who can respond to the next unit of demand for that good in the market (Weidema et al., 2013). This database is deemed to be a marked improvement over those utilized in earlier dietary comparison studies, since it includes an expanded set of food production processes and utilizes a full-fledged consequential LCI modelling framework. Where appropriate processes were lacking in ecoinvent 3, custom processes were built using inventories from reliable sources such as peer review LCAs or the Danish LCA Food database (Bengoa, 2005, Cederberg et al., 2009, Meier and Christen, 2013 and Nilsson et al., 2010). These were kept consistent with the consequential modelling by using system-expansion with marginal data, where necessary. The ecoinvent processes and data sources utilized for custom processes are outlined inTable 2. Breakdowns of custom processes are in Supplementary Material S3.

Typically relevant in the modelling of agri-food systems, indirect land use change is defined as the life cycle consequences of the land use in the analyzed system, e.g. deforestation or cropland intensification taking place as a result of the change in demand from the system (Schmidt et al., 2015). The inclusion of indirect land use change (iLUC) effects may alter the IPs of an LCA through increased GHG emissions and biodiversity loss from deforestation (e.g. Dalgaard et al. (2014)) potentially changing the best performing product-system, and it is widely accepted that the problems related to iLUC should be integrated into decision-making (Schmidt et al., 2015). However, despite the recent release of frameworks for performing iLUC, e.g. Schmidt et al. (2015), there is yet no consensus on the approaches to integrate iLUC into LCA modelling, which is still the source of debate, particularly in the assessments of biofuels (Finkbeiner, 2014,Finkbeiner, 2013, Munoz et al., 2014 and Schmidt et al., 2015). For this reason and due to the lack of insights into indirect land use change mechanisms triggered by the dietary changes, as reflected in the review by (Hallström et al., 2015), iLUC effects were not considered in the present assessment. As also recommended by Hallström et al. (2015), this important source of uncertainties, of which it is difficult to predict the influence on the results of the study, should however be addressed in future studies.

Impact assessment methods

There exist a number of competing life-cycle-impact assessment (LCIA) methodologies for modelling midpoint and endpoint IPs in LCA. The dissimilarities come from the varying choices used for modelling how chemicals disperse through the environment and to what extent they affect encountered organisms (Hauschild et al., 2012). The ReCiPe 2008 methodology was selected as it covers the whole spectrum of relevant environmental indicators at both midpoint and endpoint levels, and includes the possibility for differentiating across three cultural perspectives, namely the egalitarian, individualist, and hierarchist perspectives. The egalitarian perspective is sensitive to all environmental impacts (long and short term), uses preventive thinking in assessing pollutants, and aims for minimizing society’s impacts on the ecosphere. Opposing this is the *individualist*, which is concerned with current environmental impacts within their lifetime. This assumes that technological progress can solve eventual environmental woes and that ecosystems are resilient against human intervention. The hierarchist lies between these two representing a intermediary (Goedkoop et al., 2009). Table 3 outlines the indicators used in ReCiPe and how the different cultural perspectives view them. It should be noted that ReCiPe's water scarcity and land use indicators were not used here as more nuanced methods were deemed necessary for the assessment.

Table 3 - Assumptions behind cultural perspectives in ReCiPe 2008 (Goedkoop et al., 2009)

Midpoint Indicator	Assumptions at midpoint level			Assumptions moving from midpoint to endpoint		
	Egalitarian	**Hierarchist**	**Individualist**	**Egalitarian**	**Hierarchist**	**Individualist**
Climate Change	500 year time horizon	100 year	20 year	no societal adaptation, high human health impacts and biodiversity loss	medium societal adaptation, mean human health impacts and biodiversity loss	full societal adaptation, low human health impacts and biodiversity loss
Ozone Depletion	Identical			Identical		
Terrestrial Acidification	500 year time horizon	100 year	20 year	500 year time horizon	100 year	20 year
Freshwater Eutrophication	Identical			Identical		
Marine Eutrophication	Identical			Identical		
Human Toxicity	Infinite time horizon, all exposure routes for all chemicals, chemical toxicity considered	Same as egalitarian, except 100 year time horizon	100 year time horizon, limited exposure pathways for metals, selected chemical toxicity considered	Identical		
Photochemical oxidant formation	Identical			Identical		
Particulate matter Formation	Identical			Identical		
Terrestrial Ecotoxicity	Infinite time horizon, all exposure routes for all chemicals, chemical toxicity considered	Same as egalitarian, except 100 year time	100 year time horizon, limited exposure pathways for metals, selected chemical toxicity considered	Identical		
Freshwater Ecotoxicity	Identical to Terrestrial Ecotoxicity			Identical		
Marine Ecotoxity	Infinite time horizon, all exposure pathways possible	Same as egalitarian	100 years, limited exposure pathways for some chemicals	Identical		
Ionising Radiation	100 000 year time horizon	Same as egalitarian	100 year time horizon	Identical		
Mineral Resource Depletion	Identical			Identical		
Fossil Fuel Depletion	Identical			Technology will slowly substitute fossil fuels	Same as egalitarian	Technology will quickly substitute fossil fuels

Results

Table 4 provides the impact indicator results for the three diets in terms of percentage difference from the omnivorous diet. Dark grey indicates the worst performing diet for that indicator, black the medium performing diet (where applicable), light grey the best performing, and white a tie across all diets. In our assessment results of the diet with a 25% standard deviation assumed, whereby IPs with overlapping confidence intervals were assumed to have no appreciable difference. Minute dissimilarities were thus ignored and claims about superior diet performance could not be made based on these.

The results mirror those of previous diet comparison studies, since they show a clear difference between the omnivorous and non-meat diets, with the latter showing superior performance in a number of categories (see light grey cells in Table 4). The source of the poor performance of the omnivorous diet is the reliance on animal based products, as outlined in the climate change impacts and freshwater eutrophication impacts in Table 4. Beef is particularly pernicious in that it requires large quantities of inputs (feed, water and land) and results in large amounts of digestive waste (affecting eutrophication), and greenhouse gases (Nijdam et al., 2012). In terms of compatibility with similar studies, climate change provides the best comparative indicator due to its ubiquity. Relative dietary climate change performance was within the ranges found previously (see Table 1). Climate change IPs also agreed with earlier studies for the omnivore; 4.27 kg CO_2eq/day compared to 4.1, 3.02 and 4.09 kg CO_2eq/day for the average Dutch, US MyPate and average French diets respectively (van Dooren et al., 2014) and 5.6 CO_2eq/day for the average Dane (Saxe, 2014), though low compared to other studies that

Table 4 - Relative environmental performance of the different diets shown as percentage deviation from the omnivorous diet. Light grey indicates best, black medium, dark grey worst performing. Where two diets had the same performance in an indicator, they will share the relevant color. White indicates a tie across all diets. Note that a 25% divergence from the omnivorous diet does not guarantee superior performance in a category, since possible values may overlap.

	Omnivorous			Vegetarian			Vegan		
Midpoint Impact Category	I	H	E	I	H	E	I	H	E
Climate Change	-	-	-	-56%	-46%	-38%	-70%	-60%	-52%
Ozone Depletion	-	-	-	-3%	-3%	-3%	0%	-1%	0%
Terrestrial Acidification	-	-	-	-64%	-65%	-66%	-79%	-81%	-81%
Freshwater Eutrophication	-	-	-	-6%	-7%	-6%	-24%	-24%	-24%
Marine Eutrophication	-	-	-	-33%	-33%	-33%	-72%	-72%	-72%
Human Toxicity	-	-	-	-33%	14%	30%	-53%	5%	25%
Photochemical Oxidant Formation	-	-	-	6%	6%	6%	0%	-1%	0%
Particulate Matter Formation	-	-	-	-47%	-47%	-47%	-60%	-60%	-60%
Terrestrial Ecotoxicity	-	-	-	-6%	-8%	0%	45%	41%	43%
Freshwater Ecotoxicity	-	-	-	2%	2%	2%	-1%	-2%	-1%
Marine Ecotoxicity	-	-	-	3%	1%	24%	-5%	-7%	20%
Ionization Radiation (human health)	-	-	-	66%	15%	14%	67%	15%	14%
Metals Depletion	-	-	-	11%	11%	11%	8%	7%	8%
Fossil Depletion	-	-	-	-18%	-18%	-18%	-22%	-22%	-22%
Water Scarcity Index		-			26%			31%	
Land Use		-			-67%			-78%	
Endpoint Impact Category	I	H	E	I	H	E	I	H	E
Human Health	-	-	-	-54%	-44%	-10%	-68%	-57%	-19%
Ecosystems Damage	-	-	-	-56%	-46%	-38%	-70%	-60%	-52%
Resource Depletion	-	-	-	-9%	-14%	-14%	-13%	-19%	-19%

included transport and processing impacts (Berners-Lee et al., 2012 and Saxe et al., 2012).included transport and processing impacts (Berners-Lee et al., 2012; Saxe et al., 2012).

Impacts from pollution at farm

For climate change, methane emissions from cattle increase the IPs of the omnivorous diet well beyond the error threshold; this is reasonable considering that bovine enteric fermentation accounts for 18% of global methane emissions (McMichael et al., 2007) and significant N_2O release (Nguyen et al., 2010). Livestock production also perturbs the environment through feces and urine, which contain ammonia and nitrates. This also contributes to acidification or particulate matter formation if evaporated, or marine eutrophication (Gliessman, 2015). Plant production contributes to these IPs through over fertilization, which can result in nutrients runoff into receiving waters, or tilling, which activates the production of gaseous NOx by soil bacteria (IBID). In the assessment, excrement from livestock was the dominant factor resulting in the declined performance of the diets with increased animal product intake. Freshwater eutrophication is a consequence of phosphorous release to freshwater bodies from both animal excrement and fertilizer runoff, with all of the diets having similar performance in this regard as shown in Table 4. Lastly, though animal waste strongly influences photochemical oxidant formation (smog), the vegan's higher consumption of greenhouse-produced cucumbers and tomatoes led to similar scale IPs due to external heating needs.

Impacts from agricultural production inputs

A number of IPs can be traced to the chemicals and energy consumed in food production. Ozone depletion IPs are linked to diesel used on farms, but also pesticide production, with no differentiation in diet performance. Fossil fuel based fertilizer impacts are the same for all systems, though the non-meat diets are borderline superior, which is logical due to the exorbitant feed requirements for animal production (FAO, 2006). For fossil depletion the 25% standard deviation may be too liberal considering the reduced uncertainty surrounding the modelling of fossil fuel consumption; allowing for defensible prima facie conclusions here. Land occupation is adversely impacted by the imported livestock feed requirements and grazing territory (Foley et al., 2011), creating a large gap between omnivorous and non-meat diets. Of interest is that the meat-protein substitute, fava beans, contributed significantly to land occupation (∼12%), which is of note since this is a proxy for all types of beans consumed by the meat-free diets. Moreover, the ecoinvent 3.1 inventory shows that land occupation is low compared to other potential LCIs (Abeliotis et al., 2013) by up to a factor of three, however overall results remained robust to this uncertainty when land occupation for the fava beans was increased by this factor (vegetarian and vegan land occupation IPs relative to omnivorous diet changed to −62% and −73%, respectively). No differences were noted for metal depletion across the different diets or ionizing radiation (primarily related to pesticide production) since they all are heavily reliant on these inputs. However, when ionization radiation is compared at the individualist level the meat-free diets have worse performance due to significant pesticide inputs to fruit and vegetable production (particularly apples), but these differences dissolve at the egalitarian and hierarchist level, as a longer timeframe for potential impacts are taken (see Section 'Difficulties with toxicity impacts'). Water impacts are also worse for the two non-meat diets, as a result of the large irrigation inputs (exacerbated by large losses from runoff), echoing the findings of Meier and Christen (2013) and hinting a need for future research in more accurately quantifying impacts in this category. Lastly, animal feed requirements compound climate change IPs through fossil fuel based fertilizer needs and deforestation associated with soy protein feeds (Flysjö et al., 2012).

Difficulties with toxicity impacts

Ambiguous results were also illustrated in midpoint categories dominated by inputs to agricultural production. Pesticides dominated all of the toxicity IPs; a natural consequence of the toxic properties for which they are produced. Non-meat diets had lower human toxicity IPs, though only from an *individualist* perspective, since this viewpoint focuses on malicious substances with short term impacts (i.e. some chemicals are bad), while the other perspectives account for more chemicals (i.e. many chemicals are bad) (Goedkoop et al., 2009), thus blurring the performance of the diets as more pesticides are considered, with human toxicity IPs articulating this clearly. This trend can also be seen for marine ecotoxicity IPs where the egalitarian results for the non-meat diets are highest relative to the other methods, though no discernible difference was seen between diet choices. For terrestrial ecotoxicity, the vegan diet performed worst, though this was a shortcoming of the LCIA method. Soybean feed coproduces soybean oil resulting in avoided palm oil production on the market, with the avoided IPs credited to the feed. Pesticides used in palm production are included in ReCiPe LCIA methodology, while some of those for soy are not. Thus, the avoided palm impacts outweigh the soy impacts, producing a net negative IP. Thus, the animal product diets (which include feed) appear to perform better. However, with LCIA methods – using more complete chemical inventories (e.g. UseTox) (Owsianiak et al., 2014) – contradicting results are seen. Such deficiencies accentuate the difficulties of chemical toxicity modelling within LCA, forcing the question of whether a 25% uncertainty level is valid for toxicity impacts. Moreover, this modelling artefact explains why the vegan diet was borderline worse for fresh- and marine water eco-toxicity.

Endpoint impacts

Table 4 displays the ReCiPe IPs aggregated and assessed at the endpoint level, providing a comparative overview of the diets' IPs. The endpoint results succinctly showed what the midpoints communicated that the average Danish diet has larger IPs for ecosystems damage and human health than the vegetarian or vegan diets. However, the latter has ambiguous results when seen from the egalitarian perspective, due to the uncertainties in the toxicity modelling (see Section 'Difficulties with toxicity impacts'). Lastly, no discernible difference was observed for resource depletion, although – as discussed above – fossil fuel use appears higher

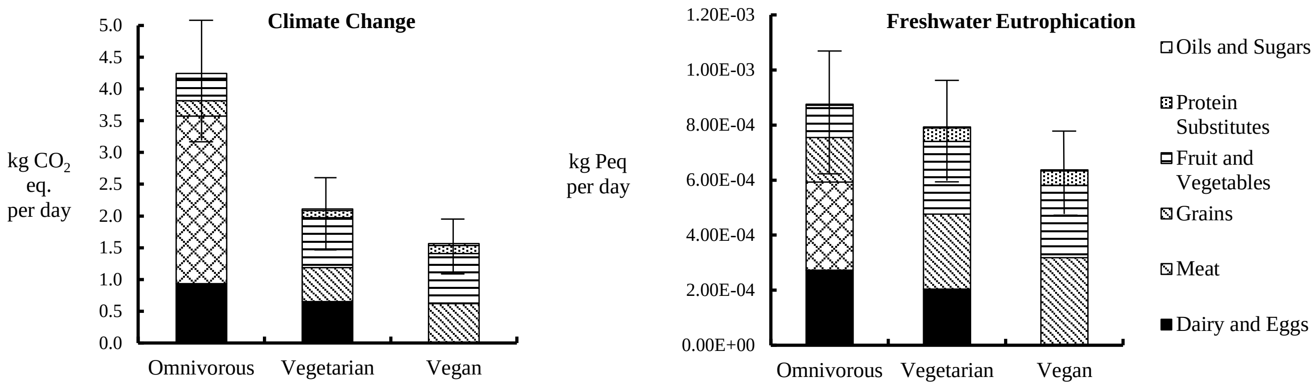

Figure 1 - Climate change and freshwater eutrophication midpoint IPs using hierarchist perspective in kg CO_2eq/day and kg Peq/day (Peq – phosphorous equivalents), respectively

in the livestock dependent diets. Notwithstanding differences at midpoint level, both non-meat diets performed equally at the endpoint.

Discussion

Results supported those of other studies in that the standard omnivorous diet performed poorly compared to model vegetarian and vegan diets. This was evident by the comparable performance of the meat-free vegetarian diet to the meat- and dairy-free vegan diet. Only within the realms of toxicity and eutrophication can we see appreciable divergence between these two choices, due to pesticide regimes and animal waste, respectively. Though conclusions could be drawn about the comparative performance for some specific IPs, ambiguity is present in others. Fig. 1 exemplifies this with the clear distinction between the omnivorous and non-meat diets for climate change, but inconclusive results for freshwater eutrophication. Table 4 takes this further by displaying both the various trade-offs between the dietary choices and the dependence of the results on the perspective of the assessor, thus challenging the objectivity of the LCA process, and necessitating a re-inspection of the cardinal ethical precepts embedded within the methodology. These ethical issues are related not only to the cultural perspectives applied in this specific study, but also to questions about the use of the LCA as a decision-making tool. Also, the ethical values that this decision-making tool entails, opens a wider debate, which must be taken up. This re-evaluation may not eradicate the elucidated uncertainties, but will at least support the validity of using LCA as decision support tool.

Sensitivity of results to ReCiPe cultural perspectives

We have taken some ethical aspects of food production and its ability to utilize differing cultural and ethical perspectives (individualist, hierarchist, and egalitarian) in characterizing environmental impacts into consideration with ReCiPe 2008. One of the purposes of the cultural perspectives is to allow for results interpretation in the face of uncertainty. For instance, with toxicity IPs where there is higher uncertainty due to challenges of adequately including the toxilogical properties in LCA models of all of the chemicals in commercial use, the user can adopt an ethical perspective that deals with this data gap in a way that aligns with their thinking of nature how nature works and the potential risks of underestimating IPs.

In ReCiPe the *egalitarian* is most worried by environmental impacts (long and short term), the individualist the least, while the hierarchist represent a middle view, although with a valence towards the egalitarian's stance (see Table 4) (Goedkoop et al., 2009). According to the cultural theories as presented here, egalitarians tend to perceive nature as an ephemeral entity, highly sensitive to perturbations, whereas hierarchists view nature as surprising in the sense that it "may hide the response when exposed to stress and at some time flip to another state in a more or less irreversible manner." (Finnveden, 1997). Obviously, the moral theory most closely related to egalitarians is egalitarianism (equal treatment for all agents affected by a situation) as it is reflected in environmental ethics such as Deep Ecology (Naess, 1973). The moral theory that might be related to the hierarchist point of view is utilitarianism, i.e. this posits maximizing happiness and minimizing pain. Finally, individualists tend to perceive nature as resilient in the sense that it will vacillate from its baseline state when exposed to stress, but can return to the baseline state if the stress is lessened or removed (Finnveden, 1997 and Shwarz and Thompson, 1990). The moral position of libertarianism and especially "Green libertarianism", which opposes regulation and advocates the maximum freedom of individual action compatible with equal freedom for all (Davidson, 2009), invoking IsaiahBerlin's notion of negative liberty (1964), is strongly related to the individualist viewpoint is.

As shown above, the LCA results were robust against the application of these attitudes, with the exception of conclusions about impacts to human health through toxicity and ionizing radiation (midpoint level) and damages to human health (endpoint level). Moving from the individualist to the hierarchist and egalitarian standpoints resulted in the accounting of more and more uncertain or long-term environmental impacts. This was illustrated by the human toxicity midpoint IPs where the individualist perspective narrowly focused on chemicals with well documented and/or acute toxicities, while the other outlooks included less immediately harmful chemicals and took a precautionary stance towards those *suspected* of being toxic (Goedkoop et al., 2009), with the general effect of blurring the comparative performance of the diets with this indicator and the endpoint human health IPs. Toxicity IPs are some of the most difficult impacts to predict with LCA, since actual toxicological impacts are extremely dependent on the unique assimilative capacity of the receiving body (ecosystem or animal). This is a major methodological challenge for an assessment tool such as LCA that does not yet spatially disaggregate chemical releases (IBID). Thus, the individualist's skepticism about the toxicity impacts may be warranted, especially when LCA is applied to numerous foods from a global market composited into a single diet.
In general all of the midpoint categories were robust against the cultural perspectives aside from the two exceptions noted above, with the further consequence of these exceptions promulgating through the LCIA calculations to affect the endpoint IP damage to human health. Table 5 outlines how the cultural perspectives affected those IPs that were sensitive to them.

Midpoint Impact Category	Effect of cultural perspective
Human Toxicity	Inclusion of increasing number of chemicals included in the LCIA when moving from I to E perspective made the model more sensitive to the herbicides and pesticides in vegetarian and vegan diets, erasing differences in environmental performance of the diets
Ionization Radiation (human health)	Longer timeframe of impacts considered when moving towards H and E perspectives meant that the impacts from more of the pesticides and herbicides used attenuated differences between the omnivorous and non-meat diets
Endpoint Impact Category	
Human Health	The impacts of the increased sensitivity of the human toxicity and ionization radiation (human health) affected the conversion from midpoint to endpoint IPs introducing ambiguity between dietary performance when the E perspective was adopted

Table 5 - Midpoint and endpoint IPs sensitive to cultural perspective chosen in the comparison of diets

The implicit values embedded within LCAs

The framework of LCA is built upon a number of implicit values. This section attempts to lift the veil of the many philosophical and ethical principles that an LCA assessor or LCA study commissioner accepts in choosing LCA as method to assess the environmental performance of product systems.

LCA methodology is predicated on the belief that we are able to predict environmental impacts. This belief is in itself based on our views of nature (Finnveden, 1997), which may differ considerably amongst both decision-makers and stakeholders. Our faith in LCA's ability to inform decisions hinges on the belief that nature is complex yet predictable rather than inherently random, influencing not only how we model IPs, but also whether LCA is even capable of providing answers to the fundamental questions being asked. If the general belief is that we are not able to predict environmental impacts, LCAs are nonsensical and one should seek alternative valuation methods that circumvent evaluating environmental damages (Finnveden, 1997).

If we accept that LCAs provides valuable insights for decision-makers, it is important to realize that LCA studies are grounded in different theoretical constructs and that the choice of assessment approaches entails ethical implications as well (Ekvall et al., 2005 and Finnveden, 1997). In our study, we used a consequential modelling approach, which falls into what Ekvall et al. (2005) term a "prospective life cycle assessment (LCA)", which provides information on the environmental consequences of individual actions in a dynamic system. This is contrasted with "retrospective" LCA's assessment of the environmental impacts in a static system without constrained suppliers. Ekvall et al. (2005) analyzed different LCA methodologies against different normative moral philosophy theories and found that each LCA type, as well as each of the moral theories, can be criticized from the alternative point of departure and that both prospective and retrospective LCAs had pros and cons. According to Ekvall et al. (2005) the use of prospective LCA is valid *if* the audience equates positive environmental outcomes with 'good' changes to a production system. It follows that decision-makers and people in general have differing opinions on what constitutes a good environmental action depending on their ethical values, since they may actually be indifferent to the state of the environment. For instance prospective LCA methodology is valid from a teleological framework, whereby consequences of an action are the criterion for success or failure (utilitarianism with its maximization of universal pleasure employs this thinking) (Finnveden, 1997), which is in opposition to deontological ethics which evaluates good or bad according to the principles underpinning actions and not necessarily outcomes (Ekvall et al., 2005).

Ekvall et al. (2005) found that “the sheer diversity of ethical theories makes it impossible (sic) to decide whether an environmentally good action is an action that reduces the environmental burdens of the total life cycle or an action with good consequences for the total environment”. In this study, the former ethical foundation was implicit within the “prospective” LCA methodology that we used; whereby the proposed changes to vegetarian or vegan diets were assessed in terms of the environmental consequences of these actions relative to the status-quo, omnivorous Danish diet.

The fact that it is impossible to model the full consequences of an action in an LCA (or any model for that matter) has been noted as the most important limitation of prospective LCI methodology. Prospective LCA accounts only for simple causal relationships, whereas full outcomes depend on a variety of causal loops and delays. Often we do not know the significance of these excluded causal relationships or how well modeled outcomes accurately reflect reality (Ekvall et al., 2005). The issue at hand is that LCA practitioners endeavor to provide robust decision support, whilst being realistic about data and model weaknesses. How do we know if the results of an LCA are defensible and can agents make ethical decisions based on these results? As a rule, an LCA is meritorious if through judicious design, working within the limits of those aspects of external reality that can be known and modeled, and accounting for those aspects that are most salient to the model outcomes, it directs decision makers towards a reasonable facsimile of the outcomes of the modeled scenarios. The LCA in this paper was successful towards these ends, insofar that is transparent about model shortcomings while robustly identifying meat-free diets as viable alternative with superior environmental performance over the Danish status quo. Thus, people who equate actions that reduce impacts on the environment with ‘good’ actions would be justified in moving from omnivorous to meat-free diets in a Danish context.

Subjectivity in LCA modelling

This section departs from the previous in that these decisions do not relate to the choice of using LCA to assess environmental performance of diets, but those choices made by the LCA practitioner in developing their LCA that are based on the assessor’s or study comissioner’s values.

To start the user of the LCA must decide on the scope of the LCA, clarifying what aspects of the many facets of sustainability they will try to quantify. In the current study there was a pure focus on the environmental aspects, explicitly avoiding social and economic aspects of sustainability that could be addressed through the nascent social-LCA tool and the life-cycle costing methodologies, respectively. Making this choice implies either a low valuation of these sustainability aspects on the part of the LCA assessor or the belief that these issues are better handled within other assessment frameworks. Even if these sustainability aspects were assessed within their respective frameworks, there would still be an anthropocentric lean to the results due to the omission of issues related to animal welfare.

Although many strive to make LCA as objective, detailed, and scientifically robust as possible, it is well known that the use of LCA as a decision-making tool is not value free (Hellweg and Frischknecht, 2004 and Hertwich et al., 2000). For instance, the 25% error threshold employed here was based on the authors' professional judgment and experience that this is reasonable for the assessment, though a different threshold could have been used with nontrivial implications. The exclusion of post farm-gate impacts from the model, though grounded in previous findings, is a value-laden decision, whereby our focus on comparative performance implicitly eschews quantifying the complete environmental footprints of the diets (e.g. cradle to grave). The notion of absolute sustainability is likewise ignored, since the results cannot relate the food consumption of a typical Dane to the planet's seemingly limited ability to absorb impacts and continue operating in a manner amenable to human life. More crudely: we cannot determine whether the 'footprint' fits the 'shoe'. As such, this study adopts a weaker sustainability stance: We assume that acting to minimize current environmental harm is 'good', even in the absence of knowing whether this action is enough in an absolute sense. Ironically, one can end up employing ostensibly deep ecological principles through the egalitarian perspective to support opposing weak sustainability actions.

Moving from midpoint to endpoint in this study involved the acceptance of all of the weighting factors to aggregate to the three endpoint categories and their implicit assumptions, whilst the potential to move to a weighted single score, if taken, would have been imbued with values choices of the weighting factors and the belief that this is good scientific practice; decisions that are all loaded with implicit fundamental ethical and ideological judgments (Goedkoop et al., 2009). Hauschild (2005) notes that ethical values do not only come into play in the valuation step of LCA, but already in the definition of impact categories and how emissions are classified and characterized (e.g. toxicity in the individualist, hierarchist, and egalitrian perspectives in this study). This is most pronounced through the near ubiquity of carbon footprint as the preferred assessment in previous diet comparisons (see Section 'Introduction') focusing on climate change over other environmental challenges. One of the most important aspects of LCA, where societal and ethical values come into play, is the weighting of environmental impact potentials, as the weighting factor for an environmental impact reflects the importance of the impact category relative to the other environmental impact categories considered in the LCA (Hauschild and Barlaz, 2011). Accordingly, the determination of the weighting factors should therefore involve both an analysis of the causal relationships subject to the LCA as well as an analysis of the ethical values of the major stakeholders of the study who the LCA practitioner wishes to accept the result of the LCA. If the major stakeholders do not share the ethical values inherent in the weighting this can change the outcome of the LCA (Hauschild and Barlaz, 2011). This is especially important when one considers that LCAs can be funded by companies and industry groups (e.g. an association of a particular type of farmer) that might have a stake in presenting a certain outcome to the public, potentially leading to the weighting of selected midpoint IPs or the exclusion of others to achieve results that align with the aims of the funding entity.

LCA Aspect	Note
Implicit within LCA	
Use of LCA	Focus on environmental aspects of sustainability. Inclusion of social-LCA and life-cycle costing expand the scope of sustainability assessment, but still eschews discussions on animal welfare. Also implicitly believes that the behavior of nature is in many ways predictable and equates an environmentally preferable choice with the adoption of a set of technologies that provides a function with potentially lower burdens than other comparable sets of technologies.
Choices in LCA process	
Selection of IPs	Involves the valuation of available indicators and the prioritization of those included. Can be used to obfuscate poor performance of a product-system through the purposeful exclusion of those IPs where the system has negative performance. Funders of a study may influence this. Various levels of certainty and consensus in modelling methodology exist for different LCIA indicators. At the midpoint level only climate change, ozone depletion and particulate matter formation indicators are widely considered to be the most mature (Hauschild et al., 2013). Results in all other categories should be viewed with a higher level of skepticism and require significant divergence between assessed systems in those categories before conclusions regarding comparative performance should be drawn.
Use of cultural perspectives	Egalitarian, hierarchist and individualist perspectives prescribe to egalitarianism, utilitarianism and libertarianism, respectively. Can be used as a lens to deal with uncertainty in modelling by adopting precautionary principle or as a way to focus on short- to mid-term impacts.
Weighting of IPs	Moving to endpoint IPs or generating a single score (after normalization) both involve weighting which involves a subjective valuation of the importance of various IPs. Can be used to minimize the impacts of IPs and obfuscate poor LCA results for a system. Funders of a study may influence this.
Life cycle stages included	Choice of excluding life cycle stages (e.g. assessing from cradle to farm-gate) ignores full impacts and precludes any assessment of absolute sustainability

Table 6 - Overview of the ethical perspectives built within LCA and the subjective choices made while performing LCAs

The current case displays this clearly, as the toxicity and water scarcity index results remain ambiguous or even antagonistic to the general trend. One could easily reverse the conclusions of this study by employing single indicator methodologies focused on these IPs (à la carbon footprint) or through hefty weighting factors when moving towards a single indicator score. Relating this to the discussion of toxicity impacts in Section ‘Sensitivity of results to ReCiPe cultural perspectives’, the favored adoption of a global environmental indicator, such as carbon footprint, in comparing diets may be more appropriate considering the lack of spatial differentiation in LCA IPs. An adoption ofHeller et al.’s (2003) spatially disaggregated food product environmental assessment method may actually be better equipped to deal with other agricultural related IPs (erosion, eutrophication, etc.) than the traditional LCA tool. Moreover, the carbon footprint’s cynosure is also product of a larger environmental community's valuation of climate change as the defining environmental issue of our epoch, requiring amelioration on ethical grounds.

Robustness of LCA on diets

On the whole it would seem that using LCA as a method to environmentally assess diets (or anything) is fraught with uncertainty, value judgments, and even value judgments about uncertainty, begging the question: does LCA show that switching to lower animal product diets reduces environmental burdens? The fact that this assessment, along with earlier diet LCAs, all point in the same direction hints either that these models are all similarly flawed, or their conclusions are substantiated. The former is unlikely considering the methodological variability employed across the studies (system boundaries, LCIA models, consequential vs. attributional methods, databases utilized, etc.), which would have identified large flaws in competing methods through contradictory results. We thus accept that movements towards vegetarian or vegan diets generally constitute environmental 'goods', but only if one is disposed to value pristine environmental state.

Though this study has shown that LCA does have a role to play in assessing select aspects of the sustainability of diets, the discussion has shown that there remain a number of challenges in the application of LCA to this domain. Table 6 sums up the value choices implicit within the LCA methodology and the subjective choices made by an assessor while performing an LCA in hopes of providing the reader with the tools to critically interpret LCAs of diets.

Conclusions

Assessing diets from an environmental perspective is a complex task. Technical difficulties aside, the value systems embedded within assessment methods question the objectivity of such an endeavor, as evidenced by the normative values embedded within LCA, and the various ways these judgments influence model outcomes. Accepting that LCA can be used to predict environmental impacts, the assessment found that the results were robust against changing the 'cultural perspectives' allowed within the ReCiPe 2008 LCIA methodology, adding credence – along with earlier studies – to the idea that shifts from diets with high meat intake towards vegetarian or vegan diets generally predicts positive environmental outcomes, with the exception of water scarcity, which was influenced by the higher grain, fruit, and vegetable intake of these diets.

While our results support the general argument for reducing food-related environmental impacts through behavioral changes, difficulties in assessing toxicity impacts with LCA were noted. These require further methodological development or different assessment tools for those impacts – preferably at the local level – to account for the idiosyncrasies of receiving ecosystems (e.g. environmental risk assessment) or containing larger inventories of agrochemicals (e.g. USETox). Moreover, following vegetarian or vegan diets should not be conflated with sustainable lifestyles, since one can adhere to a low meat diet while causing negative environmental impacts in other aspects of life (e.g. commuting long distances by private vehicle, frequent air travel, large dwelling, etc.) that more than negate the positive environmental impacts of food choices. Dietary habits are only one of many areas where individuals can actively reduce their ecological burdens (Gjerris and Gaiani, 2014).

It should be elucidated that polemical dietary shifts that completely eliminate meat or dairy products are not necessary to induce positive environmental change. Animal husbandry methods that are well situated within ecological cycles can be positive for the environment. However, these remain the exception, since ecologically destructive factory farming is still the conventional approach (Cribb, 2010). Saxe et al.'s (2012) work showed that a more environmentally focused omnivorous diet in a Nordic context (reduced food miles, strategic organic content, reduced ruminant consumption) could potentially have similar environmental performance to a fully vegetarian diet. However, given organic agriculture's typically lower yields, a societal scale change to consuming primarily organic agriculture though positive in terms of fossil fuel reductions and toxicity attenuation, would consume more of the quintessential, constrained agricultural resource: land (Seufert et al., 2012). Notwithstanding, even shifting diets away from beef consumption would provide considerable environmental benefits (Nijdam et al., 2012).

LCA is limited insofar as it is an environmental assessment tool that ignores numerous other issues surrounding food consumption. The positive health impacts of vegetarian and vegan diets (Singh et al., 2010) have been neglected here for instance, though these effects may also result from generally healthier lifestyle choices amongst their proponents (more active, lower rates of smoking, etc.) and not solely the diets (Chang-Claude and Frentzel-Beyme, 1993). Furthermore, active lifestyles also have their own related environmental impacts (sports facilities, physiotherapy centers, etc.) that warrant consideration if a complete assessment of lifestyles is performed. Issues of animal welfare have also been ignored here, even though these could have significantly changed our comparison of the vegetarian and vegan diets, likely supporting a switch to a vegan diet despite their generally similar environmental performances. Despite these exclusions, the evidence of the environmental benefits of lower meat and dairy consumption continues to mount, not only in Denmark, but also in countries with similar food cultures.

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S.1 – Omnivorous Danish Diet

Food Product	Consumption at plate according to Danish Household Survey[i] (g/day)[a]	Apparent consumption according to Danish Statistics[ii] (g/day) [year][b]	Breakdown of consumption within food group[c]	Edible Losses[iii] (post farm)[d]	Demand at farm for 2000 kcal (g/day)[e]
Milk Products	356	-	-	8%	-
Milk	-	253.4 [2011]	86.0%	8%	278.4
Cream	-	28.5 [2011]	9.7%	8%	31.3
Crème fraiche	-	7.1 [2011]	2.4%	8%	7.8
Butter	-	5.5 [2011]	1.9%	8%	6.0
Cheese	31	-	-	8%	28.2
Eggs	16	-	-	8%	14.5
Meat	105	-	-	19%	-
Beef and Veal	-	77.0 [2011]	44.0%	19%	47.7
Offals of cattle	-	2.2 [2011]	1.3%	19%	1.4
Pork	-	87.4 [2011]	50.0%	19%	54.2
Offals of pigs	-	3.0 [2011]	1.7%	19%	1.9
Mutton	-	3.3 [2011]	1.9%	19%	2.0
Game	-	1.9 [2011]	1.1%	19%	1.2
Poultry	22	-	-	19%	22.7
Cereals	212	-	-	31%	-
Wheat flour	-	157.3 [2010]	55.1%	31%	141.6
Durum wheat	-	16.7 [2009]	5.9%	31%	15.1
Rye flour	-	37.5 [2010]	13.2%	31%	33.8
Oats	-	27.4 [2010]	9.6%	31%	24.7
Rice and rice flour	-	17.5 [2009]	6.1%	31%	15.8
Other flour and groats	-	26.6 [2010]	9.3%	31%	24.0
Potato flour	-	2.2 [2010]	0.8%	31%	2.0
Vegetables	153	-	-	26%	-
Potatoes	94	-	-	32%[f]	115.6
Cucumbers	-	23.6 [2006]	14.7%	26%	25.3
Pepper	-	0 [2011]	0.0%	26%	9.0
White cabbage	-	8.4 [2006]	5.2%	26%	9.0
Red cabbage	-	8.4 [2006]	5.2%	26%	0.6
Brussels sprouts	-	0.5 [2006]	0.3%	26%	11.8
Cauliflower and Broccoli	-	11.0 [2006]	6.8%	26%	11.8
Chinese cabbage	-	5.5 [2006]	3.4%	26%	6.8
Leeks	-	6.3 [2006]	3.9%	26%	6.8
Beetroots	-	4.7 [2006]	2.9%	32%[f]	5.4
Celeriac	-	2.2 [2006]	1.4%	32%[f]	2.6
Carrots	-	38.1 [2006]	23.7%	32%[f]	44.5
Onions	-	29.0 [2006]	18.1%	26%	31.2
Lettuce	-	23.3 [2006]	14.5%	26%	25.0
Fruits	280	-	-	26%	-
Tomatoes	-	84.1 [2006]	33.6%	26%	106.2
Cherries (sweet and sour)	-	5.2 [2006]	2.1%	26%	6.6
Strawberries	-	7.7 [2006]	3.1%	26%	9.7
Apples	-	134.5 [2006]	53.7%	26%	169.9
Pears	-	18.9 [2006]	7.5%	26%	23.9
Sugar	36	-	-	0%	30.1
Oils	34	-	-	8%	-
Margarine	-	-	-	8%	27.8
Margarine: Rapeseed Oil[g]	-	-	-	8%	16.1
Margarine: Sunflower Oil[g]	-	-	-	8%	1.5
Margarine: Maize Oil[g]	-	-	-	8%	1.5
Margarine: Palm Oil[g]	-	-	-	8%	5.9
Margarine: Palm Kernel Oil[g]	-	-	-	8%	5.9

Grey rows indicate that the food item was disaggregated into its constituent items which were then considered in the final consumption.

[a] Average Danish consumption to provide 10MJ energy per day, in the broad food groups defined and selected staples (e.g. potatoes)

[b] Taken from Danish Statistics for the most recent year available for every food item. Provided by source in kg consumed per capita per annum, and adjusted to grams/day by multiply by a factor of (1000/365).

[c] Taken is the mass of that food item divided by the sum of masses of all other food items within that food group. For example, 'milk' was taken as 253.4/(253.4+28.5+7.1+5.5)=86%.

[d] Taken as losses for the 'Processing', 'Distribution' and 'Consumption' for the food groups.

[e] Calculated as percentage of that food item in its' food group times amount consumed of that food group in first column. Adjusted for food losses with a factor of 1/(1-food losses). Adjusted to 2000 kcal/day with a factor of 2/(10/4.18).
[f] Taken as losses for 'Roots and Tubors'
[g] Breakdown of constituent oils taken from Nilsson et al. (2010)[iv]

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S.2 – Vegetarian and Vegan Diets

Food Product	Vegetarian recommended daily servings for 2000 kcal[i]	Vegan recommended daily servings for 2000 kcal [ii]	Mass per serving (g)[i]	Breakdown of food items within food group[a]	Edible losses (post farm)[ii]	Vegetarian demand at farm for 2000 kcal (g/day)[e]	Vegan demand at farm for 2000 kcal (g/day)
Dairy	2	-	-	-	8%	-	-
Milk	-	-	250	79.1/0%	8%	449.8	-
Cheese	-	-	42	20.9/0%	8%	19.1	-
Eggs	0.5	-	57	100/0%	19%	31.0	-
Whole Grains	6	7	-	-	33%	-	-
Bread[b]	-	-	30	16.6%	33%	44.8	52.2
Pasta[b]	-	-	100	16.6%	33%	149.3	174.1
Bun[b]	-	-	30	16.6%	33%	44.8	52.2
Breakfast Cereal[b]	-	-	30	16.6%	33%	44.8	52.2
Cracker[b]	-	-	30	16.6%	33%	44.8	52.2
Wheat: bread	-	-	-	-	33%	9.0	10.4
Wheat: pasta	-	-	-	-	33%	149.3	174.1
Wheat: bun	-	-	-	-	33%	31.3	36.6
Wheat: Breakfast cereal	-	-	-	-	33%	44.8	52.2
Wheat: cracker	-	-	-	-	33%	44.8	52.2
Rye: bread	-	-	-	-	33%	21.8	25.3
Rice	-	-	100	16.6%	33%	145.0	316.2
Vegetables	8	8	50	-	26%	-	-
Potatoes	-	-	-	40.5%	26%	238.4	238.4
Cucumbers	-	-	-	8.7%	26%	47.1	47.1
Pepper	-	-	-	0.0%	26%	0	0
Spring-white cabbage	-	-	-	3.1%	26%	16.7	16.7
Spring-red cabbage	-	-	-	3.1%	26%	16.7	16.7
Brussels sprouts	-	-	-	0.2%	26%	1.1	1.1
Cauliflower and broccoli	-	-	-	4.1%	26%	21.9	22.0
Chinese cabbage	-	-	-	2.0%	26%	11.0	11.0
Leeks	-	-	-	2.3%	26%	12.6	12.6
Beetroots	-	-	-	1.7%	32%	10.1	10.1
Celeriac	-	-	-	0.8%	32%	4.8	4.8
Carrots	-	-	-	14.1%	32%	82.8	82.8
Onions	-	-	-	10.7%	26%	58.1	58.1
Lettuce	-	-	-	8.6%	26%	46.6	46.6
Fruits	4	4	150	-	26%	-	-
Tomatoes	-	-	-	33.6%	26%	272.3	272.3
Cherries (sweet and sour)	-	-	-	2.1%	26%	16.9	16.9
Strawberries	-	-	-	3.1%	26%	24.8	24.8
Apples	-	-	-	53.7%	26%	435.5	435.5
Pears	-	-	-	7.5%	26%	61.2	61.2
Legumes and Soy	3	3	-		26%	-	-
Beans	-	-	100	33.3%	26%	135.1	135.1
Tofu	-	-	125	33.3%	26%	94.6	94.6
Soy beverage[c]	-	-	250	33.3%	26%	337.8	337.8
Soy beverage: soy beans	-	-	-	-	26%	23.6	23.6
Soy beverage: sugar cane	-	-	-	-	26%	8.5	8.5
Soy beverage: maize starch	-	-	-	-	26%	0.1	0.1
Nuts	1	2	30	-	26%	-	-
Peanuts	-	-	-	50%	26%	20.3	40.5
Cashews	-	-	-	50%	26%	20.3	40.5
Vegetable Oils	2	2	14	-	8%	-	-
Vegetable Oil	-	-	-	50%	8%	7.6	7.6
Palm Oil	-	-	-	-	8%	3.5	3.5
Soybean Oil	-	-	-	-	8%	2.4	2.4
Rapeseed Oil	-	-	-	-	8%	1.7	1.7
Margarine[d]	-	-	-	50%	8%	7.6	7.6
Margarine: rapeseed oil	-	-	-	-	8%	2.5	2.5
Margarine: sunflower oil	-	-	-	-	8%	0.2	0.2
Margarine: maize oil	-	-	-	-	8%	0.2	0.2
Margarine: palm oil	-	-	-	-	8%	0.9	0.9
Margarine: palm kernel oil	-	-	-	-	8%	0.9	0.9

Grey rows indicate that the food item was disaggregated into its' constituent items which were then considered in the final consumption.
[a] Splitting of the food groups was done using the same breakdown of foods consumed according to the Danish Statistics or evenly between foods within that food group where these statistics were lacking.
[b] Food items broken into constituent items using the LCA Food[iii] or assumed to be comprised only of wheat where a breakdown was lacking.
[c] Soy beverage disaggregated using Ercin et al. (2012)[iv]
[d] Breakdown of constituent oils for margarine taken from Nilsson et al. (2010)[v]
[e] Calculated as the total number of servings for that food group multiplied by that food items share of consumption in that food group and then multiplied by the factor accounting for food losses. For example, since 'Milk' accounts for 79.1% of the total dairy needs, it is calculated as 2 servings * 0.791 * 250 g/serving * 1/(1-0.08).

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ii FAO. (2011). *Global Food Losses and Food Waste - Extent, Causes and Prevention*. Rome, IT. Retrieved from http://www.fao.org/docrep/014/mb060e/mb060e00.pdf

iii LCA Food. (2007). LCA Food Database. Retrieved from http://lcafood.dk/

iv Ercin, a. E., Aldaya, M. M., & Hoekstra, A. Y. (2012). The water footprint of soy milk and soy burger and equivalent animal products. *Ecological Indicators*, *18*, 392–402. doi:10.1016/j.ecolind.2011.12.009

v Nilsson, K., Flysjö, A., Davis, J., Sim, S., Unger, N., & Bell, S. (2010). Comparative life cycle assessment of margarine and butter consumed in the UK, Germany and France. *The International Journal of Life Cycle Assessment*, *15*(9), 916–926. doi:10.1007/s11367-010-0220-3

S.3 Custom Process Inventories

Milk Production – Process inventory for milk production on the marginal Danish farm according to Food LCA[i]

Inputs	Amount	Units	Amount	Units
Spring Barley	91.9	t/a	0.18	kg/kg milk
Soy meal	77.2	t/a	0.15	kg/kg milk
Lubricant	1068	t/a	0.002	kg/kg milk
Fertilizer, calcium ammonium nitrate	6602	t N/a	0.01	kg/kg milk
Fertilizer P	909	t P/a	0.002	kg P/kg milk
Fertilizer K	2549	t K/a	0.005	kg K/kg milk
P, mineral feed[a]	137	t N/a	0.0003	kg N/kg milk
Electricity	42162	kWh/a	0.08	kWh/kg milk
Heating	690	MJ/a	0.001	MJ/kg milk
Traction	376043	MJ/a	0.75	MJ/kg milk
Land	65	ha	0.0001	ha/kg milk
Outputs				
Bread wheat	12.1	t/a	0.02	kg/kg milk
Rapeseed	1.1	t/a	0.002	kg/kg milk
Milk	499.3	t/a	1	kg
Beef meat[b]	20.6	t/a	0.01	kg/kg milk
Air Emissions				
Methane	12316	t/a	0.02	kg/kg milk
Ammonia	3426	t/a	0.007	kg/kg milk
N_2O	920	t/a	0.002	kg/kg milk
Emissions to Water				
Nitrate	31112	t/a	0.06	kg/kg milk
Phosphate	113	t/a	0.0007	kg/kg milk

Numbers have been rounded for legibility. The model also assumes that 1.12 kg of milk are required to produce 1 kg of milk at market according to LCA Food[i].
[a] Mineral feed assumed to consist of 40% dolomite and 60% zeolite by volume[ii].
[b] Slaughter weight of cows taken as the weighted average of sucklers reaching market from Nguyen et al. (2010)[iii]. Amount of avoided beef production avoided at market taken from Cederberg et al. (2003)[iv].

Cream Production – Dynamic market reactions to the production of cream[v].

Cream is a constrained by-product of milk.
The utility of cream is its fat content.
Consuming butter will means fat content in cream typically used for other products must be procured from elsewhere
25% of cream fat would have been used as fat content in powdered milk - this is substituted with marginal vegetable oil (palm oil)
75% of cream fat actually sees consumers switch from high fat to low fat cheese

Palm Oil substitution for fat content allocated to butter production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Cream	25%	Estimated
Palm Oil	100%	Estimated
	0.0625	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Cream	25%	Estimated
High Fat Cheese	35%	Estimated
Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	0.54	kg
Low Fat Cheese Produced	1.70	kg

Crème Fraiche Production – Dynamic market reactions to the production of crème fraiche.

Same market reactions as outlined for cream demand market.

Palm Oil substitution for fat content allocated to crème fraiche production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Crème Fraiche	40%	Estimated
Palm Oil	100%	Estimated
Palm Required	0.1	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Crème Fraiche	40%	Estimated
High Fat Cheese	35%	Estimated
Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	0.88	kg
Low Fat Cheese Produced	2.73	kg

Butter Production – Dynamic market reactions to the production of butter.

Same market reactions as outlined for cream demand market.

Palm Oil substitution for fat content allocated to butter production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Butter	81%	Estimated
Palm Oil	100%	Estimated
Palm Required	0.2	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Butter	81%	Estimated
High Fat Cheese	35%	Estimated
Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	1.74	kg
Low Fat Cheese Produced	5.52	kg

Cheese Production

Assumes only whey produced as a single byproduct. Values taken for soft cheese in a US context[vi].

Input	**Amount**	**Unit**
Milk	8.4	kg/kg dry. wt. cheese
Milk[a]	13.8	kg/kg cheese
Output		
Cheese	1	kg
Whey	12.8	kg

[a] Water content taken as 39%[vii]. Total milk taken as 1/(1-% wet wt.) Adjusted for 12% loss of milk at dairy.

Beef Production

Beef production taken from Nugyen et al. (2010)[iii].

Inputs	**Unit**	**per 1000 kg slaughter weight**	**per 1000 kg meat at market[a]**	**per kg meat at market**
Farm Supplied Feed				
Outdoor Grazing				
Grazed Grass	kg	9021	16174	16.17

Indoor Grazing				
Grass silage	kg	5446	9764	9.76
Maize silage	kg	2404	4310	4.31
Spring Barley	kg	2254	4041	4.04
Straw	kg	1726	3095	3.09
Imported Feed				
Soy meal	kg	12	22	0.02
Mineral Feed[b]	kg	131	235	0.24
Land Use				
Grass grazed (low)	ha a	3.01	5	0.005
Grass siliage (high)	ha a	0.68	1	0.001
Cereals	ha a	0.6	1	0.001
Fertilizer Import				
Nitrogen	kg	478	857	0.86
Phosphorous	kg	21.5	39	0.04
Direct on-farm energy use				
Electricity used in stables	MWh	1.07	2	0.0021
Electricity used in crop processing	MWh	0.64	1	0.001
Diesel	GJ	14	25	0.03
Transport				
Feed				
By ship	tkm	162	290	0.29
By Truck	tkm	12	22	0.02
Ouputs				
Gaseous Emissions				
N_2O	kg	26.2	47	0.06
CH_4				
Enteric fermentation	kg	417.6	749	0.75
Manure management	kg	58.5	105	0.10
NH_3	kg	95.6	171	0.17
Liquid Emissions				
NO_3	kg	123.1	221	0.22
PO_4	kg	2.7	5	0.005
Soil carbon loss	kg	145	260	0.26

Numbers have been rounded for ease of reading.
[a]Meat produced per slaughtered cow taken as weighted average of cattle reaching market from the article.
[b]Same assumptions as for mineral feed in the milk system.

Pork Production[viii]

Inputs	Amount	Units	Amount	Units
Feed				
Wheat	1090	kg	1.09	kg/kg pork
Barley	440	kg	0.44	kg/kg pork
Rye	161	kg	0.16	kg/kg pork
Soybean Meal	188	kg	0.19	kg/kg pork
Others	648	kg	0.65	kg/kg pork
Energy/Transport				
Heat (oil)	130.2	kWh	0.13	kWh/kg pork
Electricity	117.6	kWh	0.12	kWh/kg pork
Transport				
Ship	3375	tkm	3.38	tkm/kg pork
Truck 28t	868	tkm	0.87	tkm/kg pork
Traction	206	MJ	0.21	MJ/kg pork
Water	353	m^3	0.35	m^3/kg pork

Land	71	ha	0.0004	ha/kg pork
Outputs				
Air Emissions				
Methane	26.7	kg	0.03	kg/kg pork
N_2O	1	kg	0.001	kg/kg pork
NO_2	-2.4	kg	-0.002	kg/kg pork
Ammonia	20.7	kg	0.02	kg/kg pork
Water Emission				
NO_3	12	kg	0.01	kg/kg pork
PO_4	0.5	kg	0.0005	kg/kg pork
Avoided Fertilizer				
N	49	kg	0.05	kg/kg pork
P	13	kg	0.01	kg/kg pork
K	12	kg	0.01	kg/kg pork

Assumed that for every 120 kg of biomass produced, 94.7 kg of meat enters the market[viii]. Numbers have been rounded for ease of reading.

Cherry Production

Assumed cherry farmers supplying Denmark have similar technological level of development as Californian system. Values taken from Carlsson-Kanyama et al. (2000)[ix].

Inputs	**Amount**	**Units**	**Amount**	**Units**
Diesel	288	L	0.02	L/kg cherries
Gasoline	96	L	0.008	L/kg cherries
N-fertilizer	112	kg	0.009	kg/kg cherries
P-fertilizer	34	kg	0.003	kg/kg cherries
K-fertilizer	152	kg	0.01	kg/kg cherries
Land	1	ha	0.00008	ha/kg cherries
Outputs				
Cherries	12125	kg	1	kg

Numbers have been rounded for ease of reading.

Strawberry Production

Assumed strawberry farmers supplying Denmark have similar technological level of development as Californian system. Values taken from Carlsson-Kanyama et al. (2000)[x].

Inputs	**Amount**	**Units**
$CaNO_3$	0.02	kg/kg strawberries
KNO_3	0.03	kg/kg strawberries
NH_3PO_4	0.005	kg/kg strawberries
$MgSO_4$	0.003	kg/kg strawberries
Traction	102	MJ/kg strawberries
Irrigation	1	L/kg strawberries
Ouputs		
Strawberries	1	kg
N_2O (to air)	0.009	kg/kg strawberries

Numbers have been rounded for ease of reading.

Tofu

Assumes that 0.56 kg of soybeans are required for 1 kg of produced tofu (the balance being water and coagulant)[xi].

Soy Beverage[xi] – agricultural inputs that go into soy beverage manufacturing.

Inputs	**Amount**	**Units**
Soybean	0.07	kg/kg soy beverage
Sugar cane	0.03	kg/kg soy beverage
Maize starch	0.00003	kg/kg soy beverage
Outputs		
Soy Beverage	1	kg

Peanuts[xii]

Inputs	Amount	Units
Rye Seed	0.73	g/kg peanuts
Boron	0.17	g/kg peanuts
Lime/Gypsum	317	g/kg peanuts
Energy		
Pre-harvest fuel[a]	18	g/kg peanuts
Harvesting fuel[a]	26	g/kg peanuts
Electricity	0.06	kWh/kg peanuts
Outputs		
Peanuts	1	kg

[a] Assumed to be diesel.

Margarine[xiii] **-** agricultural inputs that go into margarine manufacturing.

Inputs	Amount	Units
Rapeseed oil	0.36	kg/kg margarine
Sunflower oil	0.03	kg/kg margarine
Maize oil	0.03	kg/kg margarine
Palm oil	0.13	kg/kg margarine
Palm kernel oil	0.13	kg/kg margarine
Outputs		
Margarine	1	kg

Eggs[xiv] **-** Based on LCA of organic eggs which likely have lower production efficiency per unit input, which may elevate the results, but only marginally when taken in the context of the diets. Waste by-products should be interpreted as the goods at the market that processed chicken waste would substitute, not products directly resulting from egg production.

Inputs	Amount	Units
Transport	0.2	kg/kg eggs
Wheat	1.37	kg/kg eggs
Rapeseed	0.20	kg/kg eggs
Soybean meal	0.29	kg/kg eggs
Barley	0.10	kg/kg eggs
Maize	0.31	kg/kg eggs
Soybeans	0.18	kg/kg eggs
Oats	0.11	kg/kg eggs
Protein Pea	0.07	kg/kg eggs
Limestone	0.20	kg/kg eggs
Water	5.79	L/kg eggs
Silage	0.09	kg/kg eggs
Straw	0.09	kg/kg eggs
Sand	0.04	kg/kg eggs
Electricity	0.32	kWh/kg eggs
Diesel	0.0054	kg/kg eggs
Gas	0.05	MJ/kg eggs
Outputs		
Eggs	1	kg/kg eggs
Fertilizer as N (manure by-product)	0.0082	kg/kg eggs
Barley (waste treatment by-product)	0.00084	kg/kg eggs
District Heat (waste treatment by-product)	0.0005	kg/kg eggs
Maize (waste treatment by-product)	0.041	kg/kg eggs
N_2O (to air)	0.00082	kg/kg eggs
Methane (to air)	0.0016	kg/kg eggs

Vegetable Oil Mix[xv] **-** Blend of the top 3 vegetable oils by production volume in 2014, accounting for over 2/3 of global production

Inputs	Amount	Units
Palm oil	0.47	kg/kg oil mix
Soybean oil	0.33	kg/kg oil mix
Rapeseed oil	0.2	kg/kg oil mix
Output		
Vegetable oil mix	1	kg

Cashews[xvi]

Inputs	Amount	Units
Land	$5.8*10^{-5}$	ha/kg cashews
Limestone	0.63	kg/kg cashews
Gypsum	0.029	kg/kg cashews
Copper	$9.6*10^{-6}$	kg/kg cashews
Manganese	$2.4*10^{-5}$	kg/kg cashews
Molybdenum	$1.2*10^{-6}$	kg/kg cashews
Zinc	$1.1*10^{-4}$	kg/kg cashews
Iron	$3.6*10^{-5}$	kg/kg cashews
Urea	0.20	kg/kg cashews
Phosphate	0.47	kg/kg cashews
KCl	0.05	kg/kg cashews
Glyphosate	0.0014	kg/kg cashews
Diesel	0.089	kg/kg cashews
Water	5.48	L/kg cashews
Outputs		
Cashews	1	kg/kg cashews
Wood	3.89	kg/kg cashews

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[xv] Martin Stevenson (2014). *Palm Oil Research*. Retrieved from http://www.palmoilresearch.org/statistics.html. Accessed: October 13, 2015

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