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Elinor Hallström obtained her Master´s in Nutrition from Stockholm University in 2010. This thesis analyzes the potential and limitation for diet change to contribute more sustainable food systems. An interdisciplinary approach is used which combines methods originating from the fields of environmental-, nutritional- and health- studies. The results show that dietary change can reduce greenhouse gas emissions and land use demand of the diet, and simultaneously improve the nutritional quality and health effects of the diet. The positive synergies suggest that dietary change can play an important role in reaching future environmental and health goals.

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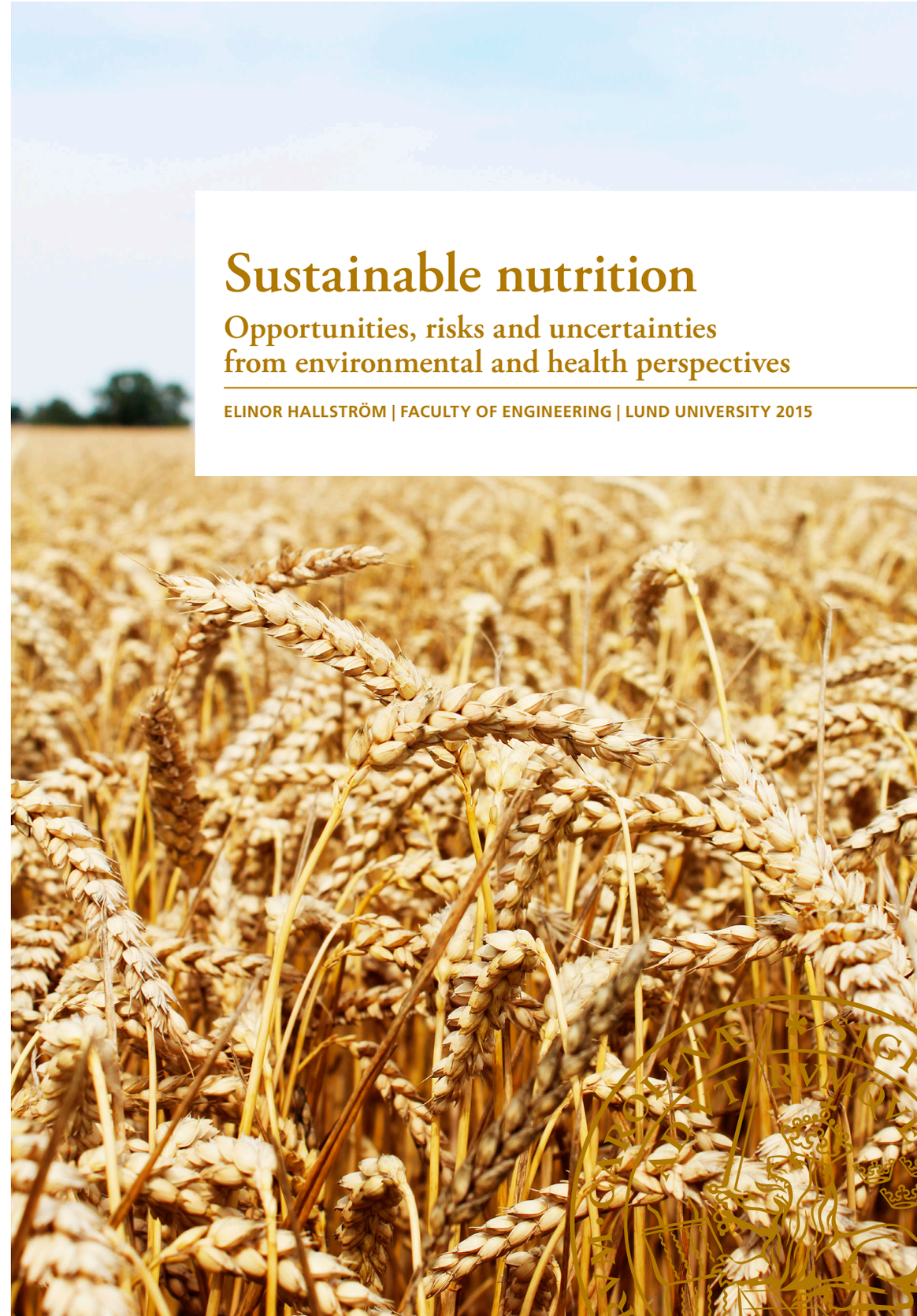

Sustainable nutrition

Opportunities, risks and uncertainties from environmental and health perspectives

ELINOR HALLSTRÖM | FACULTY OF ENGINEERING | LUND UNIVERSITY 2015

Sustainable nutrition

Opportunities, risks and uncertainties from environmental and health perspectives

Elinor Hallström

LUND UNIVERSITY

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Abstract

Food production and consumption are key drivers of environmental pressures and are essential factors in the promotion and maintenance of health. Production of food occupies more than one third of global land areas and is estimated to be responsible for some 30% of global greenhouse gas emissions. At the same time, we live in a world where nearly one billion people go hungry and even more people suffer from problems related to overweight and associated diet-related chronic diseases. This raises the question of the sustainability of current food systems and diets.

This thesis analyzes the potential and limitation for diet change to contribute more sustainable food systems. The results show that dietary change can reduce greenhouse gas emissions and land use demand of the diet, and simultaneously improve the nutritional quality and health effects of the diet. The positive synergies suggest that dietary change can play an important role in reaching future environmental and health goals.

Assessments of environmental- and health effects of food consumption and production are hampered by uncertainty and variability, and awareness of the limitations in the quality of data and methods is crucial. Transparent presentation of data and methods is necessary for a proper evaluation of the reliability and significance of the results. Improvement of data and further development of methods are required to further increase the quality of the assessments.

In this thesis, an interdisciplinary approach is used which combines methods originating from the fields of environmental-, nutritional- and health- studies. Life cycle assessment is used to quantify the greenhouse gas emissions and land use demand of food production, while the nutritional and health effects of food consumption are analyzed by using nutrient calculation and nutrition epidemiology. Integration of nutritional and health aspects into environmental assessments of food is an exciting development of the research field contributing to important new knowledge. To further broaden the perspectives and deepen the knowledge of sustainable food systems more aspects need to be covered.

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Sustainable nutrition:

Opportunities, risks and uncertainties from environmental and health perspectives

Elinor Hallström

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And most importantly my family; Ola, Mom & Dad, Anton & Claudia, Olof & Kim, Olivia & Lukas, for all your support and for loving me no matter what – I could not have done it without you!

Abstract

Food production and consumption are key drivers of environmental pressures and are essential factors in the promotion and maintenance of health. Production of food occupies more than one third of global land areas and is estimated to be responsible for some 30% of global greenhouse gas emissions. At the same time, we live in a world where nearly one billion people go hungry and even more people suffer from problems related to overweight and associated diet-related chronic diseases. This raises the question of the sustainability of current food systems and diets.

This thesis analyzes the potential and limitation for diet change to contribute more sustainable food systems. The results show that dietary change can reduce greenhouse gas emissions and land use demand of the diet, and simultaneously improve the nutritional quality and health effects of the diet. The positive synergies suggest that dietary change can play an important role in reaching future environmental and health goals.

Assessments of environmental- and health effects of food consumption and production are hampered by uncertainty and variability, and awareness of the limitations in the quality of data and methods is crucial. Transparent presentation of data and methods is necessary for a proper evaluation of the reliability and significance of the results. Improvement of data and further development of methods are required to further increase the quality of the assessments.

In this thesis, an interdisciplinary approach is used which combines methods originating from the fields of environmental-, nutritional- and health- studies. Life cycle assessment is used to quantify the greenhouse gas emissions and land use demand of food production, while the nutritional and health effects of food consumption are analyzed by using nutrient calculation and nutrition epidemiology. Integration of nutritional and health aspects into environmental assessments of food is an exciting development of the research field contributing to important new knowledge. To further broaden the perspectives and deepen the knowledge of sustainable food systems more aspects need to be covered.

Hållbara matvanor - från jord till bord

I Sverige äter vi i genomsnitt 800 kg mat per person och år. Vad och hur mycket vi väljer att stoppa i oss har stor betydelse både för planetens och vårt eget välmående. Den här avhandlingen handlar om hur våra val av livsmedel i den dagliga kosten påverkar miljön och hälsan.

Vilken typ av livsmedel vi väljer att äta och dessa producerats har stor inverkan på vår miljö och egen hälsa. Den globala livsmedelsproduktionen ockuperar drygt en tredjedel av världens markyta, står för runt 30% av den totala klimatpåverkan och är identifierad som ett av de största hoten mot vår miljö. Vi lever samtidigt i en värld där nästan en miljard människor går hungriga och ännu fler lider av ohälsa relaterad till övervikt och fetma. Mot denna bakgrund är det lätt att ifrågasätta hållbarheten i dagens livsmedelssystem.

Målet med den här avhandlingen är att bidra med ny kunskap om hur hållbarheten i dagens livsmedelsystem kan förbättras. Avhandlingen innehåller fyra artiklar som analyserar tre huvudsakliga frågor:

I. Hur stor är potentialen att minska klimatpåverkan och markbehovet från dieten genom förändrade matvanor?
II. Finns det några synergier och/eller konflikter mellan matvanor som tros främja en god hälsa och bidra till lägre miljöpåverkan?
III. Vilka metodaspekter är viktiga att beakta då både miljö- och hälsoaspekter av livsmedel och dieter ska bedömas?

Analysen av den första frågan visar att förändrade matvanor har en stor potential att minska klimatpåverkan och markbehovet från dieten. Dagens kunskapsläge visar att den enskilde individen kan minska sin diets klimatpåverkan och markbehov med hälften genom att förändra sina matvanor. Potentialen verkar främst bero på hur mycket och vilken typ av kött kosten innehåller men också på vilken typ av livsmedel som ersätter den minskade köttkonsumtionen.

Som svar på den andra frågan finns flera positiva synergieffekter mellan de matvanor som bedöms gynna både hälsan och miljön. Grönsaker, frukt, baljväxter och fullkornsprodukter, som rekommenderas utgöra en stor andel av kosten för en god hälsa har ofta även en relativt låg miljöpåverkan. Resultaten visar att förändrade matvanor som minskar dietens miljöpåverkan även kan bidra till förbättrad folkhälsa, bland annat genom minskad risk för hjärt-kärlsjukdom, diabetes typ II och tjocktarmscancer. Matvanor med lägre miljöpåverkan kan

emellertid innebära en risk för minskat intag av vissa näringsämnen, i synnerhet järn och zink.

Analysen av den tredje frågan visar att val av data, metod och antaganden kan ha stor inverkan på beräkningar av dietens miljö- och hälsoeffekter. Resultaten visar att medvetenhet om variation och osäkerhet i data och metoder kan vara avgörande för en korrekt användning och tolkning av resultaten. Statistik för hur mycket kött vi äter kan, exempelvis, redovisa dubbelt så hög konsumtion om den beskriver den tillgängliga mängden köttråvara inklusive ben jämfört med om den beskriver den tillagade mängden uppätet kött. Då både miljö- och hälsoeffekter från vårt matsystem ska analyseras krävs därmed att man är observant så att rätt data används vid rätt beräkningar.

I avhandlingen har olika analysmetoder kombinerats. För att beräkna dietens miljöpåverkan har en metod som kallas livscykelanalys används. Dietens hälsopåverkan har bedömts genom att beräkna livsmedels och dieters näringsinnehåll och med hjälp av nutritionsepidemiologi, en metod som används för att analysera hur intag av olika livsmedel och dieter påverkar risken att bli sjuk.

List of publications

I. **Hallström, E.**, Börjesson, P. (2013). Meat consumption statistics: reliability and discrepancy. *Sustainability: Science, Practice, & Policy 9 (2): 37-47.*

II. **Hallström, E.**, Röös, E., Börjesson, P. (2014). Sustainable meat consumption: a quantitative analysis of nutritional intake, greenhouse gas emissions and land use from a Swedish perspective. *Food Policy 47, 81-90.*

III. **Hallström, E.,** Gee, Q., Scarborough, P., Cleveland, D. (2015). The potential for diet change to reduce greenhouse gas emissions from the food and health care systems in the US. *Submitted.*

IV. **Hallström, E.**, C-Kanyama, A., Börjesson, P. (2015). Environmental impact of dietary change: A systematic review. *Journal of Cleaner Production 91, 1-11.*

Author´s contribution

I. I was responsible for the study design, analysis and writing of the paper under supervision of Pål Börjesson.

II. I contributed to the study design, analysis and writing of the paper and was responsible for the quantification and analysis of nutritional intake. Elin Röös was responsible for the quantification of greenhouse gas emissions and land use demand as well as the Monte Carlo analysis. Valuable input was received from Pål Börjesson.

III. I created the dietary scenarios, assembled the relative risk data and was responsible for calculating the relative risk of non-communicable diseases and GHG emissions from the food sector. I also contributed to the study design and writing of the paper. Quentin Gee assembled and analyzed the health care costs and GHG emissions in the health care sector, and was responsible for the Monte Carlo analysis. Peter Scarborough contributed to the analysis of the relative risk of non-communicable diseases and David Cleveland was responsible for the framing of the paper and reviewing of methods and analyses.

IV. I was responsible for the study design, analysis and writing of the paper under supervision of Annika C-Kanyama and Pål Börjesson.

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1 Background

Food consumption has been identified as one of the most important drivers of environmental pressures (European Commission, 2006) and is an essential factor in the promotion and maintenance of health (WHO/FAO, 2003). This chapter provides a background of the impact of food production and consumption on climate (1.1), land use (1.2), nutrition and health (1.3), the main aspects that form the basis for the research in this thesis. The chapter ends with a section describing a holistic view of the food sector (1.4) which integrates the environmental and health perspective.

1.1 Climate impact of the food sector

Global warming is a result of rising concentrations of greenhouse gases (GHGs) in the atmosphere, mainly originating from human activities. The main GHGs emitted by humans are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Since the mid-1800s, and especially during the last fifty years, global anthropogenic emissions of GHGs have increased markedly. This development has resulted in a 60% increase of GHG concentrations in the atmosphere and a 0.85°C increase of the mean temperature on earth (IPCC, 2014), with large differences in temperature change by region (Figure 1).

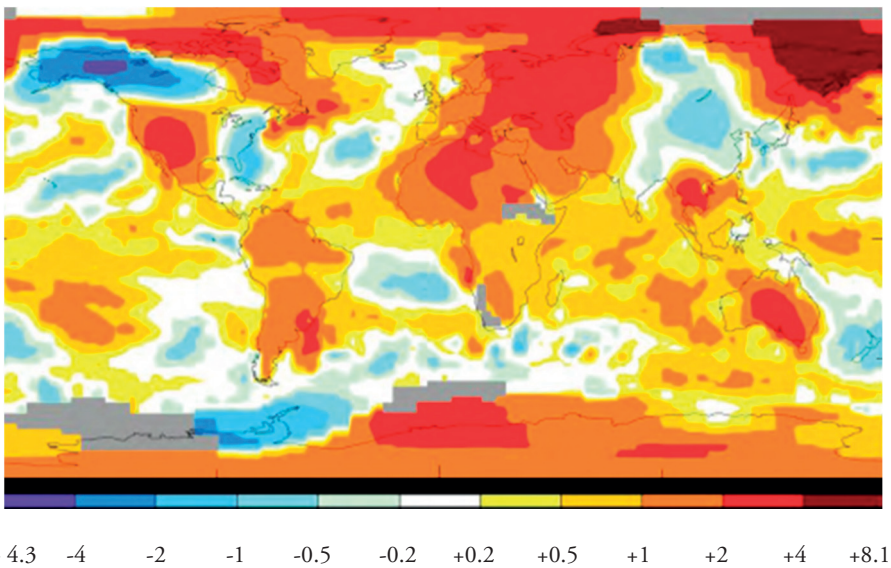

Figure 1. Surface temperature, anomaly versus 1951-1980 (°C, 2012). Adapted from (FAO, 2013a).

Greenhouse gas emissions are produced at all stages of the food chain. Approximately 80-85% of the GHG emissions embodied in the food system are attributable to direct and indirect emissions from the agricultural production (Vermeulen et al., 2012). Major sources of direct GHG emissions in agriculture are nitrous oxide emissions from soils and methane emissions from enteric fermentation, while other sources are the burning of biomass, rice production and manure management. Agriculture, being a main driver of deforestation (Tubiello et al., 2015) is also responsible for indirect emissions of carbon dioxide

from land use change. The journey of food beyond the farm gate via manufacturing, distribution, refrigeration, preparation in the home and waste disposal is dominated by carbon dioxide emissions from fossil fuel use. Including all stages of the food chain, the food sector is estimated to account for 19-29% of total global anthropogenic GHG emissions (Figure 2) (Vermeulen et al., 2012).

The estimates for global GHG emissions from the agrifood sector depend on the quantitative method used and how the boundaries of the system are defined. According to the IPCC agriculture and agriculture-driven land use change contribute 24 % of global GHG emissions (IPCC, 2014). However, this estimate does not include post-agricultural emissions, and thus exclude emissions from, for example, post-harvest processing, transportation and preparation of food. By including emissions from the whole food chain, the agrifood system is estimated to account for about 30% of global anthropogenic GHG emissions (FAO, 2013a; Garnett, 2011).

Due to the high proportion of emissions from biological processes in agriculture, the technical potential of reducing GHG emissions from the agricultural sector, and especially the livestock sector, is limited compared to many other sectors (EC, 2011; SEPA, 2012). The potential to reduce current GHG emissions from agriculture, through optimized nutrient use, improved manure management and reduced carbon intensity, has been estimated to be 15-40% (Hedenus et al., 2014; Popp et al., 2010; Weidema et al., 2008). To further reduce GHG emissions from the agrifood sector, demand-side strategies including reduced food waste and dietary change are believed to be necessary (Bajželj et al., 2014; Garnett, 2011; Hedenus et al., 2014).

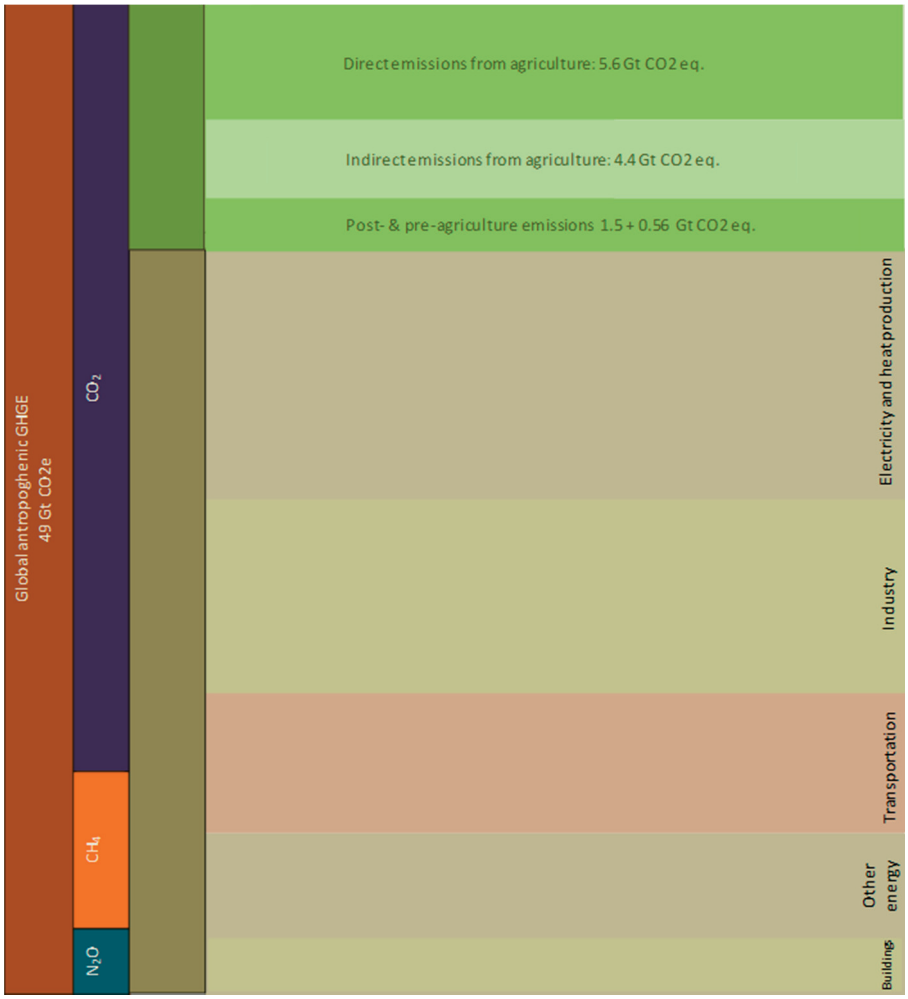

Figure 2. Global anthropogenic GHG emissions per gas and sector. Green areas represent GHG emissions from food production and consumption. Data based on (Vermeulen et al., 2012)

The amount of GHG generated in food production differs widely between and within food categories. In general, animal-based food gives rise to higher levels of GHG emissions per kg food produced compared to plant-based foods (Drewnowski et al., 2015). Ruminant meat (e.g. beef, lamb) is especially GHG intensive due to the methane emissions from enteric fermentation and the lower feed conversion rate of ruminants compared to monogastric animals (e.g. pork, poultry) (Nijdam et al., 2012). Variations in GHG emission also exist within food categories and specific foods due to differences in production systems and

mode of transportation. Fruits and vegetables produced in greenhouses heated with fossil fuels and/or transported by air, for example, generate larger emissions compared to outdoor cultivation and local production (Hoolohan et al., 2013; Sonesson et al., 2010). Dietary change offers the potential for climate mitigation, as the type and amount of food consumed can result in a wide range of emissions.

Climate change is anticipated to cause large and potentially dangerous risks for both human and natural systems. Without additional efforts to constrain GHG emissions, a further global mean temperature rise of the magnitude of 1.4 - 4.8 °C is projected by the end of the 21st century (IPCC, 2014). In order to avoid or limit dangerous effects from climate change a maximum temperature rise of 1.5-2°C relative to pre-industrial temperatures has been suggested (Randalls, 2010). To meet this target, current global GHG emissions need to be cut by more than half by 2050 (UNEP, 2013a). In a long-term perspective, this in turn requires that global average emissions of GHGs stabilize at a level in the range of one to two tons of CO_2 eq. capita^{-1} year^{-1} (EC, 2007; UNEP, 2010). This will require substantial mitigation efforts on all fronts, not least in the food sector.

1.2 Land use demand of the food sector

The agricultural sector, including crops, livestock, forests, fisheries and aquaculture, is the main human activity responsible for natural resource management (FAO, 2013a). Already in the eighteenth century Thomas R Malthus expressed his concern about the ability to feed the growing population within the global resource limits (Malthus, 1798). How to use and distribute finite natural resources without overstepping the earth´s carrying capacity or planetary boundaries is an issue that has been on the agenda in sustainability discussions ever since (Daily, 1996; Meadows, 1974; Rockström et al., 2009, Steffen et al., 2015).

Agricultural land covers somewhat over a third of the available land globally (38%, 4900 Mha). Most agricultural land consists of permanent meadows and pastures (68%, 3400 Mha) and the remaining agricultural land consists of cultivated and temporarily fallow land areas (32%, 1600 Mha) (FAO, 2015). Data from the literature suggest one third of global cultivated land is used for feed production and almost two thirds for crops dedicated to direct human consumption, i.e. plant-based crops that are not used as feed but consumed by humans directly (Figure 3) (Hallström et al., 2011).

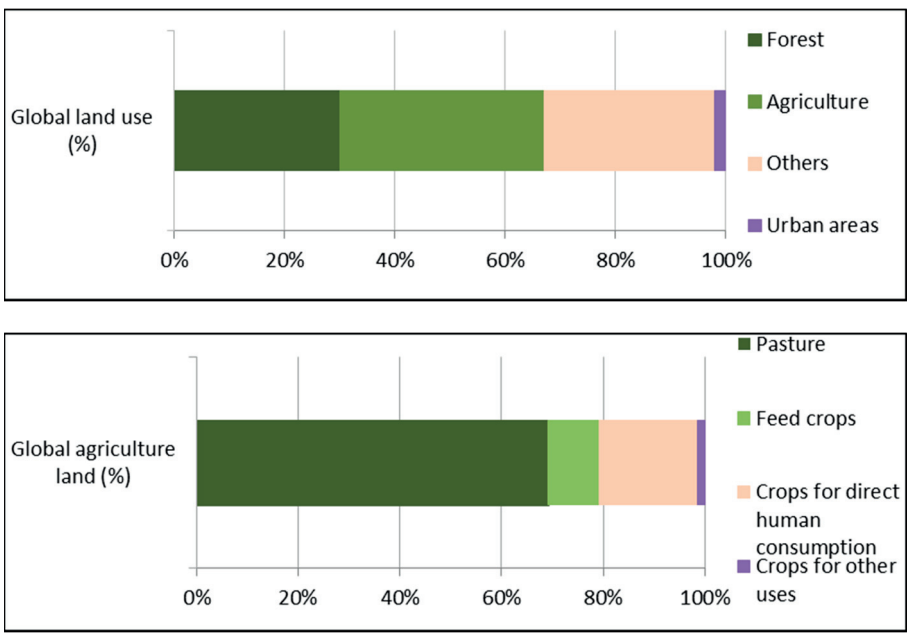

Figure 3. Distribution of use of land and agricultural land, globally.

Demand for agricultural land can simplistically be described as a function of the demand of agriproducts and the productivity of the agricultural land. The efficiency of producing plant-based products depends on the yield and cropping intensity. For animal-based products, the land use demand depends on the type of feed input and the feed conversion efficiency of the animals. Ruminant meat generally requires more land than monogastric meat, due to the lower feed conversion efficiency of these animals (de Vries & de Boer, 2010). On the other hand, meat from ruminants depend to a larger extent on perennial forage crops and grazing land which generally have less negative impact on the environment, and put less pressure on cultivated land with the potential to produce food for direct human consumption. Grazing animals can also contribute to increased biodiversity by keeping landscapes open (Gerber et al., 2013). One kg of edible meat from monogastric animals and ruminants requires about four kg and between eight and thirty-five kg of feed, respectively, depending on the type of animal and feed input (Peters et al., 2014; Wilkinson, 2011). Globally, livestock production uses about 75% of existing agricultural areas (Foley et al., 2011).

Over the past 50 years, the world population has more than doubled from 3.3 to 7.3 billion people (UN, 2015). Within the same time period, global food production has almost tripled (Smith et al., 2010). The great increase in food production was made possible mainly through technology development in the agricultural sector, including an increased use of chemical fertilizers, pesticides, high yielding cultivars, mechanization and irrigation (Bruinsma, 2003; 2009; Foley et al., 2005). This development, often referred to as the "green revolution", enabled substantially increased harvests from the same unit of land area. Additional gains in food production were realized by expanding agricultural areas which globally increased by some 470 Mha (40% cropland, 60% pastures and meadows) since 1965 (FAO, 2015). The agricultural intensification and expansion have been a mixed blessing being accompanied by large environmental pressures, including water and air pollution, soil degradation and loss of biodiversity (Foley et al., 2005).

While in some parts of the world there is still room for further expansion of agricultural land and increase in agricultural productivity, the possibilities elsewhere are more limited and/or associated with major ecological consequences (Foley et al., 2011; Godfray et al., 2010). A considerable potential for further increase in productivity is estimated by enabling farmers to narrow the gap between current and attainable yields in less developed regions (van Ittersum et al., 2013). To not repeat the mistakes made historically, the concept of "sustainable intensification" has emerged as a strategy to increase agricultural production with less negative impact on the environment (Godfray et al., 2014; Tilman et al., 2011). In contrast to more ideologically driven strategies, the

concept of sustainable intensification advocates for a mix of good practices originating from, for example, conventional and organic farming, agro ecology and biotechnology (Godfray et al., 2014).

By taking into account the three pillars of sustainability, sustainable agriculture should be economically viable, ecologically sound and socially just. While the farming methods best suited for more sustainable agricultural practices are highly debated and vary depending on the specific regional conditions, there is an increasingly common understanding that changes in production must be achieved in combination with strategies to reduce waste along the food chain and changing dietary patterns (Garnett, 2011; Godfray et al., 2014; Heller et al., 2013; Tilman & Clark, 2014).

In the coming four decades the world population is projected to increase by another 2.5 billion (UN, 2013). Trends of changes in dietary preferences and increased utilization of non-food crops for biofuels and other bio-based materials, put additional pressure on the future demand of agricultural products. In addition, changes in the climate and soil degradation are anticipated to result in a global decline of crop yields (IPCC, 2014), indicating a possible "peak" of maximum global food production in the near future (Brandão et al., 2010). At the same time, projections indicate that global agricultural production (in mass) will have to increase by 70-100% by mid-century in order to meet the rising demand for food (FAO, 2009; Tilman et al., 2011). The challenge ahead lies in meeting the rising demand for agricultural products with minimal negative consequences for the environment, and minimizing the risk for land use competition and conflicts.

1.3 Diet, nutrition and health

Health is defined as a state of complete, physical, mental and social well-being (WHO, 1948). This thesis analyzes the nutritional aspects of food consumption, one of several parameters influencing health. Food and nutrition have a direct impact on physiological and mental functions but is also a major factor influencing the long-term risk of developing several non-communicable diseases and life expectancy (Katz & Meller, 2014; WHO, 2011).

It is estimated that currently 800 million, or one in nine, people in the world do not get sufficient food to maintain health. Another two billion people suffer from the "hidden hunger" of micro-nutrient deficiencies (WHO/FAO, 2014). Even in high income countries, deficiency of various micro-nutrients is common particularly in certain groups of the population with special requirements. Inadequate intake levels have, for example, been identified in Europe for vitamin C, D, B_{12}, folate, I, Mg, Ca, Se, Zn, and Fe for women in childbearing age (Mensink et al., 2013; Roman Vinas et al., 2011). Even though, global hunger and nutrient deficiency are still dominant problems globally, the focus of nutrition policy has shifted to increasingly address the growing health issues related to over-consumption of food (NCM, 2014).

Over the past several decades major shifts in dietary patterns have occurred throughout the world. Economic growth, urbanization and greater access to cheap energy-dense food has driven a nutritional transition in high-, middle- and low-income countries from diets primarily based on carbohydrate-rich staple food towards a diet rich in energy-dense food such as vegetable oils, animal products and sugar-sweetened beverages (Imamura et al. 2015; Kearney, 2010; Popkin et al., 2012). The observed changes in diet have resulted in food consumption patterns characterized by higher intake levels of energy, added sugar, salt and saturated fat which are known risk factors for several chronic diseases, including cardiovascular disease, cancer and diabetes (WHO, 2009). In combination with the trend towards more sedentary lifestyles, the nutrition transition has resulted in a global epidemic of overweight and obesity, affecting some 1.9 billion adults, or more than a quarter of the global population (Finucane et al., 2011; WHO, 2012; 2015).

Nutrition-related health problems can thus be due to inadequate, unbalanced or excessive food consumption. A healthy diet, which provides energy and nutrients for long-term health, is characterized by a varied intake of micronutrient- and fiber dense foods, such as vegetables, legumes, whole grains, fruits, berries, nuts, seeds, and seafood (no fiber), a restricted intake of refined grains and sugars (e.g. cakes, soda and candy), salt, red meat (e.g. pork, beef, and lamb), and processed

meat products (e.g. bacon, salami, sausages, hot dogs). Furthermore, the body´s energy balance, i.e. the balance of calories consumed and calories used, is an important rule of thumb for healthy food habits (NCM, 2014; WCRF/AICR, 2007; WHO/FAO, 2003).

Table 1 illustrates dietary changes, commonly, recommended for meeting the guidelines for a healthy diet, in regions with unrestricted (e.g. "western") diets.

Table 1 Recommendations for healthy diet. Adapted from (NCM, 2014).

INCREASE	REPLACE			LIMIT
Vegetables Legumes	Refined grains	→	Whole grains	Processed meat Red meat
Fruit and berries	Butter	→	Plant-based oils, Oil-based fats	Sugar sweetened foods and beverages
Fish and seafood Nuts and seeds	High fat dairy products	→	Low-fat dairy products	Salt Alcohol

With between two and three billion people suffering from either under- or over-nutrition, malnutrition is estimated to be the largest single contributor to disease in the world (WFP, 2015). Over the past two decades, the prevalence of under-nourishment decreased from 19 to 11% globally and from 23 to 14% in developing countries (FAO, 2014). While progress has been made to reduce global hunger and under-nourishment, health problems associated with over-consumption of food are growing rapidly. Between 1980 and 2014, the global prevalence of obesity more than doubled (WHO, 2015). Projections for the future estimate that another billion people will become overweight or obese by 2030, if current trends continue (Kelly et al., 2008). This development is alarming as overweight is a large risk factor for many chronic diseases such as cardiovascular disease, diabetes and cancer (WHO, 2009). The implications on society of this development are and will be widespread, not least by the effects on human productivity and health care expenditures (Wang et al., 2011). Improving dietary patterns to curb the epidemic of "wellness diseases" around the world is undoubtedly one of the main challenges for future sustainable food systems.

1.4 The holistic perspective of the food sector

By analyzing the food system in a holistic perspective it becomes apparent that aspects from the environmental, nutritional and health perspectives interrelate and interact at multiple levels. The origin of food and how it is produced influences the nutritional content and the physiological effects of food consumption, just as the quantity and choice of food eaten will affect the demand for resources and environmental impact. In addition, use of resources, inputs and environmental impact from the agrifood system (e.g. fresh water, antibiotics, air pollution and pesticides) has implications for our health (e.g. fresh water shortage, antibiotic resistance, the occurrence of asthma, cancer, autoimmune diseases etc.), for current and future food and energy security (e.g. drought, extreme weather events, shortage of energy, phosphorus, healthy soils, arable land, and the loss of biodiversity, etc.).

Although multidimensional frameworks to study interactions between environmental and nutritional aspects have existed since the 1970s (JHEN, 2015; JNEFR, 2015; Schneider & Hoffmann, 2011), these aspects have traditionally been treated as two separate fields. However, as awareness of the environmental impact of food and diet has increased, the fields have gradually been more integrated in both the research and policy fields (Haines et al., 2009; Heller et al., 2013; Millward & Garnett, 2010).

Guidelines and recommendations for healthy food habits have been found to align well with suggested measures for reducing the environmental impact of food consumed. Identified synergies include the recommendations for a reduced intake of red and processed meat, increased intake of vegetables, fruits and whole grain and balancing the overall energy intake (Ciati & Ruini, 2012; Macdiarmid, 2013; Reynolds et al., 2014). These synergies suggest that the adoption of healthy diets could offer multiple benefits, including improved public health and reduced environmental and ecological impacts.

The increased recognition of the inter-relationships between the environmental and health effects of food has contributed to a trend in which environmental considerations are increasingly considered in the field of nutrition, for example, in the development of dietary guidelines and recommendations as well as food policies. Dietary recommendations and guidelines which integrate environmental considerations, for example, exist in the Nordic countries (NCM, 2014), in the Netherlands (HCN, 2011) and are planned to be implemented in the US (USDA, 2015).

Likewise, the importance of including the nutritional perspective in environmental assessments is acknowledged (Röös et al. 2015; Van Kernebeek et al., 2014) and there is an expressed need for development of methods for the incorporation of these aspects in the analysis of the environmental impact of food and diets (Heller et al., 2013; Schneider & Hoffmann, 2011). Within the policy area a discussion has emerged on which instruments can contribute to the development of a combined healthy and environmentally sustainable food system (Buttriss, 2009; Edjabou & Smed, 2013; Reisch, 2011).

2 Introduction

2.1 Outline and overview of the thesis

This thesis contains 10 chapters, which are shortly described below. Table 2 provides a summary and overview of the main objectives, methods and results of the thesis.

Chapter 1 provides a background of the main aspects that form the basis of the thesis. The chapter covers climate impact of the food system (1.1); land use demand of the food system (1.2); diet, nutrition and health (1.3), and the holistic perspective of the food system (1.4).

Chapter 2 provides an introduction and overview of the thesis, including the following: this outline and summary of the thesis (2.1); presentation of research goals and objectives (2.2), clarification of central concepts (2.3); and description and motivation of papers I-IV (2.4).

Chapter 3 describes the methodological background of the thesis. The chapter includes; an overview and summary of the theoretical framework (3.1); a description of the research theory (3.2); and the scope and boundaries of the thesis (3.3).

Chapter 4 provides a general description of sustainability indicators, metrics and targets and the methodological approach used in this thesis. The chapter includes, an overview and summary of the chapter (4.1); metrics to measure greenhouse gas emissions and climate impact (4.2), land use and land use change (4.3) and nutritional quality and health (4.4) of food consumption and production.

Chapter 5 presents the research methodology of the thesis. The chapter includes, an overview of the methods used (5.1); and a more thorough description of review-analysis (5.2), scenario analysis (5.3), life cycle assessment (5.4), nutrient calculation (5.5), nutrition epidemiology (5.6) and Monte Carlo analysis (5.7).

Chapter 6 describes the results of the thesis, and includes, the main findings of paper I (6.1), paper II (6.2), paper III (6.3), and paper IV (6.4).

Chapter 7 discusses the implications of the results of the thesis in relation to the overall research goal and questions. The chapter includes, an overview of main uncertainties in of the thesis (7.1), and further discusses the potential for diet change to improve sustainability of food systems (7.2), synergies and conflicts between healthier diets and diets with lower environmental burden (7.3), and integration of health and environmental methodology (7.4).

Chapter 8 presents a future outlook that puts the thesis in a wider perspective. The chapter includes, a summary of identified research gaps (8.1), and a description of six areas in the field that require further exploration (8.2-8.7).

Chapter 9 provides the concluding remarks of the thesis.

Chapter 10 provides the references used in the thesis.

Papers I-IV are provided in the end of the thesis.

Table 2. Overview of objectives, methods and results of paper I and II.

	OBJECTIVES (Chapter 2)	**METHODS** (Chapter 5)	**RESULTS** (Chapter 6)
PAPER I	Identify discrepancies between methods for producing meat consumption statistics, and identify the method and data best suited for different subsequent assessments.	Review analysis (5.2)	Three main methods exist to produce meat consumption statistics. Consumption data can either be derived from statistics of agricultural supply, household budget surveys, or individual dietary surveys. The definition of meat consumption may vary depending on the method and source. Thus, awareness of the method used and the implications thereof is essential for a correct use of the data (6.1.1).
	Identify main uncertainties in meat consumption statistics produced with different methods.		The most important uncertainty factors in meat consumption statistics include; assumptions on bone weight; food losses and waste; weight loss during cooking; and content of meat in processed and mixed meat products and prepared meals (6.1.2).
	Analyze and estimate the individual and combined impact of identified uncertainty factors for subsequent assessments of environmental and health impacts of meat consumption.		Factors contributing to discrepancies in meat consumption statistics may individually affect the data by 15-50%. Per capita meat consumption statistics may differ by a factor of two depending on the data used. Misuse of data is an obvious risk when consumption statistics are used for subsequent calculations of environmental and health effects (6.1.3).
	Suggest improvements in the methodology for producing, presenting and handling meat consumption statistics.		Improved transparency in information pertaining to methods and assumption in the generation of meat consumption would be helpful (6.1.4).

	OBJECTIVES (Chapter 2)	**METHODS** (Chapter 5)	**RESULTS** (Chapter 6)
PAPER II	Estimate the amount of meat consumed in Sweden and evaluate whether current consumption patterns follow health recommendations.	Scenario analysis (5.3), Life cycle assessment (5.4), Nutrient calculation (5.5), Monte Carlo analysis (5.7)	Swedish average meat consumption is estimated to be 170 g of boneless, uncooked meat per day, of which almost one quarter consists of charcuteries. To meet health recommendations, current Swedish meat consumption must be reduced by approximately 25%, and consumption of charcuteries by as much as possible. (6.2.1)
	Estimate the nutritional effects of adopting healthier scenarios of meat consumption in Sweden.		Healthier meat consumption in Sweden would reduce the average intake of energy from 14 to 7%, total fat from 34 to 10%, saturated fat from 54% to 11%, protein from 54% to 40%, zinc from 65% to 34%, and iron from 32% to 14% of the recommended or maximum recommended daily intake levels capita^{-1} day^{-1} (nutrient intake from meat only) (6.2.2).
	Estimate the effect on GHG emissions of adopting healthier scenarios of meat consumption in Sweden.		Healthier meat consumption in Sweden would reduce average GHG emissions from 0.6 to 0.2-0.4 tons of CO_2eq capita^{-1}year^{-1} (GHG emissions from meat only) (6.2.3).
	Estimate the effect on land use demand of adopting healthier scenarios of meat consumption in Sweden.		Healthier meat consumption in Sweden would reduce average land use demand from 0.11 to 0.04-0.07 ha capita^{-1}year^{-1} (land use demand for meat production only) (6.2.4).

	OBJECTIVES (Chapter 2)	**METHODS** (Chapter 5)	**RESULTS** (Chapter 6)
PAPER III	Estimate the amount of food consumed in the US and evaluate whether current consumption patterns follow health recommendations.	Scenario analysis (5.3), Life cycle assessment (5.4), Nutrient Calculation (5.5), Nutrition epidemiology (5.6), Monte Carlo analysis (5.7)	Average US consumption of red and processed meat, vegetables, fruit and grains are estimated to be about 90 g, 200, 135, and 165 g capita^{-1} day^{-1} (cooked weight). To meet health recommendations, current intake of red and processed meat must be reduced by approximately 45%, fruit and vegetable consumption should double, and grain consumption be reduced by about 20%, while the proportion of whole grains should be increased substantially (6.3.1).
	Quantify the effect on relative risk of disease and health care costs of adopting scenarios of healthier diets in the US.		Healthier diets in the US would reduce the RR of coronary heart disease, diabetes type II and colorectal cancer by 20-45%, and combined health care costs by $B 220 year^{-1} (6.3.2).
	Estimate the effect on GHG emissions of adopting scenarios of healthier diets in the US.		Healthier diets in the US may reduce GHG emissions in the food system and health care system by 68-1512 and 65-106 kg CO_2 eq. capita^{-1}year^{-1}, respectively (6.3.3).
	Analyze the effect of using different GWP values and time horizons on the GHG emission reduction potential of diet change.		Use of old and inaccurate GWP assignments, particularly in the case of methane, in LCAs may result in the underestimation of the GHG emissions and mitigation potential of diet change. The GHG mitigation potential of adopting healthier diets in the US is estimated to be 2-3 times higher if emissions are evaluated in a 20-year rather than a 100-year perspective (6.3.4).

	OBJECTIVES (Chapter 2)	**METHODS** (Chapter 5)	**RESULTS** (Chapter 6)
PAPER IV	Analyze the scientific basis of dietary scenario analyses assessing the impact of dietary change on GHG emissions and land use demand.	Review analysis (5.2), Life cycle assessment (5.4)	14 articles, including 49 dietary scenarios, which met the inclusion criteria, were identified and included in the review (6.4.1).
	Estimate the potential to reduce GHG emissions, in regions with current unrestricted diets, via diet change.		GHG emissions of current diets (up to retail) range from 1.4-3.2 tons of CO_2 eq. capita^{-1}year^{-1}. The largest potential to reduce GHG emission by dietary change is estimated to be 60% (6.4.2 & 6.4.3).
	Estimate the potential to reduce land use demand, in regions with current unrestricted diets, via diet change.		Land use demand of current diets range from 1400-2100 m^2 capita^{-1}. The largest potential to reduce land use demand by dietary change is estimated to 50% (6.4.2 & 6.4.4)
	Identify and analyze methodological aspects of importance for the quality of results of dietary scenario analyses.		Methodological aspects identified to be of importance are; choice of functional unit; system boundaries; method for scenario development; and method for uncertainty analysis (6.4.5).
	Identify gaps in knowledge in the performance of dietary scenario analyses.		Improved knowledge is required on the uncertainty in dietary scenario analysis, the environmental impact of substitutes and complements for meat, and the effect of dietary change in different groups of populations and geographical locations (6.4.6).

2.2 Research goals and questions

The point of departure of this thesis work is the observation that the current food system is not sustainable, either from an environmental or a health perspective. The baseline assumption that has been the cornerstone and driving force of this work is that sustainability in the global food system can be improved via changes in the current consumption and production of food.

The overall goal of this thesis is to contribute to the knowledge of how sustainability in the food system can be improved. More specifically, the focus of this thesis is on the following three research questions:

A. How large is the potential to improve nutritional quality and health, and reduce GHG emissions and land use demand of the food system via dietary change?

B. Are there synergies and/or conflicts between nutritious and healthy diets and diets with lower impact on GHG emissions and land use demand?

C. Which methodological aspects are of importance for integration of nutrition, health, GHG emissions and land use in sustainability assessments of food and diet?

To achieve the overall goal, four articles (**papers I-IV**), each with a specific aim, have been performed. Table 3 provides an overview of how the research questions of the thesis relates to the specific aims of **papers I-IV**. To answer the questions set out above, the aim of each paper are further refined into more specific objectives. The objectives of **papers I-IV** were previously specified in Table 2.

Table 3. Aims of papers I-IV and the overal research question they are linked to.

PAPER	AIMS	RESEARCH QUESTIONS
I.	To analyze uncertainty and discrepancy in meat consumption statistics and illustrate the consequences of these on subsequent assessments of the effect of meat consumption on the environment and health.	C
II.	To analyze the contribution of nutrients, GHG emissions and land use demand of current Swedish meat consumption (2009), and of two alternative scenarios of healthier meat consumption based on Swedish dietary guidelines.	A, B, C
III.	To analyze the effect of adopting healthier food consumption patterns in the US on upstream GHG emissions from the food sector and downstream GHG emissions in the health care sector due to reduced prevalence of disease by comparing GHG emissions of current US food consumption (2012) and of three alternative scenarios of healthier diet based on American dietary guidelines.	A, B, C
IV.	To estimate the potential of dietary change to reduce GHG emissions and land use demand of current food consumption patterns and identify methodological aspects of importance for such analyses based on a systematic review of dietary scenario analyses published between 2005 and 2013.	A, B, C

2.3 Key concepts

2.3.1 Sustainable nutrition

The title of this thesis is "sustainable nutrition" and the overall goal is to contribute to knowledge that can lead to a more sustainable food system. However, the concepts and definitions of sustainability and nutrition vary, which motivates a further elaboration of how 'sustainable nutrition' is interpreted in this thesis.

The concept of "sustainable development" was coined in the 1980 World Conservation Strategy (IUCN, 1980). The most widely recognized definition of the sustainability concept comes from the UN report "Our Common Future", better known as the Brundtland report (WCED, 1987). Sustainable development can, according to this definition, be described as:

> "A development that meets the need of the present without compromising the ability of future generations to meet their own needs"

Sustainability is often understood as a three-dimensional concept including ecological, social and economic aspects. In accordance with this perception, sustainable food systems reflect a holistic view in which sustainability within these three aspects is considered in all phases throughout the food supply chain (FAO, 2013b). In practice, this means that sustainability of food can be evaluated from several different perspectives, such as on the basis of its environmental impact, resource requirements, nutritional content, health impact, acceptability, accessibility and economic value (FAO, 2011):

> "Sustainable diets are those diets with low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations. Sustainable diets are protective and respectful of biodiversity and ecosystems, culturally acceptable, accessible, economically fair and affordable; nutritionally adequate, safe and healthy; while optimizing natural and human resources"

Another central concept of sustainability is resilience. Resilience refers to the stability of a system, or in other words, the amount of disturbance a system can absorb without changing state (Holling, 1973). Resilience can also be described as the time required for a system to return to an equilibrium or steady-state following a perturbation, or as the disturbance that can be absorbed before a system redefines its structure (Gunderson, 2000). Thus, resilience of the food system can be understood as the adaptive capacity of the system (Gunderson, 1996). A more thorough description of the food system is provided in section 2.3.2.

‘Nutrition’ is often defined as the process of providing or obtaining the food necessary for health and growth. In this branch of science, ‘nutrition’ deals with nutrients and nutrition particularly in humans (OAD, 2015). The expertise of a nutritionist is occasionally queried, as is also the aim of the work done by this profession. In Sweden, a nutritionist has a bachelor or master degree including the subjects chemistry, biology, physiology and nutrition, ranging from a molecular to a public health perspective (NF, 2015). Nutritionists work, among other things, with the provision of information on diet-health and diet-disease relationships, assessments of diet-related research questions, development and application of methods to assess nutritional quality of diets, biochemical assessments of food and diets and/or assessments of how food and diet are affecting public health in the society (Naturvetarna, 2015).

In this thesis the concept of ’sustainability’ is understood as a moving target rather than being a static state. Thus, ’sustainability’ is interpreted as a dynamic target that may vary with time, the region and population, and therefore requires continuous reassessment.’ Sustainable nutrition’ is interpreted to reflect the definition of sustainable diets, but with larger emphasize on the nutritional quality and health effects of food and diets.

2.3.2 The food system

The ‘food system’, as a concept, is often used when discussing why and what we eat, and thus incorporates aspects ranging from how food is produced and reaches our plates, why we eat what we do and how food production and consumption affect the society, economy and natural environment (Tansey & Worsley, 2014). In the literature, a system is described as a network of interacting, interrelated, or interdependent elements that function together as a whole (Meadows, 2008). From a technical perspective, the food system is often referred to as the food supply chain, including all activities from farm to fork and beyond, via processing, transportation, retail, consumer and waste disposal. In reality, many different systems feed into making a food system, including biological-, economic-, social-, health- and political systems (Figure 4). A more thorough description of systems and system thinking is provided in section 3.2.6.

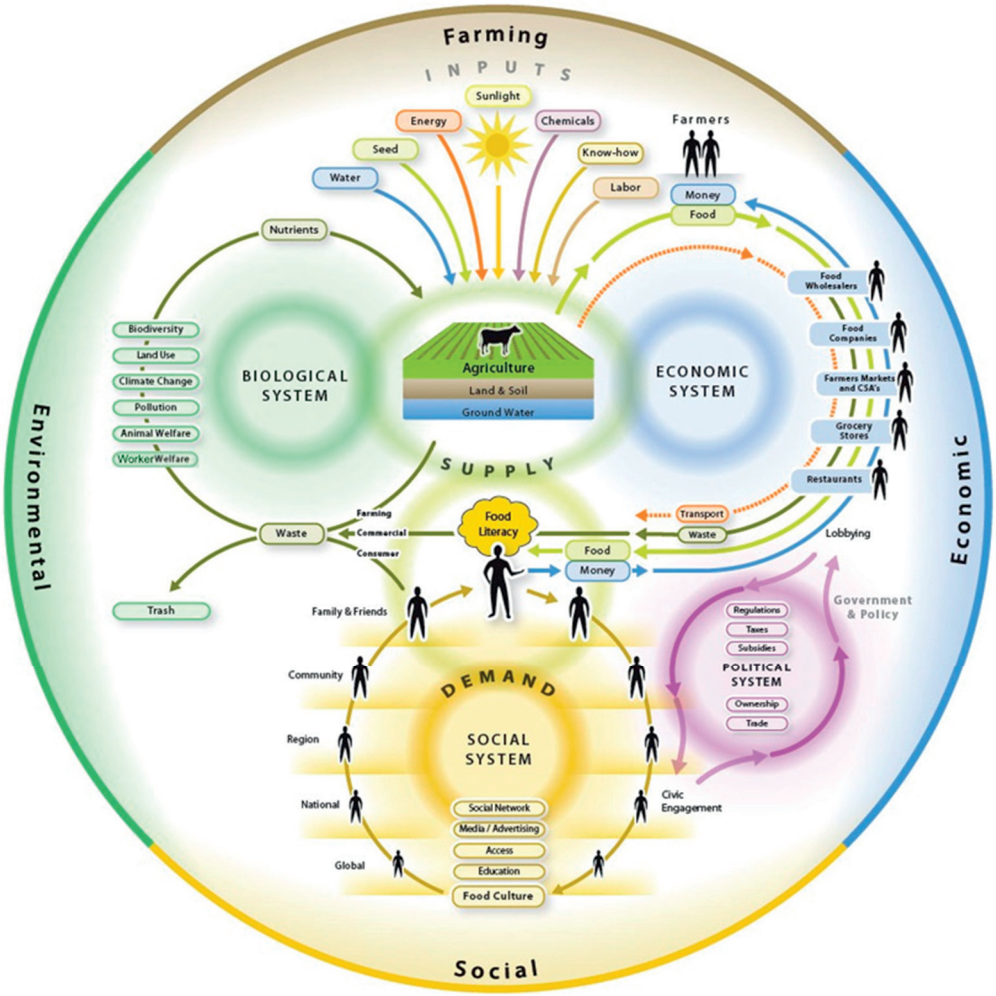

Figure 4. Schematic illustration of the food system. Adapted from Nourishlife (2015).

The food system is a complex system. Food concerns us all as it is a basic need we cannot live without. However, for many, food fulfils a greater purpose than fulfilling nutritional needs, by its importance in social interaction, culture and community. Considering the fact that the global food system needs to produce food to feed seven billion people every day, one quickly understands that the food system involves a myriad of actors at different levels of the society. In today´s globalized world, the everyday food consumed may originate from different and often widespread corners of the world. This implies that food consumption may have consequences both locally and in distant regions of the globe.

2.3.3 Risks and opportunities

There is no general agreed definition of the concept of 'risk', and therefore it´s meaning varies depending on the discipline (Aven, 2012). According to the ISO 31000, 'risk' is "the effect of uncertainty on objectives", where an 'effect' is a positive or negative deviation from what is expected. Mathematically, 'risk' is calculated as the "cumulative effect of the probability of uncertain occurrences" (Pritchard et al., 2014).

'Risk' is often understood as a synonym to 'danger' or 'hazard'. However, risks can be categorized into both downside risks, e.g. the risk we generally think about referring to a negative or unpleasant situation, and upside risks, referring to positive situations and opportunities (Ward & Chapman, 2003). The distinction between downside and upside risks may be complex, as a situation may be perceived as a downside risk in a specific setting, situation or for a specific individual, whereas the same situation may be perceived as an opportunity somewhere else or by somebody else (Aven, 2012). A distinction can therefore be made between subjective, objective, real, observed and perceived risks (Althaus, 2005). A distinction is sometimes also made between technical and non-technical risks (Hansson, 2014). Risk assessment, as a theory and method, is further described in section 3.2.4.

In this thesis, four risks are assessed (Table 4). These risks are considered non-technical risks, although technology plays an important role for their outcome. In order to evaluate whether and/or how dietary choices affect these risks, four indicators are analyzed and measured. Obviously, other potential risks and indicators are of importance than the ones assessed in this thesis. Thus, the results presented in this thesis must be complemented and refined with other perspectives and sources for a more complete understanding of the system studied. The scope and boundaries of the thesis are further described in section 3.3, and a description of sustainability indicators, metrics and targets used is provided in chapter 4.

Table 4. Risks and indicators assessed and used in this thesis

	DOWNSIDE RISKS	OPPORTUNITIES	INDICATORS
I.	Climate change[1]	Mitigation of climate change[1]	Greenhouse gas emissions[1]
II.	Land use change[2]	Mitigation of unsustainable[3] land use change[2]	Land use demand[2]
III.	Mal-nutrition[4]	Improved nutritional quality[4]	Nutrient intake and nutritional quality[4]
IV.	Disease	Health[4] promotion and disease prevention	Food intake and dietary quality[4]

The concepts of [1]'greenhouse gas emission and climate change', [2]'land use and land use change', [3] 'sustainability', [4]nutritional quality and health are described in section 4.2; 4.3; 2.3.1 and 3.3.1; and 4.4, respectively.

2.3.4 Uncertainty and variability

A key concern in this thesis has been how to handle uncertainty and variability in data, methodology and results. For a better understanding of the concept 'uncertainty', a distinction can be made between 'uncertainty' and 'variability'. 'Uncertainty' exist due to incomplete knowledge and can, in general, be reduced by improving the performance, and thereby the accuracy and quality of an assessment. For example, uncertainty can be reduced by validation of the data and methods used. 'Variability', on the other hand, is a consequence of the heterogeneous nature of data, and can thus not be reduced by improving the performance. Thus, 'variability' is instead handled by estimating and accounting for the variability within specific input parameters, and their overall impact on the results of the assessment (Björklund, 2002; Huijbregts, 2002). Despite the difference in meaning between the two terms, in the literature 'uncertainty' is often used to express both 'uncertainty', as defined above, and 'variability'. Therefore, in the following, 'uncertainty' refers to both uncertainty and variability, if not otherwise stated.

Methods with low uncertainty and variability provide results with high precision. High precision can thus be understood as assessments with high reproducibility, i.e. good agreement between expected and actual results. However, methods with high precision do not necessarily measure what they are intended to measure. Methods that, with a high degree of certainty, measure what they are intended to measure are said to have high validity. The difference between measuring precision and validity is sometimes illustrated by picturing a shooting target. Here high precision corresponds to all shots being assembled in one area, whereas high validity corresponds to the shots being, on average, centered around the "bull´s eye" (Figure 5). Thus, the overall reliability (uncertainty and variability combined) of an assessment depends on both the precision and validity of the underlying data and methods (Ahlbom, 2006). Uncertainty assessment, as a theory and method is further described in section 3.2.5.

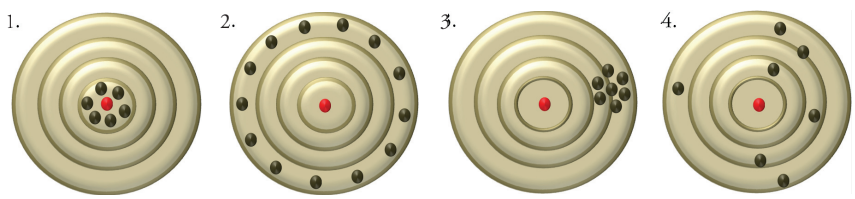

Figure 5. Illustration of precision and validity. 1. High precision & validity; 2. Low precision, high validity; 3. High precision, low validity. 4. Low precision & validity.

2.3.5 Sustainability indicators, metrics and targets

As described in section 2.3.1, 'sustainability' is understood as a dynamic concept which requires continuous reassessment. To evaluate and provide guidance on what can be considered sustainable, sustainability indicators, metrics and targets are used. Indicators refer to a characteristic or condition that enables decisions or value judgments about something, for example, an activity, undertaking, result or situation. Metrics are quantitative standards of measures or rating, whereas, targets indicate commitments to achieve a final performance or status (Blackburn, 2012).

The term environmental sustainability is based on the idea that the biophysical characteristics of the earth impose certain constraints on human activities. Various frameworks have been developed and suggested in order to conceptualize the constraints or limitations linked to environmental sustainability, for example, "limits to growth" (Meadows, 1974), "carrying capacity" (Daily, 1996), "ecological resilience" (Holling, 1973), "ecological footprint" (Wackernagel, 1995) and "planetary boundaries" (Rockström et al., 2009; Steffen et al., 2015). These frameworks have further been used to establish specific criteria and targets to determine what is sustainably acceptable, and indicators and metrics to assess the progress towards environmental sustainability (Moldan et al., 2012).

Depending on the chosen indicator and target, sustainability can be assessed based on qualitative or quantitative metrics, absolute or relative metrics, or by using indices combining several metrics (Blackburn, 2012). Common practices include the use of sustainability reference values, i.e. the use of a baseline value against which the progress can be related, or a threshold value indicating the distance between the current situation and a set critical limit. Sustainable development can also be evaluated against performance indicators such as national or international policy targets (Moldan et al., 2012). As an example, the Millennium Development Goals include eight global-scale policy goals to be achieved by 2015, that are broken down into twenty one quantifiable targets measured by sixty sustainability indicators (UN, 2010a).

As a response to the growing demand from the policy side, various indicators, metrics and targets have been developed over the past decades to assess the sustainable development within different areas, sectors and activities (Moldan et al., 2012). In many cases several indicators, directly or indirectly related to a specific sustainability target, have been developed. To increase the robustness and/or extend the perspective, complementary indicators and/or metrics at different levels along the cause-and-effect chain can be used to assess the progress (or failure of progress) towards sustainable development (Blackburn, 2012). The sustainability indicators, metrics and targets used in this thesis are further described in chapter 4.

2.4 Description and motivation of publications

This chapter provides an overview and describes the motivation of **Paper I-IV**.

2.4.1 Paper I

In **paper I** the generation, presentation and use of meat consumption statistics is analyzed. The topic is of interest because consumption statistics often are combined with data on, for example, the environmental impact or nutritional content of food in order to assess the 'sustainability' of food consumption patterns. However, analyzing the effects of current and alternative food patterns requires knowledge to understand which data to use. Thus, the motivation for **paper I** was to gain knowledge of the aspects of importance for a proper use and interpretation of meat consumption data in subsequent analyses. The paper only covers meat consumption statistics, but is in part applicable for food consumption statistics in general. Why does meat consumption statistics vary between different data sources? How reliable is data on meat consumption? How is meat consumption data presented? And for what analyses can the data be used? These were underlying questions that formed the basis of **paper I**. Answering these questions was an important and necessary step to perform the scenario analyses in **papers II** and **III**.

2.4.2 Paper II

Paper II analyzes the implications of limiting Swedish meat consumption in accordance with guidelines for healthy diet. At the time when **paper II** was being prepared, environmental assessments of individual foods and meals had pointed out meat as one of the food groups with highest environmental impact. However, so far the knowledge of the environmental impact of complete diets and their constituent food groups, and the potential for dietary change to reduce the environmental impact of the food system, was limited. At the same time, a growing number of publications and actors suggested that diets in line with healthy recommendations offered synergistic effects of reduced environmental impact. While the health and environmental synergies from increased fruit and vegetable consumption were more or less indisputable, a concern was expressed for the consequences of reduced meat consumption on nutrition. With this background, the main motives of **paper II** were threefold. Firstly, the motive was to estimate the environmental impact of meat consumption from an unrestricted diet, based on Swedish food consumption patterns. Secondly, the motive was to estimate the potential to reduce the environmental impact of the diet by adopting healthier consumption patterns of meat. Finally, the motive was to estimate the effects of reduced

meat intake on the nutritional quality of diet and nutritional status in populations with unrestricted diet, in this case the Swedish population.

2.4.3 Paper III

Paper III analyzes the effect on GHG emissions by adopting healthier food consumption patterns in the US, based on the American guidelines for healthy diet. Thus, **paper III** provided an opportunity to gain further knowledge of the environmental impact associated with unrestricted diets and of the potential for dietary change to reduce the environmental impact of the food system. While the climate impact of dietary patterns typically is quantified on the basis of GHG emissions produced along the food chain, the motive of **paper III** was to expand the system boundaries to include the effect of dietary choices from GHG emissions produced in the health care sector. To enable such analyses, a methodological framework that combines nutritional epidemiology and LCA, was developed. The framework enabled assessments of the potential synergies between GHG emissions of diets, relative risk of non-communicable diseases and the indirect effect on associated health care expenses. Meanwhile working with **paper III**, IPCC released their fifth assessment report on climate change. Among other things, the report suggested that emissions of methane have a stronger impact on climate change than what was previously anticipated. Based on this, an additional motive of **paper III** was to estimate the effect of the revised GWP values for methane on the GHG emissions of food consumption and the GHG mitigation potential of dietary change.

2.4.4 Paper IV

In **paper IV** a systematic review of dietary scenario analyses assessing the environmental impact of dietary change is performed. Over the past decade, research on the environmental impact of food and diets has grown, from being a relatively new and unexplored field, to a major research area. Knowledge about the environmental impact of existing and alternative diets has been improved as the number of published dietary scenario analyses has been growing. The motive of **Paper IV** was to provide an overview of the state of knowledge in 2014, on the potential of dietary change to reduce GHG emissions and land use demand of current food consumption patterns. As the methodological approach of dietary scenario analyses vary and can have a decisive effect on the results of the analysis, an additional motive of **paper IV** was to analyze the study designs used in published dietary scenario analyses and their implications for the outcome.

3 Methodological background

3.1 Overview of theories, perspectives and boundaries

This chapter describes the methodological background of the thesis. The chapter includes an overview and summary of the theoretical framework (3.1) and descriptions of the research theory (3.2), scope and boundaries (3.3) of the thesis. Figure 6 provides an overview of the goal and scope formulation process of the thesis.

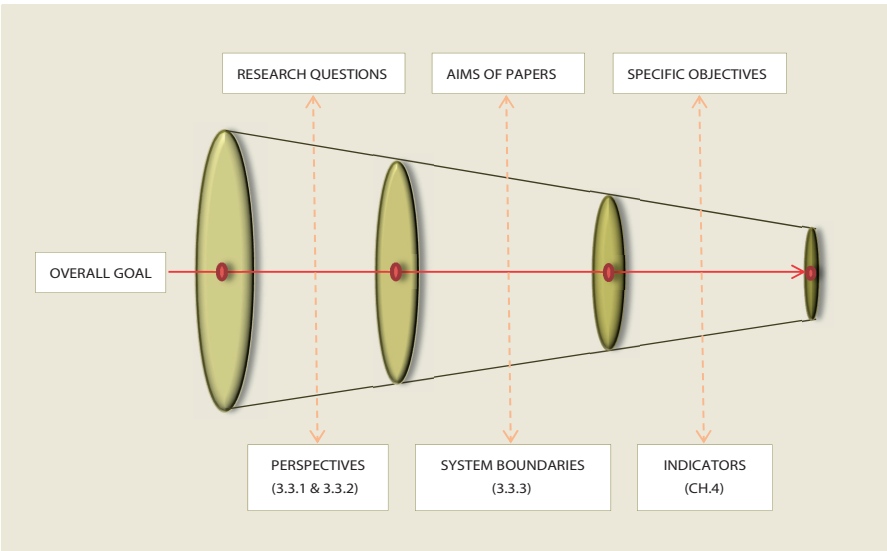

Figure 6. Schematic illustration of the goal and scope formulation process in which the overall goal was refined into specific objectives by the choice of perspectives, system boundaries and indicators studied.

3.2 Research theory and positioning

3.2.1 Theoretical background

The work of this thesis is based on the belief that our perception and knowledge of the world is being constantly reshaped, as new theories and hypotheses are formed and rejected. In this sense the development of theories, for example, on how the choice of diet impacts on natural ecosystems, society and physiology of the human body and mind, is the basis of how new and improved knowledge is developed.

In research based on social constructions, such as in this thesis, uniform "truths" and "solutions" rarely exist. Therefore, in this thesis the "truth" is assumed to correspond to the "reality" or "knowledge" of today, i.e. the best current understanding based on scientific evidence. This approach is applied with great humility for the possibility that "today´s reality" is not necessarily consistent with that of the future, and that the "best solution" of a problem may vary depending on the region, population, perspective and specific situation. To solve a complex problem may require a combination of several theories and "truths" from different disciplines. In this thesis the intention is not to advocate a specific position or solution, but rather to explore the implications of available options that can improve sustainability of the food system.

3.2.2 Inter-disciplinary research

In the literature there are various ways of describing inter-disciplinary research (Aboelela et al., 2007; Repko, 2011). The following definition is well consistent with the approach used in this thesis:

> "Interdisciplinary research is a mode of research that integrates information, data, techniques, tools, perspectives, concepts, and/or theories from two or more disciplines or bodies of specialized knowledge to advance fundamental understanding or to solve problems whose solutions are beyond the scope of a single discipline or area of research practice" (National Academy of Sciences, 2005)"

The traditional approach in research is to break down the problem to be analyzed into particular branches of disciplines that are studied separately. As real-world problems often cut across the borders of many disciplines, the need for bridge-building between different, but interrelated disciplines has become increasingly acknowledged.

Depending on the level of integration, a distinction can be made between multi-disciplinary, inter-disciplinary and trans-disciplinary research. In multi-disciplinary research the same issue or problem is studied by different disciplines which work in

parallel without integrating their methods. Inter-disciplinary research brings together insights produced from different perspectives and disciplines to enable an integrated analysis of a research topic. At an even higher level of integration, trans-disciplinary research is a problem-based approach which combines knowledge, theories and methods that span over several disciplines within both the academic and non-academic sector (Frodeman et al., 2010).

The research approach in this thesis ranges from multi-disciplinary to trans-disciplinary. Frameworks used in this thesis are based on theories originating from nutrition ecology (3.2.3), risk and uncertainty assessment (3.2.4; 3.2.5), and system analysis (3.2.6). The perspectives and methods used in the thesis originate in the fields of environmental, nutrition and health studies, and are further described in section 3.3 and chapter 5.

3.2.3 Nutrition ecology

Nutrition ecology is a systemic and solution-oriented scientific approach developed to solve complex and multi-dimensional, nutrition-related problems (Schneider and Hoffmann, 2011).

The term 'nutrition ecology', was first used to describe interactions between nutrition and the environment by Gussov (1978). 'Nutrition ecology', as a framework, was further developed in Germany in the mid-1970s (Leitzmann, 2003; von Koerber, 2011). In science, nutrition ecology is used as a method to integrate knowledge from different disciplines dealing with nutrition related questions, problems and practices (Schneider & Hoffmann, 2011). Solving a problem by using nutrition ecology as a methodological framework has been described by using the metaphor of a Rubik´s Cube (Dörner, 1996; Schneider & Hoffman; 2011; Vester, 2007):

> "The difficulty in solving the cube is that never one part can be changed. As soon as one part is turned, other parts of the cube move as well. Therefore, it is impossible to do one thing without simultaneously changing other things. Similar difficulties arise when dealing with complex nutrition-related problems that are characterized by a multitude of factors, inter-relatedness, and associated feedback, dynamics and intransparencies"

Nutrition ecology can, thus, be understood as an interdisciplinary research approach that focuses on nutrition-related problems. Due to the heterogeneity of complex nutrition-related problems, there is no standardized set of methods available in nutrition ecology. Rather, appropriate methods from different fields are used and chosen depending on the problem studied (Mertens, 2015; Schneider & Hoffman, 2011). In this thesis, theories and methods originating from nutrition ecology were partly used to assess environmental, nutritional and health effects of dietary change.

3.2.4 Risk assessment

Risk assessment and management are tools developed to systematically identify, assess, monitor and mitigate risks in different projects, activities, organizations or systems. In society, most robust prioritization processes involve risk assessment and management principles, for examples, development and evaluation of environmental and health legalization both local, national and international (ISO, 2009; Ricci, 2006).

The processes and practices involved in risk assessment and management can be described by six steps, 1) the planning of the necessary infrastructure and organization, 2) the identification of potential risks, 3) the qualification of these identified risks, 4) the quantification of these risks, 5) the planning of responses to the risks, and 6) the monitoring and controlling of the risks (PMI, 2013).

As described in section 2.3.3, the concept of risk can be understood as the cumulative effect (i.e. total accumulated effect of successive additions) of the probability of uncertain occurrences (Pritchard et al., 2014). Ricci (2006) describes the concept of 'risk' nicely, with the following example:

> "If the risk of prompt death due to a specific choice is stated as 1% per year, above background, this means that there is a 0.01 probability of death per year in that activity. Intuitively and mathematically, the probability of surviving that particular hazard is 0.99 per year. If the activity is beneficial, the value of the benefit can be expressed in monetary units or some other unit, such as utility. Clearly, there will be several combinations of probability and magnitude. All of them, in the appropriate order, form the distribution of the magnitude of a particular adverse outcome."

Essential in risk assessments, is the estimation of the magnitude of potential consequences of activities. 'Consequence' describes the outcome of an event, or a range of events, which can have both positive and negative effects (ISO, 2009). The consequence of an activity is calculated by multiplying the estimated probability by the impact of the activity (Aven, 2012). Figure 7 illustrates how the consequence/s of a number of activities can be assessed, based on their individual and combined probability and impact. The impact of an event can be assessed on the basis of both qualitative and quantitative measures. In practice, the magnitude of the consequence is often estimated based on the economic cost of the activity (Ricci, 2006). However, many other indicators can be used, which will be discussed further in chapter 4.

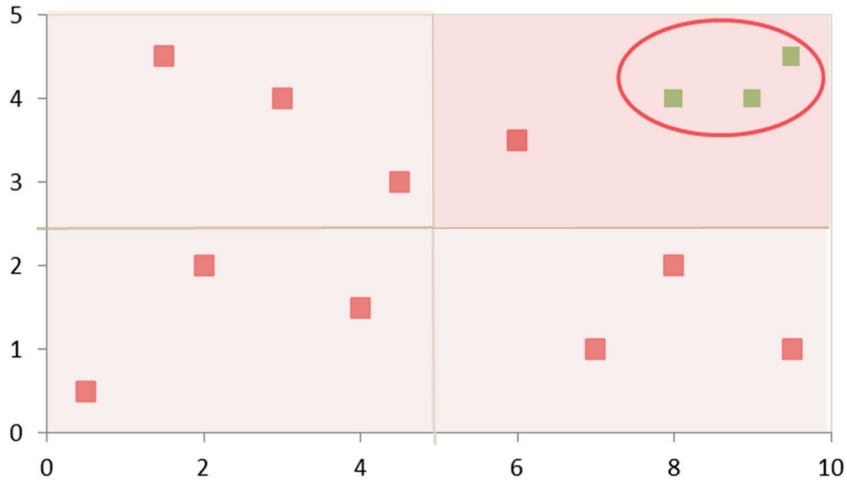

Figure 7. Schematic illustration of a risk assessment model of the probability (Y-axis) and impact (X-axis) of activities. Red circle indicates activities with large consequence/s (in green). Courtesy of Brian Atkin.

Once a risk has been identified and assessed, it will be required that the risk is managed. In the literature, several strategies for risk management are suggested. Common strategies for risk mitigation are; avoidance (e.g. avoid the risk in the first place), acceptance (e.g. take the risk), transference (e.g. reduce the risk by taking an insurance), or control of risks (e.g. detect and/or prevent the risk) (Pritchard et al., 2014).

In this thesis, the risks and opportunities studied are those that may arise from dietary change. Dietary change is, thus, studied both as a potential risk, and as a potential opportunity to manage and mitigate risk. The risks assessed in this thesis were previously described in section 2.3.3.

3.2.5 Uncertainty Assessment

As previously mentioned, a key concern in the methodological approach of this thesis has been to assess and manage uncertainty. When studying complex inter-disciplinary systems, such as the food system, the methods available are often hampered by both uncertainty and variability (2.3.4), and thereby provide results that can rarely be fully generalized.

Calculation and/or modelling of environmental, nutritional and health effects of food production and consumption, may introduce several types of uncertainties at different levels (Hallström, 2013; Röös, 2013). Such uncertainties are sometimes categorized as model uncertainty, data uncertainty and scenario uncertainty. Röös (2013) describes

these differences, by using the example of calculating GHG emissions of food (Figure 8). Model uncertainty is exemplified by GHG emissions from biological processes associated with agriculture (e.g. N_2O soil emissions) which vary greatly depending on regional and local conditions; data uncertainty (or variability) is exemplified by yield levels, energy use, type of fertilizers and feed, i.e. all data that are fed into the model in order to perform the calculations; and scenario uncertainty is exemplified by the choice of functional unit, system boundaries, allocation principles etc., i.e. all methodological choices and assumptions that are built into the calculation or model. The total uncertainty in GHG emission calculations of food is thereby the result of the uncertainty in the underlying methodology and data.

Monte Carlo analysis, which is further described in section 5.7, is the main method used for uncertainty assessment in this thesis. Uncertainties in the methods and the results in this thesis are further discussed in section 7.1.

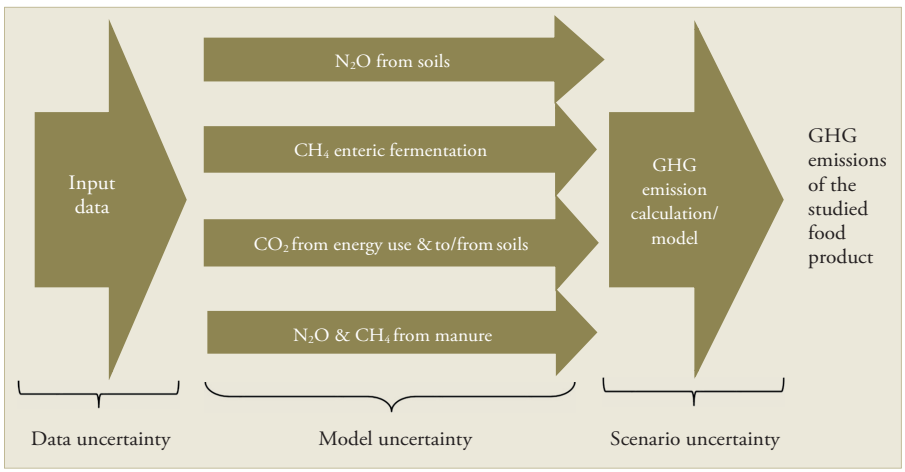

Figure 8. Types of uncertainty contributing to the total uncertainty in GHG emissions from a food. Adapted from Röös (2013).

3.2.6 System thinking

System thinking, analysis and dynamics were developed, as frameworks and tools, in the late 1950´s by the American professor Jay Wright Forrester. System thinking can be understood as a branch within interdisciplinary research and refers to the science dealing with non-linear behaviors of complex systems. System dynamics and analysis have further been described as methodologies used to observe dynamic relationships between variables in a system (Haraldsson, 2000) and/or as tools to structure logic in order to identify what, how and when to act in complex situations (Dörner, 1996).

A system can be thought of as a network of interacting, interrelated, or interdependent elements that function together as a whole (Meadows, 2008). Thus, on a system level a single intervention within the system can release a cascade of changes, bringing both positive and negative side effects in addition to the intended effect/s. Therefore, a principal attribute of a system is that its behavior can only be understood by viewing the system as a whole (Grant, 1997).

Hughes (2011) describes the foundation and organization of systems on five levels. The first consists of the individual compounds of the system, and the following levels illustrate how these compounds are organized and interact with each other. To understand the concept of system thinking, I personally like the metaphor of learning a language. The compounds can be viewed as the vocabulary, i.e. the individual words that build up the language; the next level can be thought of as the grammar describing how individual words interrelate and can be combined to make sentences. At an even higher level, the process of learning to read, understand and express oneself, is required to completely understand and master the language. In a larger perspective, a system is often embedded within a larger system, and itself consists of subsystems (Haraldsson, 2000). As an example, the human system is a subsystem within the broader biophysical world system (Figure 9).

In this thesis, the primary focus of research is the food system, previously described in section 2.3.2.

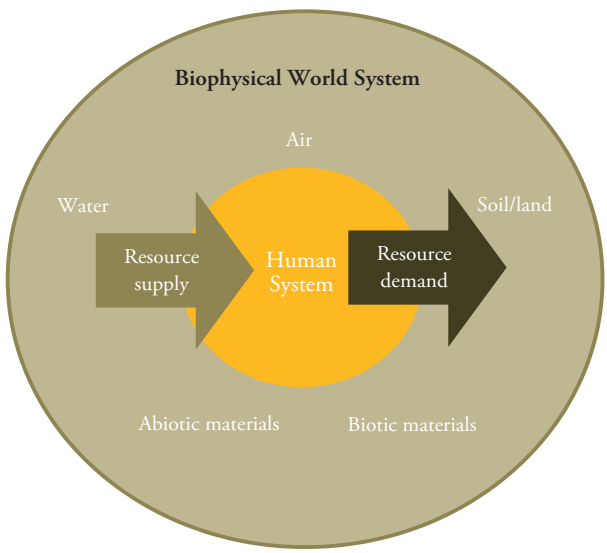

Figure 9. Schematic illustration of the human system as a subsystem of the biophysical system. Courtesy of Deniz Koca.

3.3 Perspectives and boundaries

As described in section 2.3.1 and 2.3.2, the concept of sustainable food systems is very complex. Thus, how to improve sustainability in the global food system is a question with more than one answer. The goal of this thesis is not to cover all sustainability perspectives, nor is the intention to assess all impact categories, or all perspectives in time and space. The following sections describe the perspectives and boundaries used in this thesis and for which the results are applicable.

3.3.1 Sustainability perspective

The sustainability perspectives used to assess the potential for improving the sustainability of the food system are limited to the environmental, ecological and human health perspectives. The environmental and ecological effects of food production has primarily been studied in a top-down approach, where the planetary boundaries or earth´s carrying capacity have been used as thresholds indicating the environmental and ecological limitations or playground to which we humans must adapt. To assess the nutritional quality and health effects of food and diets, instead primarily a bottom-up perspective is applied.

More specifically, the impact categories studied in this thesis are limited to climate impact, land use demand, human nutrition and risk of disease. Annual per capita GHG emissions and use of agricultural land are used as sustainability indicators to assess the climate impact and land use demand of food consumption and production. The assessment of the nutritional quality of diets, and their effect on human health and disease is based on reference values for healthy intake levels of nutrients and food, as well as on public health dietary recommendations and guidelines. The impact categories and indicators studied in this thesis may, but are not necessarily, also indirect indicators for additional, related aspects.

Figure 10 provides a schematic overview of a selection of perspectives and impact categories of importance to enable sustainability within the food system, and illustrates which of these are directly or indirectly studied in this thesis. As shown in Figure 10, many aspects of importance for the sustainability of the food system are not considered in this thesis. Thus, to get a complete picture of all relevant aspects of the sustainability of food and diets, the scope of this thesis needs to be complemented by perspectives from other angles.

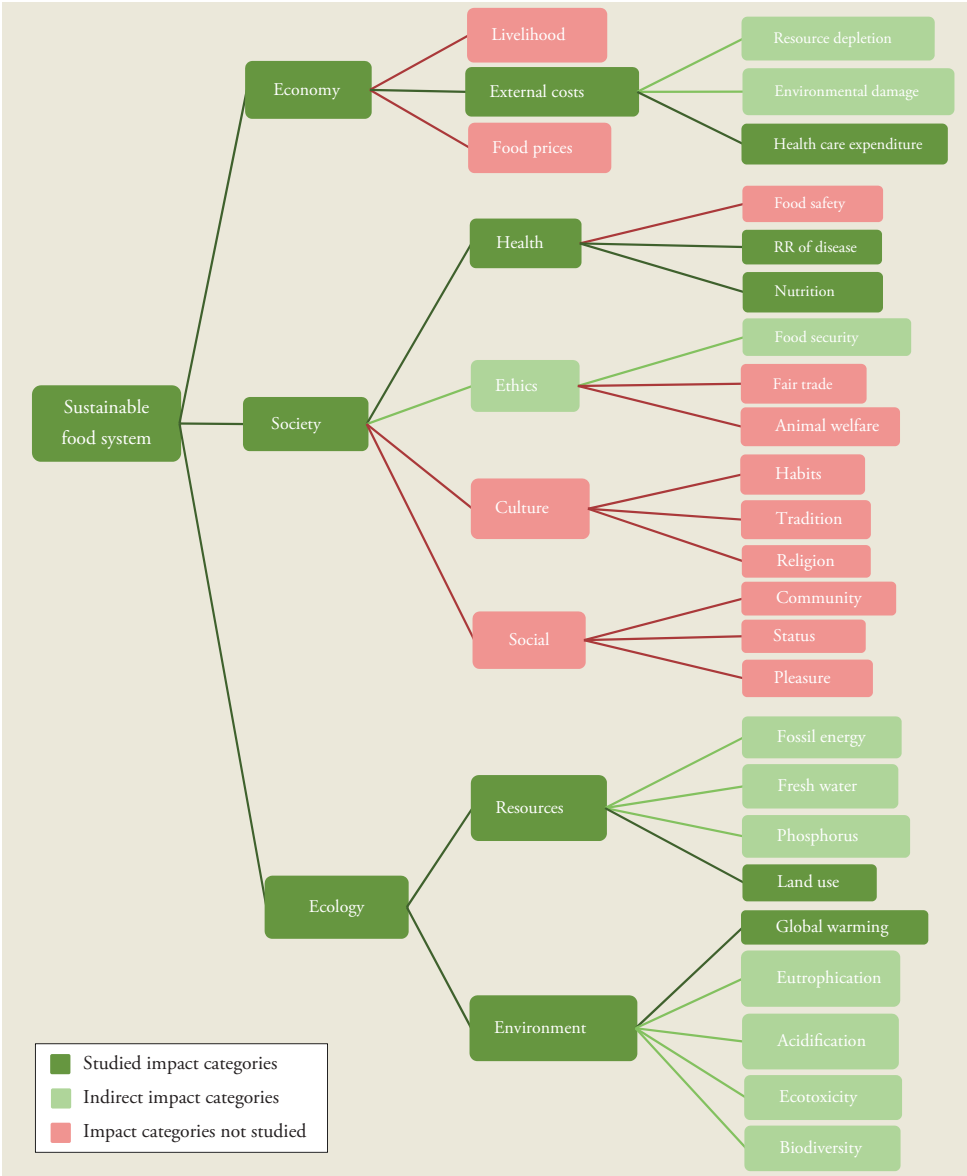

Figure 10. Schematic overview of the impact categories studied and not studied in the thesis

3.3.2 Nutrition and health perspective

In terms of nutrition and food-disease relationships, a distinction is made between the individual and the public health perspective. Nutritional requirements differ between individuals and populations due to a large number of variables, including gender, age, prevailing nutritional and health status. For this reason, it is difficult to determine the nutritional status and give nutritional recommendations on an individual level, without performing biochemical measurements (e.g. blood samples).

In this thesis, a public health perspective is applied to assess the nutritional quality and health effects of food consumption. This means that the nutritional and health assessment is based on dietary recommendations and guidelines that promote public health, i.e. health for the majority of the people within the studied population, rather than among individuals. The public health perspective further implies that the focus is on health promotion and disease prevention, in contrast to the clinical health perspective that focuses on reducing or curing the symptoms of a disease or illness.

Nutritional and health effects of food consumption are moreover analyzed in the perspective of high-income countries, and therefore mainly apply to populations with unrestricted diet. While the importance of focusing on whole diets is emphasized both in nutritional and environmental assessments of the diet, in this thesis the focus has mainly been on some selected food groups; meat, vegetables, legumes, fruit and grains. Thus, the effects of the consumption and production of fish, dairy products, sugar and oils have not been part of the analysis in this thesis. While acknowledging the need to include these food groups for a complete assessment of the environmental, nutritional health effects of the diet, this is beyond the scope of this work.

3.3.3 Boundaries in time and space

In this thesis, the time perspective is limited to a near-time perspective, meaning that production systems correspond to today´s performance without any assumptions on technical development. The spatial boundaries regarding are limited to food consumed in Sweden and the US, including upstream and downstream effects.

3.3.4 Other limitations

Improved sustainability in the food system can be realized through many different measures, including both supply- and demand-side strategies. In order to feed the growing population without exceeding the environmental and ecological capacity of the planet, a combination of these measures is believed to be necessary. In this thesis, the primary focus is on the demand side strategies, while supply-side strategies related

to agricultural management and technological performance are considered to a lesser extent. More specifically, the demand-side strategies studied are limited to the amount and type of food consumed and produced. The scope of this thesis is thus limited to the study of how and whether sustainability in the global food sector can be improved via dietary change.

4 Sustainability indicators, metrics and targets

4.1 Overview of indicators, metrics and targets

This chapter describes indicators, metrics and targets used in this thesis to assess and quantify the sustainability of alternative food consumption and production patterns. The chapter includes an overview of all indicators, metrics and targets used in **papers I-IV** (4.1), and a more detailed description of the specific indicators, metrics and targets used to assess GHG emissions and climate change (4.2), land use demand (4.3), and nutrition quality and health (4.4). The chapters are organized in two sections, where the first provide a general description of existing indicators, metrics and targets, and the second describes which of these, and how these were used in **papers I-IV**.

Table 5 and Figure 11 provide an overview of the indicators, metrics and methods used in this thesis.

Table 5. Overview of perspectives, impact categories and indicators and metrics used

PERSPECTIVE	IMPACT CATOGORY	IMPACT INDICATOR	METRIC
ENVIRONMENT	Climate impact	GHG emissions	kg of CO_2 eq. capita^{-1} year^{-1}
RESOURCE/ ECOLOGY	Land use demand	Use of agricultural land	Hectares of land capita $^{-1}$ year^{-1}
NUTRITION	Nutritional quality	Dietary contribution of nutrients	g of nutrient intake capita $^{-1}$ day^{-1}
HEALTH	Health promotion/ disease prevention	Risk of disease	% of change in RR of coronary heart disease, diabetes type II and colo-rectal cancer.

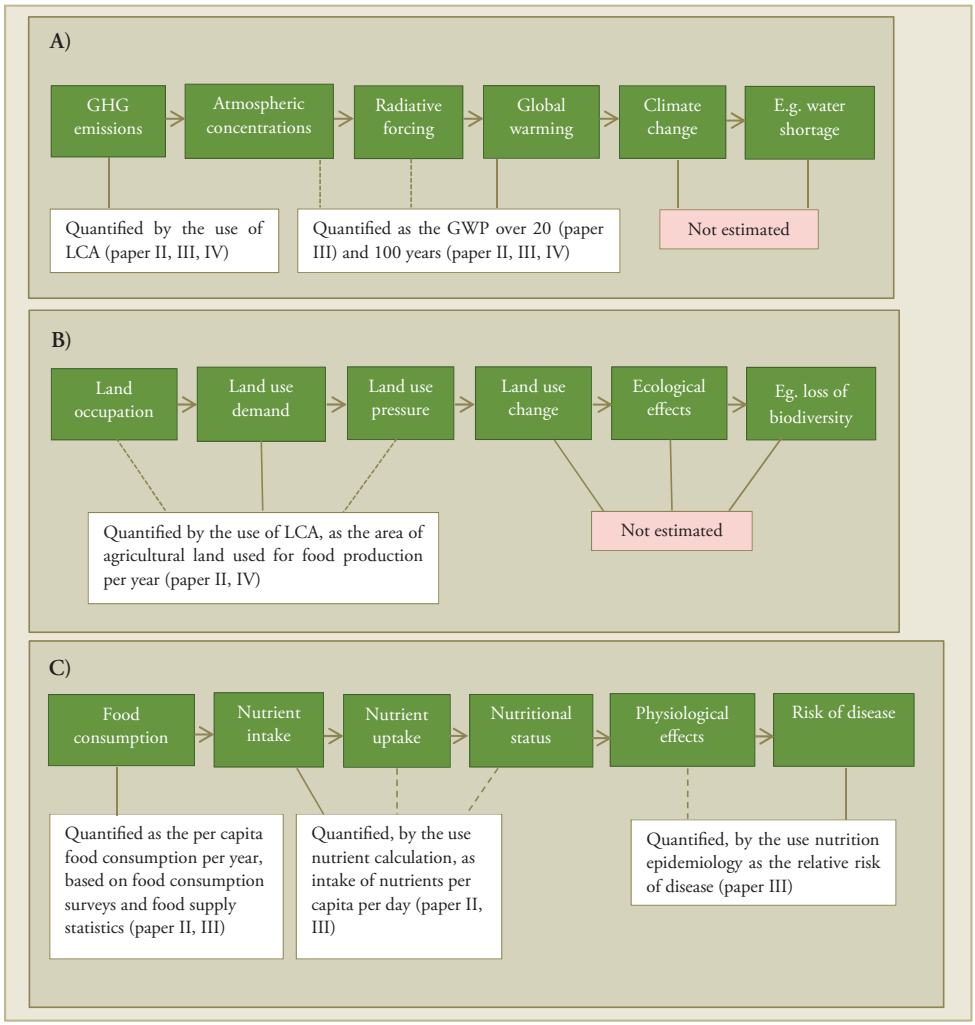

Figure 11. Overview of methods and indicators used (and not used) to measure A) GHG emissions and climate change, B) Land use demand, and C) nutritional intake and associated health effects.

4.2 Greenhouse gas emissions and climate change

4.2.1 General description

The temperature on earth is determined by the balance between incoming and outgoing energy. The difference between incoming solar energy to the earth and energy radiated back to space per unit area (W/m^2) is termed radiative forcing. A positive radiative forcing indicates a net increase of incoming energy which contributes to global warming, whereas a negative radiative forcing has a cooling effect. Increased absorption of infrared radiation, due to accumulation of GHGs in the atmosphere, is the dominant cause of the overall radiative forcing. The concept of radiative forcing forms the foundation for most metrics used to quantify and compare the climate impact associated with different factors.

The climate impact of a product or a process is most commonly expressed as the global warming potential (GWP) of its associated GHG emissions. The GWP is quantified as the ratio of the time-integrated radiative forcing of the emissions from 1 kg of a compound, relative to that of 1 kg of CO_2. In other words, the GWP value expresses how much heat a GHG will absorb in the atmosphere within a certain period of time, compared to an equivalent amount of CO_2. The GWP value of a GHG is determined by its ability to absorb heat in the atmosphere, the wavelength at which the gas in question absorbs heat, and its lifetime in the atmosphere (IPCC, 2007).

To compare, or add the climate impact of different GHGs, the emissions are converted into the common unit of CO_2 equivalents (CO_2 eq.). The amount of CO_2 eq. generated is determined on the amount of each gas emitted and its GWP value over a certain time period. As all GWP values are expressed in relation to carbon dioxide, the GWP value of carbon dioxide is standardized to one. The total GWP from emissions of carbon dioxide, methane and nitrous oxide, the main GHGs released from the food system, is thus calculated as:

$$GWP_{tot} = \text{amount of } CO_2 * GWP\ CO_2 + \text{amount of } CH_4 * GWP\ CH_4 + \text{amount of } N_2O * GWP\ N_2O$$

The atmospheric lifetime of different GHGs ranges from approximately 1 to 50,000 years. Due to the variation in lifetime of different GHGs, the GWP values vary depending on the time period considered. The most recent IPCC assessment provides GWP values for a time period of 20 and 100 years (Table 6). Most commonly a time period of 100 years is used, but the choice of time period depends on the specific objective (Myhre et al., 2013). The choice of time period has a strong impact on the GWP values. Short-lived GHGs, such as methane with an atmospheric lifetime of

about 12 years, will for example have a higher GWP value in a 20-year time period than in a 100-year time period.

Table 6. Global warming potential (GWP) values for different time horizons.

TIME PERIOD	CO_2	CH_4	N_2O
20 years	1	86	268
100 years	1	34	298

*Including climate-carbon feedbacks (Myhre et al., 2013).

The GWP values provided by the IPCC have been updated four times (1995, 2001, 2007, 2013), since the concept was first presented in 1990. The updates have particularly affected the GWP value for methane, which has increased from 21 to 34 expressed over a 100-year time period. The changes in GWP values are due to the development of new estimates on atmospheric life times and radiative efficiencies of GHGs. A major difference in the most recent GWP values is that the effect of climate-carbon feedbacks of GHGs are included, which have previously not or only partially been accounted for (Myhre et al., 2013).

Emissions of GHG emissions can, besides global warming, serve as an indicator of other effects linked to climate change, including extreme weather events, water shortage, decreasing yields and external costs of these events. In order to account for other impacts of GHG emissions, several alternative metrics for the GWP have been proposed (Tanaka et al., 2010). One example of an alternative measure is the global temperature change potential (GTP) (Shine et al., 2005) which estimates the temperature change at the end of a specified time period, relative to that of carbon dioxide. In addition to metrics based on physical values, metrics accounting for economic dimensions have been developed (Deuber et al., 2013). As no single metric can compare all consequences of emissions of different GHGs or substances, the choice of climate metric will depend on the purpose of its application and the aspects of climate change that are judged to be most important (Myhre, 2013).

4.2.2 Research approach

In **papers II, III** and **IV**, the GHG emissions embodied in food (i.e. for a specific food along a specified life-cycle) are evaluated based on the total GWP of the food item consumed.

The 100-year GWP values from the fourth IPCC assessment (IPCC, 2007) are the basis for the GHG quantifications in **papers II, III** and **IV**. Furthermore, in **paper III** the GHG emissions are calculated based on the updated GWP value for methane (i.e.

GWP_{CH4} = 32) (IPCC, 2014; Myhre et al., 2013). In **paper III,** GHG emissions are moreover calculated over both a 100- and 20-year time perspective.

Emissions of GHG from current and alternative food consumption patterns are evaluated against current per capita GHG emissions of the population studied. In addition, GHG emissions are evaluated against a theoretical level of sustainable emissions set to 1.5 tons of CO_2 eq. capita^{-1} year^{-1} (total emissions), in **paper II**, and against existing GHG mitigation targets, in **paper III.** An overview of the metrics, indicators and analytical methods used to study GHG emissions and climate impact of food consumption and production in this thesis, is provided in Figure 11.

4.3 Land use and land use change

4.3.1 General description

In the terminology, a distinction can be made between land cover and land use. Land cover is defined as "the observed (bio) physical cover on the earth's surface, including the vegetation (natural or planted) and human constructions (buildings, roads, etc.) which cover the earth's surface" (Di Gregorio, 2005). This differs from land use, which refers to human arrangements, activities and inputs undertaken in a certain type of land cover to produce, change or maintain it (Koellner et al., 2013).

Land use impacts of and on the agrifood system can further be categorized as impacts due to land occupation and land transformation (i Canals et al., 2007; Udo de Haes et al., 2002). Land occupation refers to when an area of land is used in the intended productive way and the properties of the land are maintained, whereas land transformation, or land use change, means that the properties of the land are modified to make it suitable for an intended use, for example deforestation to establish arable land (Koellner et al., 2013b). Land use change can further be categorized as direct or indirect land use change (Ahlgren & Di Lucia; 2014, Berndes et al., 2013).

Land has many functions, for example as habitat for human and non-human life and by the provision of ecosystem services including cycling of nutrients, water and carbon. In addition, how land is used and for what purpose has both economic and aesthetic implications (i Canals et al., 2007). Due to the multi-functionality of land, the choice of metrics and indicators to assess sustainability of land use is value-dependent and varies between assessments. The complexity is further increased because impacts of land use are highly dependent on the intensity, duration and site-specific bio-geographical conditions of the land used.

The many different functions of land, effects of land use and parameters influencing the ecological impact of land use demand, distinguishes land use from other environmental impact categories and makes the assessment of the total impact of the land use challenging (Bare, 2011). For a holistic perspective, land use impact assessments need to include both spatial dimensions such as the surface area occupied and temporal dimensions such as the duration of a certain land occupation or land transformation process. Of importance are also qualitative aspects including the type, intensity and location of the land being occupied (i Canals et al., 2007; Koellner et al., 2013a; Lindeijer et al., 2002).

Land occupation is in general quantified in surface-time units (e.g. ha year^{-1}) representing a certain area (e.g. 1 ha) of a given type of land (e.g. arable land) used over a certain time period (e.g. 10 years) (i Canals et al., 2007). Land occupation for a specific food is further determined by combining information on the land use demand

of the food and the amounts consumed. By analogy, per capita land use demand for food is quantified as the sum of the consumption of food items (kg capita^{-1}year^{-1}) divided by general or site-specific crop yield data of the production per area harvested (ha kg^{-1}year^{-1}) (Gerbens-Leenes & Nonhebel, 2002; Kastner, 2012). Land use demand is further dependent on the cropping intensity, i.e. the ratio between net sown area and gross cropped area, indicating the percentage share of the area sown more than once per year. Quantifying the land use demand of animal-based food introduces another level of complexity which requires information on the amounts of feed used and the yields of the feed crops in question (Gerbens-Leenes & Nonhebel, 2002), as well as a division between crop and pasture land of different qualities.

In contrast to land occupation, land transformation or land use change is measured in surface units (e.g. 1 ha of grassland converted into arable land) (i Canals et al., 2007). Land use change can either refer to the conversion of natural ecosystems into managed lands, or changes in the management of already appropriated land, such as if former pasture land is brought into cultivation or if agronomic practices on cultivated land are changed (Berndes et al., 2013). As previously mentioned, a distinction can be made between direct and indirect land use change. Land use change that is directly associated with a specific activity is referred to as direct, whereas indirect land use change refers to changes in land use that take place elsewhere as a consequence of a specific land use activity. An example of indirect land use change is if the expansion of crop A leads to crop A occupying land on which previously crop B was cultivated which therefore has to be cultivated elsewhere by converting natural ecosystems into arable land (Ahlgren & Di Lucia, 2014; Berndes et al., 2013).

To predict the area, type and location of land affected by indirect land use change, many parameters, e.g. price elasticity (indicating how sensitive price is to changes in supply and demand) and transformation elasticity (indicating the ease of converting one type of land to another), need to be accounted for (Broch et al., 2013). Due to the complexity of such calculations, the area of land affected by indirect land use change is quantified by using agro-economic equilibrium models. Because indirect land use change is a relatively new research field, methods to account for indirect land use change are inconsistent and associated with large uncertainties (Ahlgren & Di Lucia, 2014; Broch et al., 2013).

As the amount of inputs (e.g. fossil fuels, water, pesticides, nutrients) is related to the area of land used for agriculture, land use demand can serve as an indicator of several ecological and environmental impact categories (e.g. climate impact, water demand, biodiversity, eutrophication) (Figure 11). Using smaller areas of land is, however, not necessarily an indicator of reduced ecological or environmental burden. For example, use of less land area could be the result of increased intensity in agriculture, which may be more damaging when considering impacts on, e.g. biodiversity and soil quality. To fully understand the impact of land occupation and land transformation, thus requires

more comprehensive assessment methods. Land use impacts from occupation and transformation of land is in general calculated as the difference observed for a selected impact category between the current land use and a reference scenario (i Canals et al., 2007). To enable such assessments requires firstly a decision on which sustainability indicators to measure and secondly access to relevant characterization factors for the specific type of land use and region (Koellner, 2007).

Over the past decades, several indicators to assess the impact of land use have been suggested, including an array of proxies for land quality, e.g. biodiversity, erosion, salinization, microbial biomass and diversity, soil organic matter, carbon deficit and soil quality, and efforts to link land use to ecosystem services, deforestation dynamics and GHG emissions (Bare, 2011; Brandão & i Canals, 2013; i Canals et al., 2007; Koellner, 2007; Koellner et al., 2013b; Lindeijer et al., 2002). One of the many complexities in estimating land use impacts is the fact that ecological effects may extend for a long time after the study of the use of the land (i Canals et al., 2007). Another methodological challenge is how to allocate the total damage of a series of land use activities (Brandão & i Canals, 2013; Koellner, 2007). Although great efforts have been made to develop a methodological framework to account for the ecological and environmental effects of land use, land use impacts are often not included in LCA (Brandão & i Canals, 2013; i Canals et al., 2007).

4.3.2 Research approach

The demand for land for food production is quantified in **papers II** and **IV.** Land use demand is defined as the land area occupied by the food consumed per capita per year. Land use demand is assessed as the total land area occupied by agriculture, without any distinction between different types of agriculture land. Land use demand is assessed as a natural resource, rather than a proxy for environmental impacts.

Per capita land use demand for food consumption is evaluated against a theoretical limitation for global crop land expansion set to 2000 Mha or 15% of the global land surface (Rockström et al., 2009). Based on this and assuming a world population of nine billion by 2050 (UN, 2010b), the available area for long-term, sustainable cropping was estimated to be 0.22 ha capita^{-1} year^{-1}. An overview of the metrics, indicators and analytical methods used to study the land needed for the cultivation of food in this thesis, is provided in Figure 11.

4.4 Nutritional quality and health

4.4.1 General description

Unlike assessments of environmental and ecological impacts, the nutrition and health effects of food and diets require a consumption-oriented methodology. As described in **paper I**, the term food consumption can have several meanings. For example, the average food consumption pattern of a population is often used as a proxy for per capita food supply, per capita food purchase, or per capita food intake. Depending on the type of food consumption referred to, the underlying data can be derived from several sources, including agricultural supply data or food balance sheets, household budget surveys and individual dietary surveys (Hallström & Börjesson, 2013). When studying nutritional and health effects of food and diets, data on the amount of food actually eaten by individuals and groups is the preferred measure. However, due to economic and other resource constraints, and lack of access to other data, per capita food supply and food purchase data are often the basis for nutritional and health impact assessments (Hallström & Börjesson, 2013).

Nutritional quality of food and/or food consumption patterns (e.g. diets) are judged based on how the nutritional content of the foods consumed affect the body´s functions, and its associated short- and long-term risk of disease (NCM, 2014). The requirement of total energy can be assessed, on an individual level, by the use of technical measurements (i.e. doubly labelled water), or by the use of factorial methods in which the energy requirement, for an individual or population, is estimated based on the resting energy expenditure and physical activity level. The nutritional intake of an individual or a population is, generally, estimated by multiplying the amount of food consumed by its nutritional content (NCM, 2014). Associations between food intake, health and disease are often assessed by using nutrition epidemiology.

To quantitatively measure the association between nutrition and health, and identify cause-effect relationships is very complex, time- and resource-demanding. Depending on the method used, the nutritional quality of food, meals and diets can be estimated and evaluated against nutritional and dietary guidelines or recommendations. New findings and knowledge from such assessments (e.g. epidemiological studies) are thereafter compiled and assessed in reviews and meta-analyses (5.1) to create and establish evidence-based theories, knowledge and recommendations on nutrient, food, and diet-disease relationships (NCM, 2014). In addition, a large and growing number of nutrient, food and diet quality scores, indices, scales and rankings, have been developed to assess dietary quality (Kourlaba & Panagiotakos, 2009; Ocké, 2013; Waijers et al., 2007).

Dietary guidelines and recommendations are intended for healthy individuals and populations and their main objective is to promote a diet that provides energy and nutrients for long-term health (NCM, 2014). Three types of nutrition recommendations exist; recommendations for intake of macro-nutrients, micro-nutrients, and food-based dietary guidelines (Table 7).

Table 7. Overview of Nordic nutrition recommendations for adults[1]

Types of nutrition recommendations	Refer to intake of[2]	Recommended intake levels
Recommended intake of macro-nutrients (i.e. energy providing nutrients)	Total fat	25-40 E %[3]
	Saturated fatty acids	< 10 E %[3]
	Trans-fatty acids	As low as possible
	Total carbohydrates	45-60 E %[3]
	Fiber	25-35 g/d
	Added sugars	< 10 E %
	Total protein	10-20 E %[3]
Recommended intake of micro-nutrients (i.e. vitamins & minerals)	Vitamin D	10-20 µg/d
	Folate	200-500 µg/d
	Vitamin B_{12}	2.0-2.6 µg/d
	Iron	7-15 mg/d
	Zinc	7-12 mg/d
	Selenium	40-60 µg/d
Food-based dietary guidelines (i.e. specific foods and food patterns)	Vegetables, fruit & berries	500 g/d
	Whole grains	70-90 g/d
	Red and processed meat	< 500 g/week
	Salt	6 g/d

[1]Based on NCM (2014) and SFA (2015b). [2]A selection of nutrients. [3]E %= percent of total energy intake.

To evaluate the probability of an adequate nutritional intake, intake levels of minerals and vitamins are evaluated against reference values of the lowest intake level, average requirement and/or recommended intake of nutrients. In addition, there are reference values on upper levels of intake for some nutrients and foods (Table 8). The concept 'requirement' is generally defined as "the absorbed amount of a nutrient that is needed to prevent clinical deficiency symptoms, or as the amount that maintains satisfactory body stores and tissue function" (NCM, 2004). Usually, a 'requirement' of a nutrient is interpreted as "the smallest amount of a nutrient that is needed to prevent all physiological signs of insufficient nutrition that can be attributed to an insufficient supply of that nutrient" (NCM, 2004).

Table 8. Overview of existing nutrition quality metrics

Nutrition quality metrics	Description of metrics
Lowest intake level	Intake level that may lead to nutrient deficiency in some people within a defined population.
Average requirement	Intake level representing the average requirement of a defined population.
Recommended intake level	Intake level representing the estimated requirement to maintain good nutritional status and health among practically all healthy individuals.
Upper intake level	Intake levels, for some nutrients, representing the maximum level judged to be unlikely to pose a risk of adverse or toxic health effects

The percentage of a population that has an intake below the average requirement indicates the proportion at increased risk of inadequate nutrient intake. In order to account for individual variations, a safety margin of usually two standard deviations or more is added to the recommended intake levels (NCM, 2014). In reality, there is no fixed point for nutritional requirements that can be seen rather as transitional stages between estimated minimum and optimal intake levels (Figure 12) (NCM, 2014).

The nutritional content of the dietary intake is the most common indicator of the nutritional quality. In addition, it is often used as an indirect indicator to evaluate the effects on nutritional status, on the physiological consequences and the health effects of the diet. However, to use the contribution of nutrients from the diet as an indicator of nutritional and/or health status is a simplification of reality for many reasons. Firstly, as discussed in **paper I**, methods to estimate and assess dietary and nutritional intake are hampered by many uncertainties. Secondly, there may be a difference between the nutrient content in raw and cooked food. Thirdly, there is a difference between intake and uptake of nutrients from the diet. Finally, the nutrient content of food varies from food to food within the same food group depending on, for example, regional differences. How these aspects are handled and accounted for is further discussed in section 5.7 and chapter 7.1.

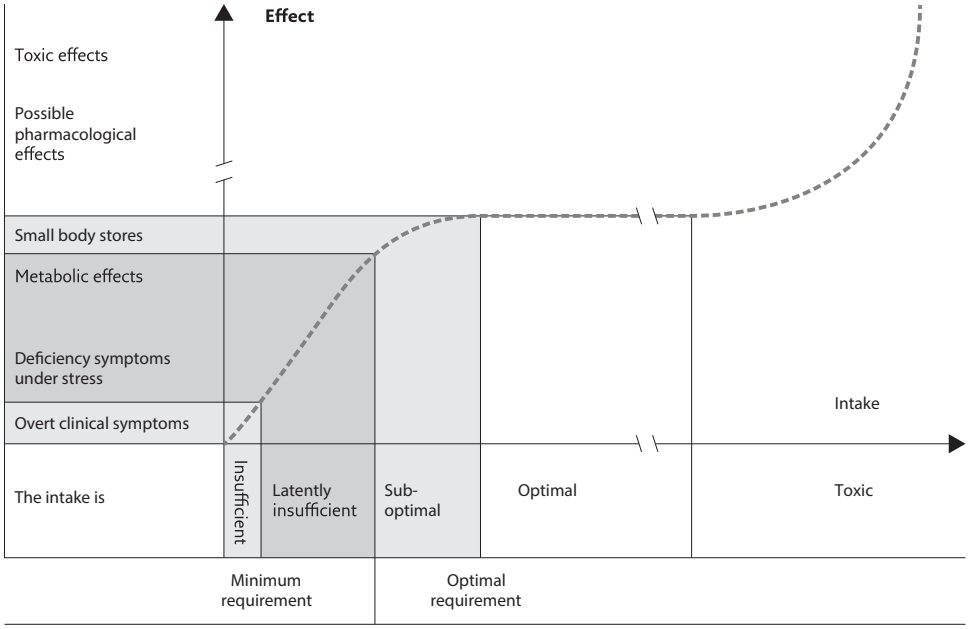

Figure 12. The theoretical relationship between intake of a nutrient and the effect on the organism. Adapted from (NCM, 2014).

As mentioned earlier, several indicators have been and can be used for the assessment of nutrition and health characteristics or quality of the diet. The most common indicator used is the 'daily energy intake' or indicators based on specific nutrients, such as 'dietary iron', 'fiber' or 'saturated fat' (Lukas et al., 2015). Another approach is to use dietary quality scoring methods which in general are based on existing nutrition recommendations (Table 8), or to adhere to specific dietary patterns considered healthy (Chiuve et al., 2012). For example, the Healthy Eating Index quantifies the quality of diets based on their compositions and on their adherence to the U.S. dietary guidelines (Kennedy, 1995). To validate dietary quality scores, they can be related to the nutrient adequacy and health outcome (Waijers et al., 2007). Higher scores of the Healthy Eating Index are suggested to be associated with a lower risk of several chronic diseases (Chiuve et al., 2012).

The use of nutrient profiling is another method to assess the quality of foods, meals and diets (Azais-Braesco et al., 2006; Drewnowski & Fulgoni III, 2008; Garsetti et al., 2007). The typical objective of nutrient profiling is to provide a quantitative scoring scheme for the quality of foods based on their nutritional contribution to the overall diet (Heller et al., 2013). In other words, nutrient profiling rates the quality of individual food items or dietary patterns based on their content of micro- and macro-nutrients and their associated (positive or negative) health effects. As an example, the Nutritional Quality Index is based on an algorithm incorporating over 30 micro- and

macronutrient food properties which are weighted based on their nutritional and health effects (Heller et al., 2013). Weighting coefficients can for example be based on the prevalence, severity, and strength of the association between dietary factors and risk of chronic disease (Heller et al., 2013; Katz et al., 2009). An alternative approach to validate the nutritional quality of food and diets is using biochemical indicators (Willet et al., 2013).

Quality of a diet, based on its dietary and/or nutritional composition, can thus be evaluated against set recommended intake levels of foods and/or nutrients. To measure and evaluate the health effect of dietary factors, in general, epidemiological methods are used. The health effect of a nutrient or food can, for example, be assessed by comparing the prevalence of disease in groups of populations exposed to different nutrients, food or diets. The difference in disease prevalence can be used to calculate the relative risk of disease associated to different dietary factors. A more thorough description of epidemiological methodology and metrics used to express the risk of disease is provided in section 5.6. Health effects from an exposure can further be measured by the use of metrics expressing the estimated burden of the disease associated with the exposure, within an individual or population. Examples of such metrics are; years of potential life lost, healthy life expectancy, disability-free life expectancy, quality-adjusted life years, and disability-adjusted life years (Bonita, 2010).

4.4.2 Research approach

In this thesis dietary quality is assessed based on the nutritional contribution (e.g. estimated nutrient intake) of the dietary scenarios studied and their estimated associated health effects. The dietary scenarios studied in **papers II** and **III** are based on food-based dietary guidelines for healthy meat and overall food consumption (Table 8).

In **paper II**, the nutrient intake from dietary scenarios is quantified by the use of nutritional calculation (section 5.5). The nutritional quality of the diet and the effect on nutritional status are further analyzed based on the contribution of energy, protein, total fat, saturated fat, iron and zinc from meat consumption. The contribution of nutrients (e.g. estimated nutrient intake) in each scenario studied is further evaluated against average recommended daily intake levels of nutrients among healthy adults in Sweden (NCM, 2004).

In **paper III**, the nutritional quality of the dietary scenarios studied is evaluated by the use of nutrition epidemiology (section 5.6), as the estimated effect, on the relative risk of coronary heart disease (CHD), type II diabetes (T2D) and colorectal cancer (CRC), of the respective scenario. An overview of the metrics, indicators and analytical methods used to study the nutrient intake, nutritional quality and health effects of food consumption in this thesis is provided in Figure 11.

5 Research Methodology

5.1 Overview of research methodology

This chapter describes the main methods used in the thesis. The chapter includes an overview of the methods used (5.1), and descriptions of a review analysis (5.2), scenario analysis (5.3), life cycle assessment (5.4), nutrient calculation (5.5), nutrition epidemiology (5.6), and Monte Carlo analysis (5.7). The sections are divided into two parts, the first proving a general description of the methods, and the second describing how the methods are used in **papers I-IV.**

Table 9 provides an overview of the methods used in this thesis.

Table 9. Overview of impact categories studied and methods used in the thesis

PAPER	IMPACT CATEGORY STUDIED				METHOD USED[1]					
	Climate impact	Land use demand	Nutritional quality	Risk of disease	R.A	S.A	LCA	N.C	N.E	M.A
I					X					
II	X	X	X			X	X	X		X
III	X			X	X	X	X	X	X	X
IV	X	X			X					

[1]R.A=Review analysis, S.A=Scenario analysis, LCA= Life cycle assessment, N.C=Nutrient calculation, N.E=Nutrition epidemiology, M.A=Monte Carlo analysis.

5.2 Review analysis

5.2.1 General description

Research should preferably be based on the best available evidence or knowledge. The increasing availability, accessibility and flow of information in society puts an increasing pressure on the method used to retrieve research information. Review analysis is a method used to compile and summarize knowledge on a particular research topic (Akobeng, 2005).

Because the quality of research methods varies and the results of individual studies may be contradictory, it is preferable to use several studies when assessing a research topic. By the use of review analysis several studies can be brought together to establish whether scientific findings are consistent and generalizable, and to evaluate the strength of the evidence (Akobeng, 2005). The use of aggregated data from several studies also makes it possible to evaluate the effect of bias, which in turn improves the reliability of the results of the review analysis compared to the results of the individual studies (Paul & Leibovici, 2014).

Is treatment A better than treatment B? Does intake of nutrient/food C influence the risk of disease D? These questions are examples of topics that can be addressed by review analysis in the field of medicine and nutrition (Mente et al., 2009; Paul & Leibovici, 2014). Review analysis can also be used to analyze other topics, such as research theories and/or methodologies in different fields (Aboelela et al., 2007; Drewnowski & Fulgoni III, 2008; Heller et al., 2013) and to perform data compilation of, for instance, the environmental impact of different food items (de Vries & de Boer, 2010; Nijdam et al., 2012).

A methodological distinction can be made between narrative- and systematic review analysis (Rys´, 2009). Narrative review analysis, the traditional way of performing reviews, uses informal methods to collect and interpret information, and often provides a qualitative summary of the topic studied (Rys´, 2009; Uman, 2011). In contrast, systematic review analysis generally uses formal and explicit methods to limit bias in the assembly of literature and the quantitative synthesis thereof (Cook, 1997, Rys´, 2009). The method used in systematic reviews includes a defined research question, a comprehensive literature search with predefined selection criteria, a critical quality assessment of the literature, and predetermined methods to synthesize results and draw evidence-based conclusions (Akobeng, 2005; Cook, 1997; Paul & Leibovici, 2014).

In interdisciplinary research, narrative review analysis is commonly used due to the difficulty to systematically addressing topics assessed from different perspectives and by different methods, especially if they are descriptive (i.e. qualitative analysis). In the field

of medicine, systematic review analysis is considered the preferred method, due to the lower risk of bias (Akobeng, 2006).

Following a systematic review analysis, results of individual studies can be pooled and, by the use of statistics, recalculated into a single measure of the estimated average effect of an intervention. This method is called meta-analysis (Paul & Leibovici, 2014; Thacker, 1988). By pooling results from several studies, the sample size increases which both improves the precision of the result and enables statistical analyses (Akobeng, 2005; Thacker, 1998). Conducting a meta-analysis involves two main steps. Firstly, the effect of the studied intervention is calculated, with its 95% confidence interval, for each of the studies included. Secondly, the overall effect of the intervention is calculated based on a pooled analysis of all the studies combined. The overall effect of an intervention is generally calculated using weighted averages, with greater weight given to studies of higher quality and completeness (Akobeng, 2005; Rys´, 2009). The final results gained from meta-analyses, expressing the statistical difference between the intervention- and control group (Rys´, 2009), are often presented graphically in a forest plot (Figure 13).

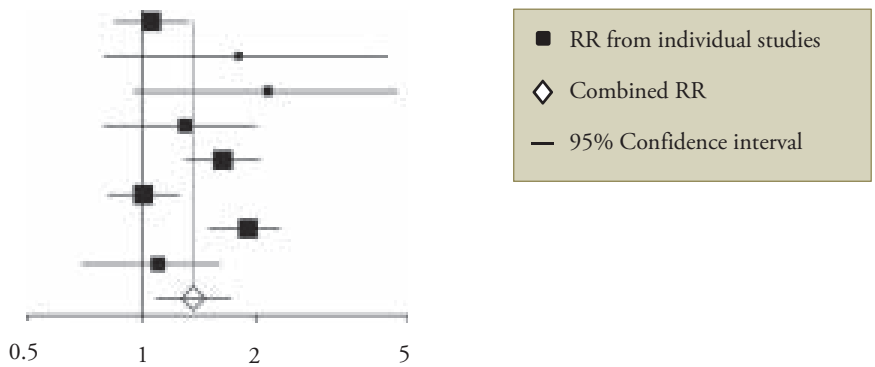

Figure 13. Forest plot of red meat intake (more frequent than once a day) and relative risk of colon cancer development. Combined Relative risk=1.27, 95% CI 1.09-1.71), based on the results of eight studies. Figure and data adapted from (Smolinska & Paluszkiewicz, 2010).

5.2.2 Research approach

In this thesis, review analysis is performed in **papers I, III and IV**. Narrative review analysis is performed in **papers I** and **III**, whereas systematic review analysis is performed in **paper IV**.

In **paper I**, the topic reviewed is meat consumption statistics. The review includes data and information from scientific articles, reports on statistics, online databases, and personal communications with authorities in the field. Information and data identified are further analyzed, processed and categorized based on the type of survey method (i.e. whether the data are based on agricultural supply, household-budget surveys, or individual dietary surveys), the type of meat data (i.e. whether meat-consumption data refer to carcass weight or bone-free weight, are adjusted for food losses and waste, refer to raw or cooked meat, include or exclude non-meat components in mixed-meat products and prepared meals), and the type of statistical sources (i.e. whether meat-consumption data are taken at the national, regional, or international level).

In **paper III**, the relationship between food categories and risk of disease is reviewed. The review was performed in the NCBI Pub Med database in March 2014, using as keywords different food groups, e.g. "vegetables", "fruit", "fruit and vegetables" "meat", "processed meat" etc., and non-communicable diseases; coronary heart disease, hypertension, type II diabetes, and a range of cancers. Articles included in the review were peer reviewed meta-analyses of prospective cohort and randomized controlled trial studies, published between 2005 and 2014, that provide relative risk estimates with 95% confidence intervals. The review analysis was the basis for the selection of which food categories and diseases to analyze further.

In **paper IV**, a systematic review analysis of dietary scenarios, assessing the GHG emissions and land use demand of diets, is performed. For improved quality and reduced risk of bias, the study design of the systematic review follows the PRISMA Statement protocol (Moher et al., 2009). The literature search was performed in February 2014 with the use of Web of Knowledge (ISI), Scopus and Google Scholar. To assess the effect of human dietary change on GHG emissions and land use demand, the terms: 'diet' or 'food' and 'scenario' were combined with the terms 'climate' or 'greenhouse gas' or 'land' or 'sustain'. In addition, related and relevant articles found in reference lists were reviewed (Figure 14).

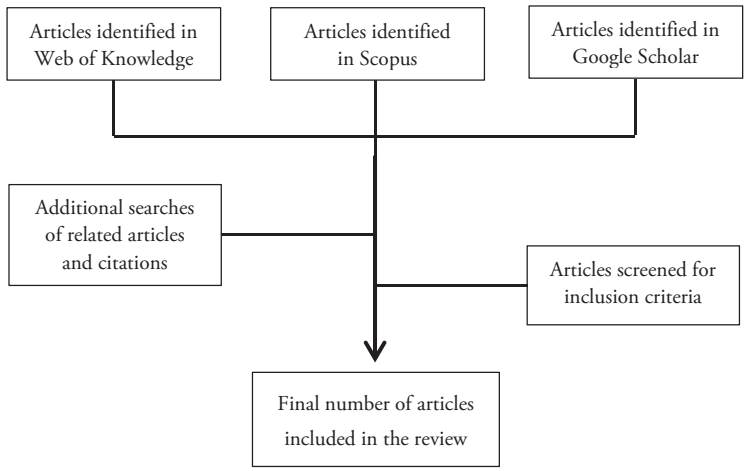

Figure 14. Approach for literature search and selection in paper IV

Articles included in the systematic review of **paper IV** met the following six inclusion criteria: i) English-language publications; ii) published between 2005 and February 2014; iii) dietary scenario analysis is performed for a complete diet; iv) quantitative estimates of the effect on GHG emissions and/or land use demand of human dietary change are provided; v) published in peer-reviewed scientific journals; vi) results are compared against reference scenarios of current (1990-2010) average food consumption of a population. The inclusion criteria were set to increase the comparability between studies, to capture the effect of dietary change in the current food system and to ensure that the articles included were of acceptable quality. The selection of articles that met the inclusion criteria was based on information available in the titles and abstracts of the articles. In total, 14 articles that fulfilled the inclusion criteria are identified.

The methodological aspects reviewed in **paper IV** were chosen as they were identified as key parameters having major impacts on the GHG emissions and land use demand of the diet, and thereby on the overall results and quality of dietary scenario analysis.

5.3 Scenario analysis

5.3.1 General description

The use of scenarios, to study a possible future course of events and the consequences of strategic decisions, dates back to the fifties when Herman Kahn at Rand Corporation introduced the method for use in military planning (Kahn & Weiner, 1967). Since then, scenario analysis have been used for a wide range of purposes in both the private and public sector, not least as a common and useful tool in environmental science and policy (Rothman, 2008). The GHG emission scenarios (SRES) set up by the IPCC (IPCC, 2014) and World Energy Outlooks published by the International Energy Agency (IEA, 2014), are examples of how scenario analysis is used in the context of sustainability.

There are different definitions of the meaning of scenarios. According to the IPCC, scenarios can be described as "images of the future, or alternative futures that are neither predictions nor forecasts" (Nakicenovic, 2000). The definition of scenarios by the United Nations Environmental Programme is "descriptions of journeys to possible futures reflecting different assumptions about how current trends will unfold, how critical uncertainties will play out and what new factors will come into play" (UNEP, 2002). Scenario analysis is further described as the process of developing scenarios, comparing their results and evaluating their consequences (Alcamo, 2009).

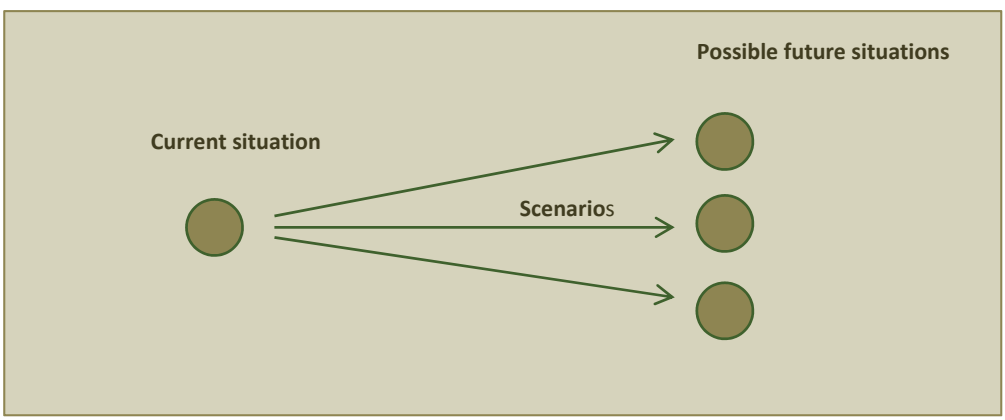

Figure 15. Schematic illustration of scenario analysis.

The approach for performing scenario analysis varies. Scenarios can be quantitative or qualitative, of simple or of very complex character to be analyzed in computer models (Glenn & Gordon, 2003). Several other distinctions can be made between the type of

scenarios studied (Börjeson, 2006). Börjeson et al. (2006) distinguishes between three types of scenario studies, predictive, explorative and normative scenarios. Predictive scenarios typically answer the questions "what will happen…?" and are useful for planners and investors to explore the consequences of future trends or measures. Explorative scenarios respond to the question "what can happen" and aim to explore the consequences of possible alternative future developments and can, for example, be used by policy makers to analyze the effect of strategic decisions. Finally, normative studies analyze how a specific target can be reached and can, for example, be used to develop a strategy for how to reach environmental or health goals.

Although no standard procedure exists for scenario analysis, four steps are typically included; clarification of purpose and structure (e.g. defining the scope, target and indicators); laying the foundation of the scenarios (e.g. identifying drivers, uncertainties and creating the scenario framework); scenario development and assessment; and communication and spreading of results (UNEP, 2013b). To evaluate the effect of the different scenarios the results are often compared to a reference or baseline scenario, reflecting, for example, the current situation or the estimated future situation based on current trends. A more thorough description of theories and methodologies used in scenario analysis is provided in Alcamo (2009).

Scenario analysis is also used to study the effects of consumption on the environment and society, for example, to evaluate the environmental or health impact of different dietary patterns. In dietary scenario analysis, alternative diets, varying in quantity and composition of food and/or production method and origin, are typically assessed. Dietary scenarios can be developed based on registered or hypothetical diets. For example, registered dietary data can be used to develop reference scenarios, reflecting the average food consumption pattern within a specific population. Compared to hypothetical dietary scenarios, dietary scenarios based on registered consumption data have the advantage of being realistic, not only in theory, but in practice. On the other hand, as discussed in **paper I**, dietary surveys and food consumption data may be hampered by a large uncertainty. By assessing hypothetical dietary scenarios, any food consumption patterns, realistic or not, can be investigated.

As previously described scenario analysis can be used to assess possible developments in the future. By using future scenarios, the effect of technology and product development, population growth and other influencing parameters and trends can be accounted for in the assessment. However, it is also possible to analyze present situations, for example, to assess the current food system under different, alternative circumstances.

Methodological aspects of particular importance for dietary scenario analysis include the choice of functional unit, system boundaries, and the approach to assess and account for food waste and other uncertainties in the data and results (Hallström et al., 2015). These aspects are further discussed in **papers II, III** and **IV**, as well as in section

5.4. A challenge in dietary scenario analysis is the development of scenarios that are comparable also from a nutritional perspective. This is further discussed in section 5.5.

5.3.2 Research approach

In this thesis, scenario analysis is used in **papers II, III** and **IV**. In **papers II** and **III,** the impact of dietary change is analyzed by the use of exploratory quantitative scenario analysis. The scenarios developed and analyzed in **paper II** and **III** represent a near-time perspective, meaning that food production systems correspond to today´s performance, without any assumptions on future technical development. In **paper IV**, dietary scenarios in the articles reviewed are compiled, categorized and further assessed.

Table 10 provides an overview of the scenarios developed and/or assessed in **paper II-IV**. More details about the scenarios studied in **papers II**, **III** and **IV.**

In **paper II**, the scenarios analyzed represent three variants of meat consumption in Sweden. The reference scenario (REF) represents the current per capita meat consumption in Sweden (2009), while NUTR-1 and NUTR-2 are hypothetical scenarios, in which the amount and type of meat correspond to Swedish and international dietary guidelines and recommendations for healthy meat intake (Enhardt Barbieri & Lindvall, 2003; WCRF/AICR, 2007). In NUTR-2, the type of meat consumed is, in addition, adjusted to optimize land use efficiency (Table 10).

In **paper III,** the scenarios analyzed represent four alternative food consumption patterns in the US. The reference scenario (SAD), reflects the standard US American diet, i.e. the per capita US food consumption in 2012, while HAD-1, HAD-2 and HAD-3 are hypothetical scenarios of potentially healthier American diets, developed based on American and international dietary guidelines and recommendations (USDA, 2010; WCRF/AICR, 2007) (Table 10).

The assessment of US food consumption patterns includes only a portion of the total diet. The foods groups analyzed meet the following criteria; USDA dietary recommendations are consistent with international nutrition and health authorities, ii) documented GHG emission estimates are available, iii) evidence of diet-disease relationship based on relative risk estimates of high quality are available. The food groups analyzed are red and processed meat, vegetables, fruits and berries and grains.

In **paper IV,** scenarios in the reviewed articles are compiled, analyzed and categorized, based on dietary composition, into healthy diets, diets in which meat is partially replaced by plant-based food, mixed foods or dairy products, diets in which ruminant meat is preplaced by pork and poultry, vegetarian diets, vegan diets and diets with balanced energy intake (Table 10).

Table 10. Scenarios developed and assessed in this thesis.

Article	Scenarios	Scenario descriptions
Paper II		
	REF	Reflects current (2009) per capita consumption of meat in Sweden.
	NUTR-1	Total meat consumption is limited to 126 g uncooked, pure meat per day, as recommended by Swedish dietary guidelines. Consumption of red meat is restricted to 60 g (uncooked weight) per day (50% beef, 50% pork) and consumption of charcuteries is reduced to zero, which corresponds to the public health recommendation by the World Cancer Research Fund (max 300 g cooked red meat per week, avoid processed meat).
	NUTR-2	As in NUTR-1, the total meat consumption in this scenario is limited to 126 g uncooked meat per day and the intake of charcuteries is reduced to zero. Beef comes entirely from production systems that produce both milk and meat, which are more resource efficient than systems producing only meat. The supply of beef at farm gate is restricted to 14 g per day, an amount corresponding to the Swedish dietary guidelines for dairy consumption (0.5 litres of milk eq. per person and day). The remaining amount of meat is assumed to be consumed as chicken since chicken has a high efficacy converting feed to meat, and thereby uses cropland more efficiently than beef and pork.
Paper III		
	SAD	Reflects current (2012) per capita consumption of red meat, processed meat, vegetables, fruit, legumes and cereals, in the US.
	HAD-1	Intake levels of fruits, vegetables, red and processed meat, whole grains and refined grains are based on USDA dietary recommendations. Processed meat is limited to 20% of total red meat (no consumption of white processed meat), based on the recommendation by the WCRF that processed meat should be limited as much as possible. Whole grains and refined grains contribute 60% and 40% respectively, of total grain intake, based on the USDA recommendation that at least half of the grain consumption should come from whole grains. Fruit juice is limited to 20% of total fruit consumption, based on the USDA recommendation that the major part of fruit intake should come from whole fruits.

	HAD-2, HAD-3	Intake levels in HAD-2 and HAD-3 are the same as in HAD-1, with the exception that consumption of red and processed meat is further reduced and replaced by increased intake levels of beans and peas. Red and processed meat intake is reduced to 25 g of cooked meat per day in HAD-2 and to zero in HAD-3. Replacement of meat with plant-based protein is based on a USDA framework in which the nutritional interchangeability of plant-based and animal-based food is estimated.
Paper IV		
	Healthy	Omnivorous diets based on dietary guidelines.
	Balanced energy intake	Diets in which the dietary composition is unchanged and the caloric content is reduced to recommended levels.
	Meat partially replaced by plant-based foods/mixed foods/dairy products	Diets in which a proportion of all or a specific type of meat is reduced in favor of either plant-based foods, a mix of non-meat food groups or dairy products.
	Ruminant meat is replaced by pork and poultry	Diets in which a proportion or all ruminant meat (e.g. beef) is replaced by monogastric meat (e.g. pork and poultry).
	Vegetarian diets	Diets in which all meat is replaced by non-meat food groups.
	Vegan diets	Diets in which all animal-based products are replaced by plant-based food.

5.4 Life cycle assessment

5.4.1 General description

Life cycle assessment (LCA) is a methodological framework for calculating the environmental impact of a product, process or service for all stages throughout its life cycle (ISO, 2006a; 2006b). The life cycle concept implies a holistic approach by which the whole system, "from cradle to grave", is included in the assessment rather than individual parts being studied separately (Figure 16).

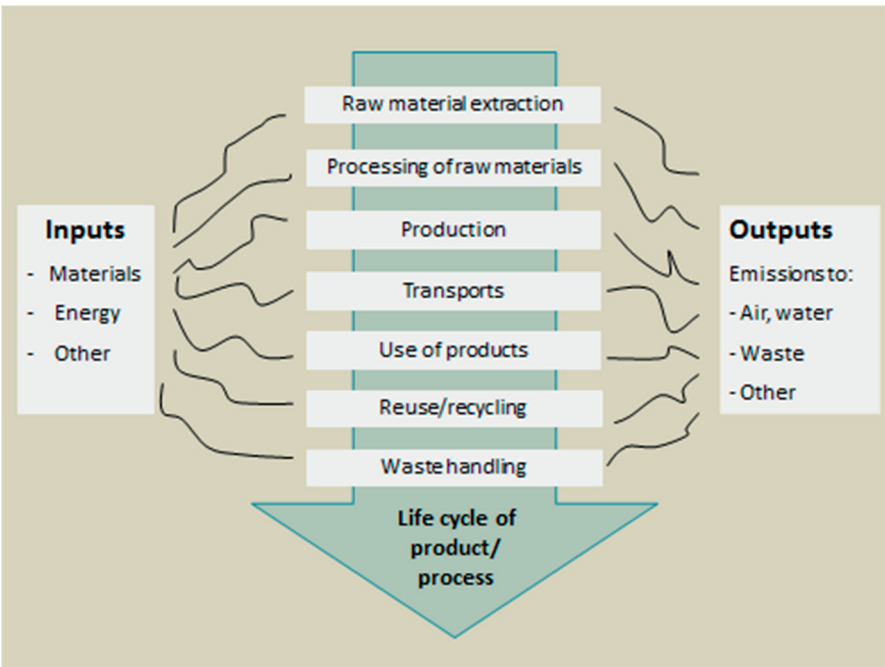

Figure 16. Schematic illustration of the life cycle assessment methodology. Courtesy of Linda Tufvesson.

In 1969 Coca Cola was among the first to use the life cycle concept to explore the resource and environmental profiles of different packaging materials for their products (Hunt, 1974). In the 1980´s a framework for LCA was developed by the Society for Environmental Toxicology and Chemistry (SETAC) and in 1997 the methodology was standardized by the International Organization of Standardization (ISO) (Huppes, 2012).

Life cycle assessment can be used to explore the environmental improvement potential of products and processes, in decision making, for example for strategic planning and prioritizing in industry, for marketing, such as eco-labeling of products, and for comparing the environmental impact of different products or processes with similar functions (ISO, 2006a). Figure 17 illustrates how LCA can be used to assess the total environmental impact embodied in a product or process, as well as the stages of the life cycle that contribute most and least.

In the research field of sustainable food, LCA has evolved from primarily being used to analyze the environmental impact of separate foods, to quantify the impact of complete meals and diets. In several studies, the environmental impact of food is quantified for an entire nation or region by combining LCA food data with production and/or consumption statistics for the given population (Berners-Lee et al., 2012; Tukker et al., 2011).

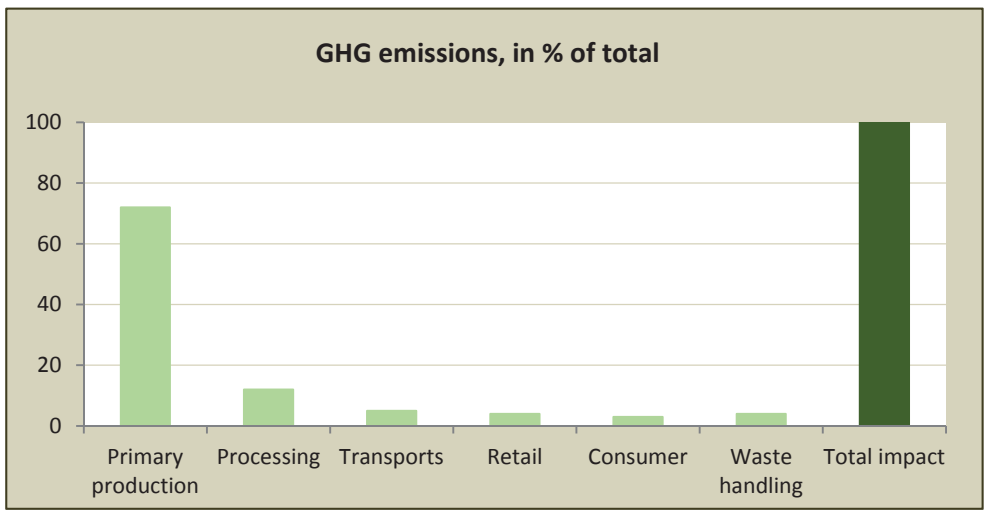

Figure 17. Schematic illustration of GHG emissions from a product or process, expressed per stage in the food chain, in % of total emissions from the entire life cycle.

According to the ISO standards 14040 and 14044, a LCA consists of four steps (Figure 18) (ISO, 2006a; 2006b). Step I, is the goal and scope definition which describes the purpose of the study and for what the results are to be used. The object and system to be studied are specified by defining the functional unit and system boundaries. In step II, the life cycle inventory (LCI) analysis, data are collected to quantify the amount of resources used and emissions produced in activities within the system studied. The environmental impact of the system studied is evaluated in step III, the life cycle impact assessment. Here, the LCI data are classified into different impact categories (classification) and thereafter the relative contribution to each type of environmental

impact is calculated (characterization). Carbon dioxide, methane and nitrous oxide will, for example, be classified as GHGs and converted into CO_2 eqs. Step IV, is the interpretation phase which is a phase that runs parallel with the other phases and aims to analyze the results, evaluate the limitations of the study, draw conclusions and make recommendations. In this step uncertainty and sensitivity analyses can be performed to assess the uncertainty of the input data and the reliability of the results.

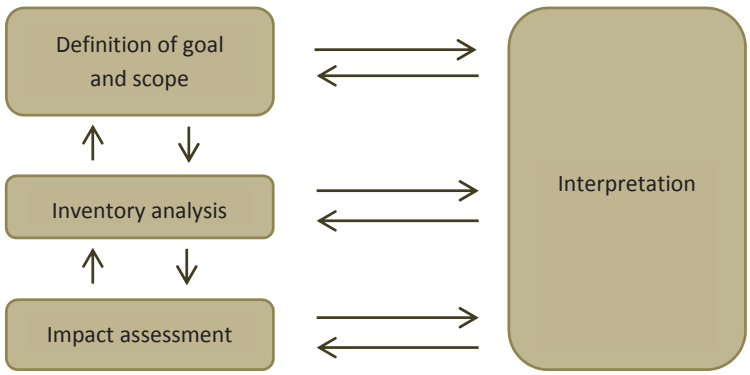

Figure 18. Phases included in life cycle assessment, adapted from ISO (2006b)

There are several types of LCAs. Firstly, a distinction can be made between process-based and input-output LCA (IO LCA). Secondly, LCAs can be divided into accounting LCA, also called attributional LCA, and consequential LCA.

Process-based LCA is a bottom-up approach in which the use of the resources and emissions produced from each process in the system studied is calculated in isolation and thereafter added together as a sum for the whole system (Baumann, 2004). In contrast, I-O LCA is a top-down approach in which the traditional LCA and economic input-output methodology are combined (Leontief, 1986). In LCA of food and diets process-based LCA is most commonly used (Röös, 2013). Economic I-O LCAs estimate the materials and energy resources required and the environmental impact from economic activities. Thus, economic I-O LCAs can provide an overview of intersectoral relationships within complex systems and enable quantitative assessments of the economic, ecological and environmental effects caused by changes in society (Hendrickson et al., 2010).

An accounting LCA aims to quantify the environmental impact of a specific product or process from a known system, whereas a consequential LCA evaluates the consequences of decisions on changes in a system (Baumann, 2004). In general, studies exploring the environmental impact of diet are based on accounting LCAs. In reality,

many LCAs use a combination of accounting and consequential LCA methodology. For example, when performing an accounting LCA it is common to perform sensitivity assessments using marginal data, in contrast to the traditional approach of using average data for the system studied.

Parameters of particular importance in the LCA methodology are the functional unit, system boundaries and the allocation procedure. The functional unit is the reference base to which input and output flows can be related. It describes the function of the object studied and enables comparison between different systems (ISO, 2006b). In LCAs of food it is common that the environmental impact is expressed in relation to a functional unit based on the quantity or volume consumed or produced (Schau & Fet, 2008), for example per kilogram, liter, serving portion or meal. However, the functional unit can also be based on the economic value of the food (e.g. profit or price) or demand for resources (e.g. land area). In order to account for the quality of food it has become increasingly common to use functional units that relate to the nutritional content, for example, to the energy and protein content or the recommended daily intake of nutrients (Heller et al., 2013; Schau & Fet, 2008).

The system boundaries specify which processes are included and excluded in the assessment. Boundaries can also be set against the life cycles of other products, to define the natural system as well as the geographical and temporal coverage of the study (ISO, 2006b). Ideally, LCAs should include all phases of the life cycle of the product (cradle-to-grave). In the case of food this means that all activities from the primary production of raw materials to the waste handling are accounted for. In practice, it is common to exclude activities deemed to have a negligible impact on the results. Thus, in LCAs of food it is common to include only activities up to the farm gate (cradle-to-gate) since, in general, the agricultural production is responsible for the largest share of the total environmental impact of food products (Schau and Fet, 2008; Sonesson et al., 2010). It is also common to set the system boundaries at the stage of retail (cradle-to-retail) and thereby exclude the consumption phase (e.g. post retail transportation, refrigeration and cooking) and waste management (Heller et al., 2013). Reasons for excluding post-retail stages in LCAs of food are, for example, lack of data, the assumption that post-retail impacts for different foods and diets are similar (Tukker et al., 2011) and the challenge to generalize how food is handled after the stage of retail which is highly dependent on personal behavior and preferences (Heller et al., 2013).

Until recently, LCAs included only direct GHG emissions from the life cycle of food. However, over the past decade it has been found that land use change is a major source of GHG emissions from agriculture. Expanding agricultural land is estimated to be the driver responsible for 80% of global deforestation (Kissinger et al., 2012). In addition to GHG emissions from direct effects of land use change, a discussion about how to account for emissions coming from indirect effects of changes in land use has emerged (Havlík et al, 2011). Emissions from indirect land use change have so far mainly been

debated in association with the production of biofuels, but are relevant in the production of all agricultural products, including food (Cederberg et al., 2011; Röös et al., 2015a).

Losses and waste occurring between production and consumption may also be of importance in the choice of system boundaries. On a global scale, about one third of all food produced is estimated to end up lost or wasted (Gustavsson et al., 2011). Due to losses along the production and distribution chain, there may be a difference of a factor of two or more between the amount (based on weight) of food available for consumption and the amount actually eaten (Hallström & Börjesson, 2013). It is therefore important to adjust consumption data if they are to be used to calculate the environmental impact of the diet, and the opposite if production data are used to calculate, for example, the nutrient intake from the diet. To make data sources comparable, ideally, all processes which contribute to weight losses between production and consumption, e.g. food loss and waste at all stages, deductions for inedible parts of the food (e.g. bones, peels etc.) and weight losses in cooking, should be accounted for. If self-reported consumption data are used it may also be relevant to consider the effect of underreporting (Hallström, 2013).

Allocation is applied if a system generates more than one product. For example, in the dairy sector the environmental impact must be allocated between the production of milk and meat. If possible, allocation should be avoided by system expansions, meaning that by-products are assumed to substitute equivalent products whose environmental impact is subtracted from the overall environmental impact of the system studied. If system expansion is not applicable, allocation is normally based on physical properties, e.g. energy content, weight and volume, or economic value (ISO, 2006b). When comparing different products it is important that the choice of functional unit, system boundaries and allocation procedure are comparable.

5.4.2 Research approach

In this thesis, LCA is used in **papers II, III** and **IV**. In **paper II**, LCA is used to estimate the GHG emissions and land use demand associated with current and healthier, alternative, Swedish meat consumption. In **paper III**, LCA is used to estimate the changes in GHG emissions, from the food sector and health care sector, by adopting healthier diet in the US. Finally, in **paper IV**, the articles reviewed used LCA to assess the effect of dietary change on GHG emissions land use demand of the diet.

Both process-based LCA and IO-LCA is used in this thesis. In **paper II**, the GHG emissions and land use demand of Swedish meat consumption are quantified by using a process-based LCA. Process-based LCA is also the method used to estimate changes in GHG emissions of the food sector in **paper III**, and the method used in most of the articles reviewed in **paper IV**.

In **paper III**, data from the Carnegie-Mellon I-O LCA database (GDI, 2014) are used to estimate current GHG emissions embodied in the US health care sector, and the potential to reduce the emissions via changes in diet. An overview of the method used is provided in Figure 19, in which step three and four describe the method for quantifying changes in GHG emissions within the health care sector. In brief, the approach used for these calculations is combining health care expenditure data (Heidenreich et al., 2011, Mariotto et al., 2011), with spending category percentages assigned to the three diseases studied (AHRQ, 2014, ADA, 2013) and adjusted for inflation to 2013$ (BLS, 2014), with I-O LCA data on GHG emissions embodied in subcategories of medical expenditures (GDI, 2014). The methodology used is more thoroughly described in **paper III**.

Papers II, **III** and most of the articles reviewed in **paper IV** are based on accounting LCAs. However, the two traditional LCA approaches, accounting and consequential, are also to some extent combined, when this was presumed to promote the quality of the assessment.

In **paper II**, the system boundaries are set at the farm gate and hence do not include emissions associated with pre- and post-slaughter transports and slaughtering, packaging, storage and preparation. The estimated GHG emissions from Swedish meat consumption also do not include emissions due to carbon sequestration in pastures or from land use change. The production of bone-free meat required to meet the amounts consumed in each scenario is calculated by assuming 5% of waste between farm gate and household. The reference year of **paper II** is 2009.

In **paper III**, the system boundaries for GHG emissions in the health care sector are the components of the health care sector associated with the diet-related diseases studied, and up to the stage of retail, i.e. excluding emissions from retail to consumer transport, storage and preparation at the consumer stages, food waste disposal and LUC, for GHG emissions in the food sector. To include emissions from food wasted at the consumer and post-consumer stages, food supply data at the farm gate level are adjusted using estimates of food losses through the consumer stage. The reference year of **paper III** is 2013. More information on the LCA methodology used in **papers II** and III is provided in the original publications, appendices and complementary materials.

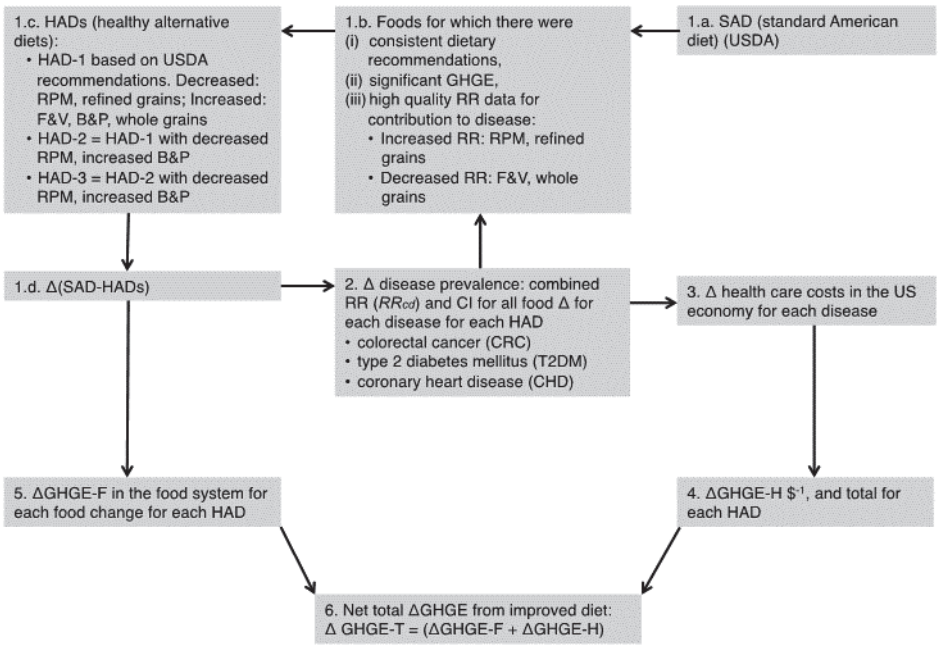

Figure 19. Flow diagram of research design used in paper III. RPM = red and processed meat, F&V = fruits and vegetables, B&P = beans and peas.

5.5 Nutrient calculation

5.5.1 General description

Nutrient calculation is the method that makes it possible to estimate the nutrient content of diets and relate it to dietary requirements and recommendations (Figure 20). The tool is used, for example, by nutritionists to develop dietary recommendations, by dietitians to counsel patients on dietary changes, by epidemiologists to correlate nutrients, foods and diets with causes and prevention of diseases, and by food service managers to plan menus for the public sector (Schakel et al., 1997).

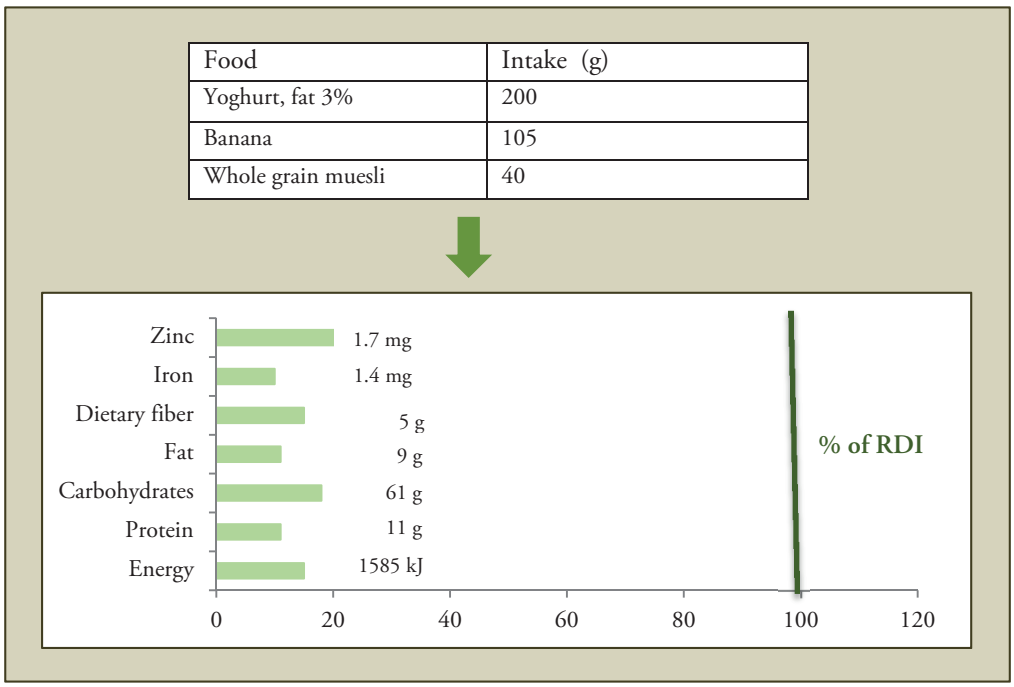

Figure 20. Schematic illustration of the procedure of nutrient calculation; nutrient intake from food intake is evaluated against the level of recommended daily nutrient intake (RDI).

In sustainability assessments of diets, nutrient calculation can be used to design the dietary scenarios to be studied, to choose an appropriate functional unit or to evaluate the effect that changes in the diet will have on the nutritional status and health. For example, to increase the comparability between dietary scenarios, it is common that the energy and protein content is standardized for all dietary scenarios analyzed or that all dietary scenarios are designed to meet specific nutritional recommendations. To

account for the nutritional function, the functional unit in LCAs of food and diets can be related to the content of a specific nutrient or to an index which reflects the content of several nutrients. Nutrient calculation can also be used to evaluate the effects of dietary change on nutritional status and health by comparing the estimated nutrient intake from different dietary scenarios with nutritional recommendations and requirements.

To calculate the nutrient intake of an individual or a population, information is required about the quantity and type of food consumed, as well as the nutritional content of the food consumed. In the past, nutrient calculation was a difficult and time-consuming method, performed by looking up the nutrient content of each food in a book or list, and multiplying it by the quantity of the food consumed. Today, nutrient calculation is, most commonly, performed by using computerized food consumption databases and software (Willet, 2013), such as the Swedish Food Agency´s food database (SFA, 2013).

The nutrient content of food in nutrient databases is generally based on either chemical analysis, values "borrowed" from other nutrient databases or estimated values. The nutrient content of composite foods and dishes, i.e. that contain several components, are generally calculated by summing the nutrients of all constituent ingredients (Schakel et al., 1997). Chemical analyses are usually made on both national and international foods that are imported to the targeted region. Since chemical analyses of food may be relatively money- and time-consuming, such analyses are generally limited only to "relevant foods" and "standard nutrients", i.e. foods consumed in low amounts within the targeted population, and nutrients relevant for health promotion and disease prevention within the targeted population (SFA, 2013).

For foods that are rarely consumed and for "non-standard" nutrients, methods that are more resource efficient are generally applied. For example, if chemical analyses of the nutrient content of a food have been made by another lab, those values can be borrowed or bought, and thereafter added to the nutrition database or software. Another cost- and time-efficient method is to estimate the nutrient content of the food by using "qualified guesses". "Qualified guesses" can, for example, be based on the nutrient content of other components in the same food (e.g. chicken wings and chicken breast), on the nutrient content of similar foods (e.g. yellow and red onion), on defined algorithms (e.g. energy from protein, fat and carbohydrates = total energy content). In some cases the nutrient content in a certain food is estimated to be zero, for example, for the case of cholesterol and vitamin B_{12} in plant-based foods or dietary fiber in animal products (Schakel et al., 1997).

Nutrient databases typically contain one value indicating the nutrient content for each food. However, precise estimations of the nutritional content of food and diets are complex and rare, since there are many parameters that will affect the nutritional content of food. The nutritional content of food, for example, varies due to differences

in regional conditions, botanical varieties, management and processing technique. The nutrient content of food may also vary depending on the type and procedure of cooking. Due to this variability, nutrient values in databases and software´s should be interpreted as mean values rather than precise values (SFA, 2013).

Another complexity is the difference between intake and uptake (i.e. absorption) of nutrients from the diet. Absorption of nutrients depends on various factors, such as the current nutritional and health status of the individual (e.g. iron deficiency enhances the physiological absorption of iron in the gut) and biochemical properties of the food (e.g. nutrition inhibitors and promotors such as phytic acid, lectines and vitamin C). In addition, the combination of foods affects the uptake of some nutrients (e.g. the protein quality, i.e. amino acid complementarity, affects the physiological digestibility and absorption of protein) (NCM, 2014). In nutrient calculation, the difference between ingested and absorbed amounts of nutrients is handled by evaluating levels of nutrients ingested (e.g. intake) against recommended intake levels that are adjusted for "losses" due to mal-absorption. Thus, recommended intake levels of nutrients and foods account for the differences between intake and uptake and thereby reflect the intake levels of micro- and macronutrients estimated to be required to meet physiological needs (NCM, 2014).

5.5.2 Research approach

Nutrient calculation is used in **papers II** and **III** of this thesis. In **paper II** the contribution (i.e. estimated intake) of nutrients from Swedish meat consumption is estimated for protein, total fat, saturated fat, iron and zinc. In **paper III**, a more limited nutritional calculation is performed to validate the energy content of the studied scenarios.

In **paper II**, meat consumption in the scenarios studied is allocated to either consumption of beef, pork, chicken, mixed charcuteries or unmixed charcuteries. Thereafter, trade data on the average sale of meat in Sweden (data for 1993) (SFA/SBA, 2011) are used to divide the consumption among 37 different meat products, with varying nutrient content. The nutrient contribution is calculated by multiplying the amount of each meat product consumed (uncooked weight) by its respective nutrient content. The nutrient data are taken from the Swedish Food Agency´s food database. As an example, based on sales trade data, chicken breast fillet without skin accounts for 46% of the Swedish per capita consumption of chicken (SFA, 2011), hence, this is also the amount assumed to be consumed in the REF scenario. Table 11 illustrates the procedure used for nutrient calculation in **paper II**.

Table 11. Illustration of method of nutrient calculation in paper II

Type of product	Percentage of total chicken consumption [a]	Energy (kJ)	Total fat (g)	Saturated fat (g)	Protein (g)	Iron (mg)	Zinc (mg)
Chicken breast fillet without skin	46%	437	1.2	0.4	23.1	0.7	0.8
Source	Swedish sales trade data.	[b]Nutrient content per 100g (uncooked weight) based on data from the Swedish Food Agency´s food data bases.					

In **paper III**, the energy content of the studied scenarios is estimated to validate their nutritional comparability. The method used for nutrient calculation is similar to that in **paper II**, with the difference that calculations are based on less specific data. Due to lack of good quality data of specific foods consumed, the energy content in the dietary scenarios is calculated based on the energy content of certain selected foods or on averages of the different food categories.

5.6 Nutrition epidemiology

5.6.1 General description

Epidemiology is the part of medicine that studies the occurrence of diseases within a population in relation to personal characteristics and different types of daily life exposure. As a sub-discipline of epidemiology, nutrition epidemiology addresses the role of food and nutrition in the risk of health outcomes in humans (Willet, 2013).

The observation that absence of specific components in the diet was the cause of diseases such as scurvy, beriberi, pellagra and rickets in the 18th and 19th century is often described as the starting point of nutrition epidemiology. Since then the central focus of nutrition epidemiology has shifted from previously primarily being the study of the effect of nutrient deficiency to mapping relationships between diet and chronic diseases, such as cardiovascular disease, diabetes and cancer (Willet, 2013). Today, nutritional epidemiology constitutes an important source of information in preventive medicine that is often the basis on which dietary recommendations and policies are constructed (Coulston & Boshey, 2008).

Several methods are used to study diet-disease relationships in humans. Epidemiological studies are sometimes classified as observational or experimental (Table 12). Observational studies assess the prevalence of disease in a population, without the researcher influencing the natural situation and circumstances. In experimental the aim is instead often to actively change a behavior or exposure to evaluate the effect under controlled conditions (Bonita, 2010).

Table 12. Examples of epidemiological methods

TYPE OF STUDY	ALTERNATIVE NAME	STUDY PARTICIPANTS
Observational studies		
Ecologic studies	Correlational studies	Populations
Case-control studies	Case-reference studies	Individuals
Cohort studies	Follow-up studies	Individuals
Experimental studies	**Intervention studies**	
Randomized controlled trials	Clinical trials	Individuals

Based on Bonita et al. (2010)

Ecologic studies compare average disease rates and per capita food intake within populations, and are important for generating hypotheses about dietary factors of importance in disease development. Case-control studies analyze differences in past dietary habits among individuals diagnosed with a specific disease (case) and individuals

free from the disease (control). In (prospective) cohort studies, large samples of individuals free from disease are followed over long time periods in order to monitor eating habits and occurrence of disease. In the category of experimental studies, randomized control trials compare health effects on individuals randomly assigned to a specific dietary exposure with individuals receiving no dietary treatment (Bonita, 2010; Coulston & Boushey, 2008; Willet, 2013).

Large differences exist between different epidemiological methodologies. For example, the number of study participants can vary from as few as ten individuals in randomized control trials to several hundred thousand individuals in large cohort studies. Similarly, the time perspective can vary from interventions lasting over a few weeks to observations made over several decades (Willet, 2013). Due to the differences in study designs, some epidemiological methods are more suitable for some assessments (Bonita, 2010). In general, prospective cohort studies and randomized controlled trials are perceived as the most rigorous methods available for determining diet-disease relationships (Coulston & Boushey, 2008).

Nutrition epidemiology analyses the question of whether and how dietary factors influence the risk of disease. Occurrence of disease can either be expressed as prevalence or incidence of disease. Disease prevalence describes the proportion of a population with a disease at a given time. Disease incidence refers to the number of new cases of a disease that occurs, or how many individuals who become ill during a certain time period. In addition, disease occurrence can be measured by the cumulative incidence, which describes the proportion of a healthy population that becomes ill during a certain time period. The different existing metrics used to describe disease occurrence relate to each other. For example, the prevalence of a disease depends on both the incidence and duration of the disease. Occurrence of disease can further be measured by the use of absolute or relative comparisons. The relative risk of disease is the metric most commonly used (Ahlbom, 2006).

The relative risk (RR) of disease indicates the probability of developing a disease in an exposed group of people compared with those not exposed. RR >1.0 indicates an increased risk of developing the disease among exposed individuals, whereas RR < 1.0 indicates that the exposure has a protective effect against the disease. If no diet-disease relationship is detected, i.e. if the risk of disease is the same in both groups, the RR=1.0. The precision of the RR estimate is given by the confidence interval (CI). A 95% CI of 1.2-1.7 means that there is a 95% probability that the true RR lies between 1.2 and 1.7 (i.e. a 20-70% increase in RR). The width of the confidence interval is determined by the size of the sample and the variability of the measure. A large sample with low inherent variability will provide a narrow confidence interval, indicating a high precision of the RR estimate. A statistically significant result has a 95% confidence interval that does not include the RR value of one (Coulston & Boushey, 2008).

5.6.2 Research approach

Nutritional epidemiology is used in **paper III** of this thesis to estimate the change in relative risk of coronary heart disease, diabetes type II, and colorectal cancer, with changes in the US average diet.

The health effects of changing the diet in the US are estimated based on relative risk estimates for the three diseases studied. Relative risk estimates of the association between different foods and the three diseases are based on results of meta-analyses in the literature (Table 13). The methodological approach used to review the literature is further described in section 5.2.

Table 13. Relative risk estimates between dietary intake and disease.

Disease	Dietary exposure	RR	95% CI	Measure for RR	Source
Coronary Heart Disease					
	Processed meat	1.42	1.07-1.89	50 g/d increment	Micha et al., 2010
	Fruit and vegetables	0.96	0.93-0.99	106 g/d increment	Dauchet et al., 2006
	Whole grains	0.81	0.75-0.86	High vs. Low	Mente et al., 2009
Type II Diabetes					
	Unprocessed red meat	1.20	1.07-1.38	120 g/d increment	Aune et al., 2009
		1.16	0.92-1.46	100 g/d increment	Micha et al. 2010
		1.19	1.04-1.37	85 g/d increment	Pan et al., 2011
		1.15	0.99-1.33	100 g/d increment	Feskens et al., 2013
	Processed meat	1.57	1.28-1.93	50 g/d increment	Aune et al., 2009
		1.19	1.11-1.27	50 g/d increment	Micha et al., 2010
		1.51	1.25-1.82	50 g/d increment	Pan et al., 2011
		1.32	1.19-1.48	50 g/d increment	Feskens et al., 2013
	Whole grains	0.79	0.72-0.87	60 g/d increment	de Munter et al., 2007
		0.69	0.60-0.80	90 g/d increment	Aune et al., 2013

Colo-rectal Cancer					
	Unprocessed red meat	1.28	1.18-1.39	120 g/d increment	Larsson & Wolk, 2006
		1.29	1.04-1.60	100 g/d increment	WCRF/AICR, 2007
		1.17	1.02-1.33	100 g/d increment	Chan et al., 2011
	Processed meat	1.09	1.05-1.13	30 g/d increment	Larsson & Wolk, 2006
		1.21	1.04-1.42	50 g/d increment	WCRF/AICR, 2007
		1.18	1.10-1.28	50 g/d increment	Chan et al., 2011
	Whole grains	0.86	0.79-0.94	90 g/d increment	Aune et al., 2011

To estimate the specific health effect of changing the diet from the reference scenario (SAD) to the two alternative dietary scenarios studied (HAD-1, HAD-2), the RR estimates obtained from the meta-analyses required further processing. An overview of the relative risk estimates used in **paper III** is provided in Table 14.

Table 14. Overview of relative risk estimates used in paper III.

RELATIVE RISK ESTIMATE	Abbreviation	Description	Equation
Relative risk	RR	RR estimate of specific food-disease relationships, obtained from meta-analyses.	
Revised relative risk	RRre	RR estimate of specific changes in food intake (e.g. effect of changing intake levels of red meat OR whole grains), based on intake levels of the scenarios studied.	I
Combined relative risk	RRcb	RR estimate of the combined effect of all changes in food intake (e.g. effect of changing intake levels of red meat AND whole grains), based on intake levels of the scenarios studied.	II

Firstly, the RR estimates were recalculated to reflect the specific changes of food intake based on the dietary scenarios developed. Equation I describes the approach used to calculate the revised RR (RRre) for each food and disease.

$$RRre = RR^{((x-y)/u)} \qquad \textbf{Eq. I}$$

where *RR* is the original RR obtained from meta-analyses for food *f* (e.g., processed meat) and disease *d* (e.g., CHD), *x* is the level of *f* in the alternative dietary scenario, *y* is the level of *f* in the reference dietary scenario, and *u* is the unit increase reported in the meta-analysis identified for disease *d*. The reductions in RR for a unit change in food consumption are assumed to follow a log-linear dose-response relationship across the whole range of intake levels in the dietary scenarios.

Thereafter, a combined relative risk (RR_{cd}) estimate is calculated for each disease, representing the total effect of all dietary changes contributing to the RR. Equation II, shows the method used to calculate the combined relative risk of disease (Ezzati et al. 2006). Health effects of different foods are assumed to be independent of each other.

$$RR_{cd} = RR_{re1} * RR_{re2} * RR_{re3} * \ldots RR_{ref} \qquad \textbf{Eq. II}$$

where RR_{re1}, RR_{re2}, RR_{re3}, and RR_{ref} are the revised RR values for each of the individual foods changed in the diet.

Figure 19 (step one and two) in section 5.4.2, provides an overview of the methodology used to estimate the change in relative risk of the three diseases studied, with changes in the US average diet.

5.7 Monte Carlo Analysis

5.7.1 General description

As previously described in this chapter, the assessment of the environmental, nutritional and health effects of food consumption and production are subject to uncertainty and variability. Uncertainty assessment provides for more accurate interpretation of the results. Traditional statistical uncertainty quantification methods are difficult to apply when there are many dependent and independent input variables to model. In this thesis, Monte Carlo analysis, a commonly used stochastic simulation method (Rubinstein, 2007), is used to establish how uncertainty and variability in input data affect the uncertainty in the final results.

The approach of Monte Carlo analysis varies, but in general includes the following steps; 1) static model generation (i.e. development of a deterministic model that defines process inputs and outputs); 2) identification of variability factors (i.e. factors contributing to variability) and their underlying probability distribution (i.e. statistical distributions describing the nature of each variability factor); 3) random variable generation (i.e. repeated random sampling from each probability distribution); and 4) statistical analysis (i.e. simulation of total variability based on the results from the random samples) (Raychaudhuri, 2008). Figure 21 provides a schematic illustration of the methodological approach of Monte Carlo analysis.

To perform a Monte Carlo analysis, the distribution of each variability factor must be identified or estimated. Typically, historical data (e.g. minimum, maximum, mean, median, mode) are used to identify the most appropriate probability distribution for each variability factor. The variance in data describes the spread of the distribution, and the square root of the variance provides the standard deviation (Rubinstein, 2007). The procedure of identifying the most suitable probability distribution in a series of data is sometimes called distribution fitting (Ricci, 2006).

The goal of distribution fitting is to predict the probability of, for example, the amount of GHG emissions embodied, or the nutrient content of a specific food item. There are several probability distributions to which the data can be fitted. Which distribution is identified to be most appropriate depends on the characteristics of the historical data. Normally distributed data are symmetrically distributed around the mean value, which is also the median and mode value of the distribution, with diminishing occurrence further away from the mean. If the distribution is skewed, the data may instead follow a log-normal distribution. A triangular distribution, based on a minimum, maximum and "best estimate" is often used if the historical sample of data is limited (Rubinstein, 2007).

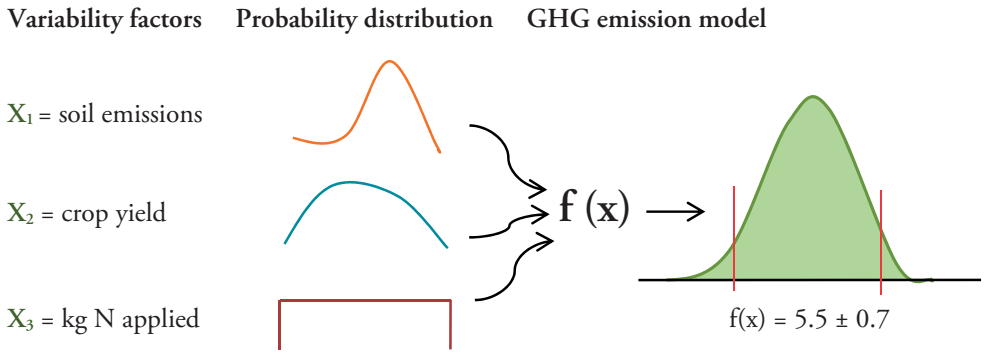

Figure 21. Schematic illustration of Monte Carlo analysis of GHG emissions. GHG emissions from the studied food product are estimated to be 5.5 ± 0.7 kg of CO_2 eq. per kg.

5.7.2 Research approach

In this thesis, Monte Carlo analysis is used to assess the variability of the results in **papers II** and **III**.

In **paper II**, the uncertainty in the nutrient content, GHG emissions and land use are captured by using Monte Carlo analysis. The uncertainty range for GHG emissions and land use for different meat production systems are established based on an uncertainty-importance analysis where realistic, minimum and maximum values are used for the parameters with the greatest influence on the end results. By setting these influential parameters at realistic maximum and minimum values (found in statistics, information from trade associations and scientific literature) an uncertainty range was established. For the sake of simplicity and due to the limited amount of data, a triangular distribution is used to describe this uncertainty range within production system,

In **paper II**, probability distributions for the GHG emissions and land use for beef production in consist of a discrete distribution for 13 different production systems in different geographical regions, and the triangular distributions described above for within production system uncertainty. In the Monte Carlo analysis of GHG emissions and land use demand for meat production, the random value is determined by first drawing a production system and then drawing the GHG emission and land use for that production system from that triangular distribution. For pork and chicken, all production is assumed to be carried out in similar production systems, and therefore only one triangular distribution is used. In the Monte Carlo analysis of contribution of nutrients from meat, the random value is drawn from a discrete distribution containing the consumption percentage of 37 different meat products and their respective nutrient content.

In **paper III**, the uncertainty in GHG emissions from the food- and health care system, and the change in relative risk of disease due to diet change are estimated by using Monte Carlo analysis. Because of the limited number of GHG estimates in the literature, a triangular distribution model for each food category studied is used for the simulation of GHG emissions from the food system, from which a random value is drawn. The probability distribution is developed based on the maximum, minimum and median values of GHG data for 25 different food categories, with varying production methods and geographical origin, found in the literature. For food categories for which only one set of GHG data was found (frozen and dried fruits), the probability distribution is based on an assumed uncertainty of ± 20% of the value found. In the Monte Carlo analysis of RR of diseases, a random value is drawn from a lognormal distribution, based on 20 different RR estimates, revised to reflect the type and amount of food in the scenarios studied. The uncertainty intervals for the GHG emissions of the health care sector are based on the estimated uncertainty in RR of disease.

6 Results

This chapter describes the main results of the thesis. The results are reported for each paper separately, and thus include the results from **paper I** (6.1), **paper II** (6.2), **paper III** (6.3), and **paper IV** (6.4). The results reported are linked to the specific objectives of each article, previously outlined in section 2.2.

6.1 Paper I

6.1.1 Methods for producing meat consumption data

Three main methodological approaches exist to produce food and meat consumption statistics. Consumption data can either be derived from statistics of agricultural supply, household budget surveys, or individual dietary surveys. Being aware of the method used to produce consumption statistics is important because the definition of consumption varies, which has consequences for how the data should be interpreted and used in an appropriate manner.

Meat consumption statistics based on the agricultural supply available describes the average quantity of meat available for human consumption within a country or region. This type of data is useful to study consumption trends over time and for comparing consumption in different countries. As the data refer to the average consumption for the whole population and as household waste is not accounted for it is not suitable for studying consumption characteristics in different socioeconomic groups or individuals, or to describe what people actually eat. Factors that are important to consider in the use of meat consumption data based on agricultural available supply include how the consumption of non-commercial meat, meat in processed and prepared meals as well as food losses and waste are accounted for.

Meat consumption statistics based on household budget surveys provide information on the amount of money spent on meat per household and sometimes also on the quantity of meat purchased per household. The data can be used to study and compare consumption in different regions and socioeconomic groups. As the data do not describe what happens to the meat after purchase they are more suitable for studying meat consumption in populations than in individuals. The procedure for categorizing different types of meat, accounting for food waste and for food consumed outside the household are factors that may affect the reported amount of meat purchased.

Individual dietary surveys provide data that refer to the actual amount of meat eaten by individuals and groups, and are therefore the most accurate method for obtaining data on food consumption. These data offer the possibility to study dietary habits and their consequences at an individual level, and to match dietary habits to different characteristics within the population. When data based on individual dietary surveys are presented and used, it should be made clear whether they refer to raw or cooked meat, and whether household waste is accounted for.

6.1.2 Uncertainty factors in meat consumption data

Depending on the methodology used to produce meat consumption statistics the data may refer to the available supply, the purchased or the eaten amount of meat. In order to facilitate a correct interpretation and use of meat consumption data, four main uncertainty factors to be considered are identified; bone weight, food losses and waste, raw or cooked meat, mixed meat and prepared meals (Figure 22).

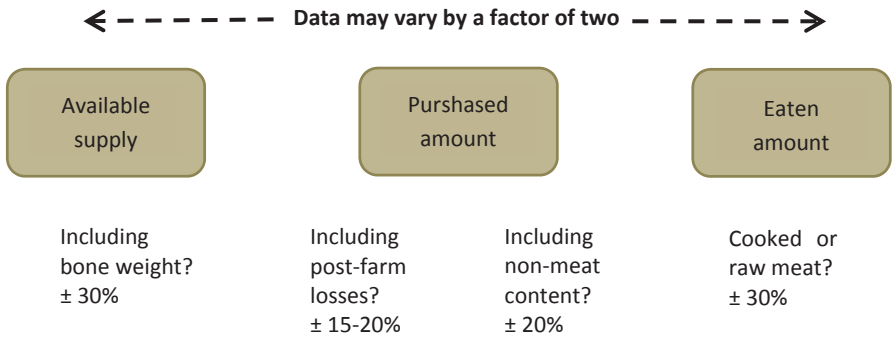

Figure 22. Illustration of identified factors causing uncertainty and discrepancy in meat consumption data.

Depending on whether meat consumption statistics are presented for meat with or without bones the data may vary by about ± 25-40%. Consumption statistics based on agricultural supply often, but not always, refer to meat including bone weight, whereas data based on household budget surveys and individual dietary food surveys usually refer to the amount of meat purchased at retail and/or the amount actually eaten, indicative to bone-free meat.

Whether losses and wastage in the different stages of the supply chain are accounted for or not may affect meat consumption data by ± 15-20%. Meat consumption data based on agricultural, available supply do not account for household wastage, whereas post-farm losses up to retail may be included. Meat consumption data based on household budget surveys and individual dietary surveys rarely or only partially account for food waste in the household.

Meat consumption statistics can be presented either as raw or cooked weight. Depending on how the data are presented the reported amount of meat consumed may vary by ± 20-50%. Meat consumption data based on agricultural, available supply and household budget surveys in general refer to raw weight, whereas data based on

individual dietary surveys as well as nutritional recommendations, may be reported either as raw or cooked weight, depending on the method used.

Meat consumption statistics may vary by about ± 40-55 % depending on whether the data refer to the total weight or only the meat content in mixed meat products and prepared meals. Meat consumption data based on agricultural data often but not always refer to only the meat content of such products whereas data based on household budget surveys and individual dietary surveys in general refer to the total weight including non-meat components.

6.1.3 Implications of uncertainty and discrepancy

Due to the use of different methodologies and definitions to produce and present consumption data, the meaning of one kg of meat can differ substantially. The problem is reflected by the divergent information in circulation in the literature and in the media on how much meat is eaten.

The findings of **paper I** demonstrate that per capita meat consumption levels can vary by a factor of two or more due to inconsistencies in the way statistics are produced and presented. In subsequent calculations of environmental and health effects of meat consumption (e.g. in LCA, nutrient calculation) there is an obvious risk that consumption data are misinterpreted and used for the wrong purpose. An incomplete understanding of meat consumption data can thus have widespread implications for research findings and recommendations based on these.

This paper emphasizes the importance of being aware of what the data represent when meat consumption data are interpreted and used for further calculations. Meat consumption statistics based on the agricultural available supply of raw meat, including or excluding bones, are often the basis for environmental assessments of dietary patterns, whereas data on the actual intake of meat, expressed as uncooked or cooked meat, are generally employed to study the nutritional and health effects of diets. If consumption data are to be combined with data on the environmental impact or nutritional content expressed per kg of meat it is critical that the functional units correspond to each other. The uncertainty factors described in section 6.1.2 can be used as a check list to evaluate the equivalence between the data.

6.1.4 Suggestions for improvements

The results of **Paper I** show that methods and assumptions used to produce food consumption statistics vary between different methods. In addition, discrepancies in how per capita food consumption statistics are produced exist between different data sources, such as national agricultural data, data from EUROSTAT or FAOSTAT.

There is a current lack of accessible and transparent information and descriptions of underlying assumptions and procedures used to generate meat consumption data. In particular, information on assumptions regarding bone weight, food losses and meat content in mixed and prepared meals is difficult to find. Misuse of data is thus an obvious risk when consumption statistics are used for subsequent calculations of environmental and health effects.

A prerequisite to avoid misinterpretation of meat consumption statistics is that accessible and transparent information about the data is provided. Currently, descriptions of the methodology and assumptions used for producing consumption statistics are often inadequate or difficult to find and interpret. We believe that a more straightforward, complete and transparent documentation of consumption statistics would increase their usefulness and facilitate a proper use of the data.

6.2 Paper II

6.2.1 How healthy is current Swedish meat consumption?

Swedish consumption of bone-less, uncooked meat in 2009 (REF) is estimated to be 170 g capita^{-1} day^{-1}, or 62 kg capita^{-1} year^{-1}, of which almost one quarter is estimated to consist of charcuteries. Charcuteries consumed are estimated to consist to 62% of meat, of which most is pork (83%) and a smaller proportion is beef (17%). If the total weight of consumed charcuteries, including the non-meat content, is accounted for, consumption of total meat increases to 190 g capita^{-1} day^{-1}, or 70 kg capita^{-1} year^{-1}. To supply the current amounts of meat consumed in Sweden (REF), requires a production of 65 kg capita^{-1} year^{-1}. Table 15 provides an overview of the estimated amount of meat consumed and produced in the reference scenario and in the two hypothetical, alternative scenarios of healthier Swedish meat consumption studied in **paper II**.

Table 15. Meat consumption and production in the scenarios studied

MEAT CONSUMPTION (g/capita and day)							
SCENARIO	BEEF	PORK	CHICKEN	MIXED CHARCUTERIES	UNMIXED CHARCUTERIES	TOTAL kg/yr	TOTAL g/d
REF	44	47	40	48	14	70	193
NUTR-1[a]	30	30	66	0	0	46	126
NUTR-2 [b]	13	0	113	0	0	46	126

MEAT PRODUCTION (kg/capita and year, g/capita and day)								
SCENARIO	BEEF kg/yr	BEEF g/d	PORK kg/yr	PORK g/d	CHICKEN kg/yr	CHICKEN g/d	TOTAL kg/yr	TOTAL g/d
REF	19	53	30	83	15	42	65	178
NUTR-1 [c]	12	32	12	32	25	69	48	133
NUTR-2 [c]	5	14	0	0	43	119	48	133

[a] Refers to the total weight of pure uncooked meat, excluding non-meat ingredients in charcuteries. Mixed and unmixed charcuteries account for 15% and 8% of the pure meat intake, respectively. [b] Refers to weight of uncooked meat of the final meat products, including non-meat ingredients in charcuteries. [c] Refers to the carcass weight of meat required to be produced, nationally and internationally, to meet NUTR-1 and NUTR-2. A waste percentage of 5% is assumed between farm gate (production) and household (consumption).

To meet Swedish and international guidelines for a healthy meat intake (NUTR-1, NUTR-2), current consumption of pure, uncooked meat should be reduced by approximately 25%, to 125 g capita^{-1} day^{-1}, or 46 kg capita^{-1} year^{-1}, and consumption of charcuteries by as much as possible. To supply the consumption of meat in NUTR-1 and NUTR-2 requires a production of 48 kg capita^{-1} year^{-1}.

6.2.2 Changes in meat consumption – Effect on nutrition?

Current Swedish meat consumption (REF) contributes about 14% of the recommended daily energy requirement, about one third of the maximum RDI of total fat and saturated fat and between one and two thirds of the RDI of protein, iron and zinc. In NUTR-1 and NUTR-2 the contribution of energy, protein and iron is, on average, equivalent to 7%, 40% and 14% of the RDI, respectively. The corresponding contribution of total fat, saturated fat and zinc is between 8-11%, 8-14% and 29-38% of RDI, respectively. Table 16 and Figure 23 provide an overview of the contribution of nutrients from Swedish meat consumption in the scenarios studied.

The reduction in meat consumption required to meet guidelines for a healthy meat intake (NUTR-1, NUTR-2) reduces the contribution of total and saturated fat by about two thirds, of energy, iron and zinc by about half and of protein by about a quarter, in comparison with current meat consumption (REF). For most nutrients uncertainty intervals are in the range of ± 50% but for some they are even larger. The results indicate that a 25% reduction in current Swedish meat consumption would have a minor effect on nutritional status concerning energy and protein intake, whereas the intake of total fat, saturated fat, iron and zinc is reduced more strongly.

Table 16. Daily per capita contribution of nutrients from meat consumption in the scenarios studied (uncertainty intervals)

	REF	NUTR-1	NUTR-2
Energy (MJ/d)	1.5 (1.1 – 2.0)	0.8 (0.6 – 1.1)	0.7 (0.6 – 0.9)
Total fat (g/d)	24 (13 – 40)	9.7 (2.6 – 21)	6.6 (1.7 – 14)
Saturated fat (g/d)	9.6 (5.0 – 16)	3.9 (1.0 – 8.2)	2.3 (0.6 – 4.9)
Protein (g/d)	34 (29 – 37)	25 (22 – 28)	26 (23 – 29)
Iron (mg/d)	3.8 (2.2 – 10)	1.8 (1.3 – 2.2)	1.4 (1.1 – 1.9)
Zinc (mg/d)	5.2 (4.1 – 6.4)	3.0 (2.3 – 3.8)	2.3 (1.4 – 2.9)

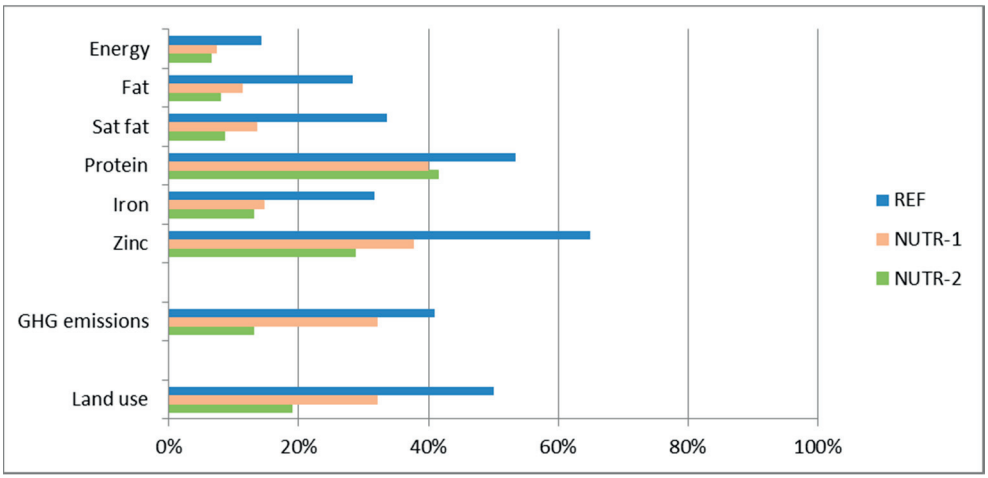

Figure 23. Overview of results from paper II. Effect of changes in Swedish meat consumption on nutritional contribution (g/mg capita-^{1} day^{-1}), GHG emission (kg CO2eq capita^{-1} year^{-1}), and land use demand (ha capita^{-1} year^{-1}), expressed in relation to recommended intake levels for a healthy diet.

6.2.3 Changes in meat consumption – Effect on climate?

The production of meat currently consumed in Sweden (REF) emits about 0.6 tons of CO_2 eq. capita^{-1} year^{-1}, representing approximately 40% of the total budget for sustainable GHG emissions. A dietary change towards healthier meat consumption would reduce GHG emissions associated with Swedish meat consumption to approximately 0.4 and 0.2 tons of CO_2 eq. capita^{-1} year^{-1}, in NUTR-1 and NUTR-2, respectively. Meat consumption would in these scenarios account for some 15-25% of the total GHG emission budget. Uncertainty intervals for GHG emissions in the scenarios studied range from approximately -15% to + 85%. Figure 23 provides an overview of the GHG emissions produced to supply Swedish meat consumption, in the scenarios studied.

A dietary change towards healthier meat consumption (NUTR-1, NUTR-2) would, according to the results, reduce per capita GHG emissions from meat by about half, compared to current meat consumption (REF). However, despite the lower climate impact in NUTR-1 and NUTR-2, meat consumption in these scenarios accounts for some 10-25% of the required emission target, which also needs to cover emissions from other foods and other activities such as housing, transportation and other consumption.

6.2.4 Changes in meat consumption – Effect on land use demand?

The production of meat currently consumed in Sweden (REF) demands about 0.11 ha capita^{-1} year^{-1}. A dietary change towards healthier meat consumption would reduce the demand for agricultural land to 0.07 and 0.04 ha capita^{-1} year^{-1} in NUTR-1 and NUTR-2, respectively. The uncertainty intervals for land requirement in the scenarios studied range approximately from -25 to +110%. Figure 23 provides an overview of the land use demand for meat consumed, in the scenarios studied.

Current Swedish meat consumption (REF) requires an area representing half of the area estimated to be available for sustainable cropping capita^{-1} year^{-1}, in 2050. In NUTR-1 and NUTR-2 the proportion of this area used for meat production is reduced to about 20-35%, which releases land that could be used for production of other types of food or for the production of bioenergy for example.

6.3 Paper III

6.3.1 How healthy is the current US diet?

US average intake of red and processed meat is estimated to be 92 g of cooked meat per day (62%, 38%), of which most is beef and pork (57%, 44%). Average intake levels fruits and vegetables and grains are estimated to be 358 and 167 g per day (cooked weight), respectively. About half of the total consumption is estimated to consist of fresh fruits and vegetables. Consumption of refined grains is estimated to account for 90% of total grain consumption. Figure 24 illustrates the estimated food intake levels in the scenarios studied.

To meet the USDA dietary recommendations (HAD-1), current intake levels of red and processed meat should be reduced by approximately 45%, of which processed meat should be as limited as possible. Intake levels of total fruits and vegetables should be approximately doubled, with the major part of the increase coming from fresh fruits and vegetables. Total grain consumption should be reduced by 22%, at the same time as the proportion of whole grain should be increased from 10% to 60%.

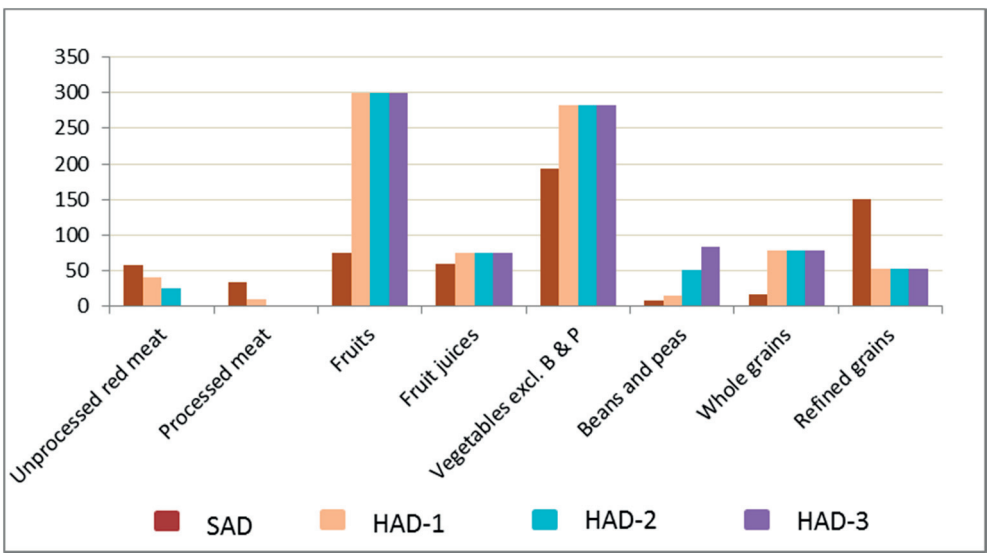

Figure 24. Food intake levels (g of cooked food capita^{-1}day^{-1}) of scenarios studied in paper III.

6.3.2 Diet change- effects on health?

Healthier scenarios of the US diet (HAD-1, HAD-2, HAD-3) reduce the relative risk of coronary heart disease (CHD), diabetes type II (T2D) and colorectal cancer (CRC) by 20-45%. HAD-3, in which all red and processed meat was replaced with legumes, provided the greatest reduction in disease prevalence of the HADs. The potential annual savings in US health care costs, with the reduction in the prevalence of CHD, T2D and CRC, are estimated to between 54 and 72 US$ billion year^{-1}, equivalent to about 20-30% of the total health care costs for these diseases of 220 billion US$ year^{-1}. Table 17 gives an overview of the health effects of adopting healthier diets in the US.

Table 17. Changes in relative risk of disease and associated health care costs.

DIET CHANGE FROM SAD TO	REDUCED RELATIVE RISK OF (95% CI)			REDUCED HEALTH CARE COSTS ($B yr^{-1})
	CHD	T2D	CRC	
HAD-1	40% (29-51)	35% (28-44)	20% (13-26)	54
HAD-2	45% (31-67)	41% (32-50)	25% (17-32)	65
HAD-3	45% (32-58)	43% (34-53)	29% (20-37)	72

In the transition from the SAD to the HAD-1, increased intake of whole grains has the greatest effect on the relative risk reduction for all studied diseases, followed by the reduction in processed meat. Adoption of HAD-2 and HAD-3 further reduce the relative risk of CHD, T2D and CRC by 5%, 6-8% and 5-9%, respectively, mainly due to the reduced intake of processed meat and unprocessed red meat. Table 18 summarizes the estimated changes in relative risk of studied diseases due to the individual effect of changes in the US diet.

Table 18. Changes in relative risk of disease due to individual changes in the US diet.

	RECALCULATED RELATIVE RISK (RRre)[a]								
	SAD-HAD-1			SAD-HAD-2			SAD-HAD-3		
FOOD ITEMS	CHD	T2D	CRC	CHD	T2D	CRC	CHD	T2D	CRC
Red meat	-	0.97	0.97	-	0.95	0.93	-	0.91	0.89
Processed meat	0.85	0.86	0.92	0.79	0.80	0.89	0.79	0.80	0.89
F&V	0.88	-	-	0.87	-	-	0.86	-	-
Whole grains	0.81	0.78	0.90	0.81	0.78	0.90	0.81	0.78	0.90

6.3.3 Diet change – effects on GHG emissions?

Healthier scenarios of the US diet are estimated to reduce GHG emissions in the food system by 70-1500 kg CO_2 eq. capita^{-1} year^{-1}. The results suggest that adoption of healthier diets in the US has the potential to reduce current GHG emissions from the food sector by between 6% and 70%, depending on the scenario and choice of GWP value for methane. Furthermore, adoption of healthier diets in the US is estimated to reduce GHG emissions in the health care system by 65-100 kg CO_2 eq. capita^{-1}year^{-1}. The effect on GHG emissions in the food and health care sector from adoption of healthier diets in the US are illustrated in Figure 25 and Figure 26, respectively.

Adoption of healthier diets in the US is estimated to reduce the combined GHG emissions from the food and health care sector by 130-1620 kg CO_2 eq. capita^{-1}year^{-1}. Emission reductions in the food sector account for 51-93% of the total GHG emissions reduction in all HADs, and thus dominate the overall GHG emission reduction potential. Figure 27 shows the combined reduction of GHGs, within the food and health care sector, from adoption of healthier diets in the US.

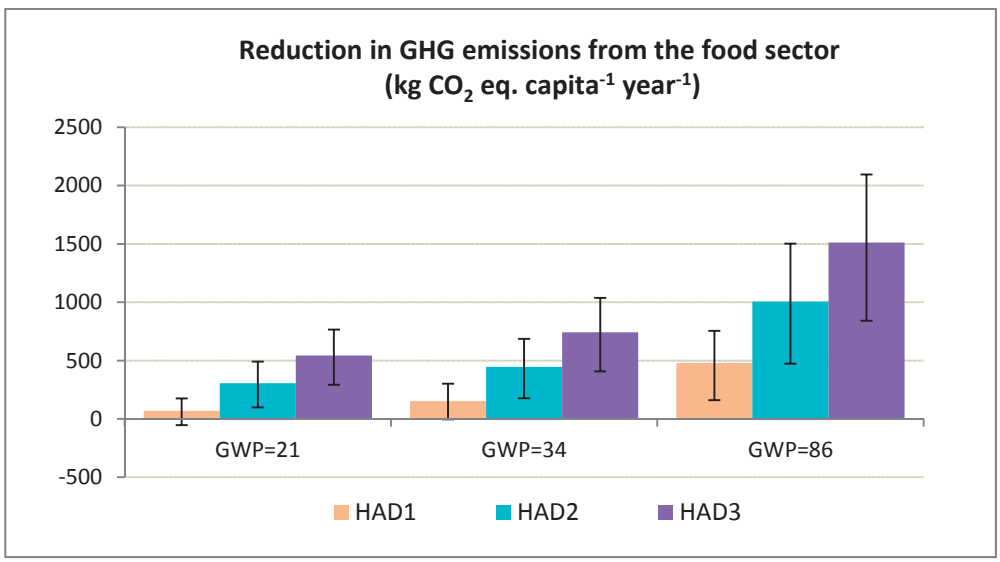

Figure 25. Reduction in GHG emissions from the food sector relative SAD

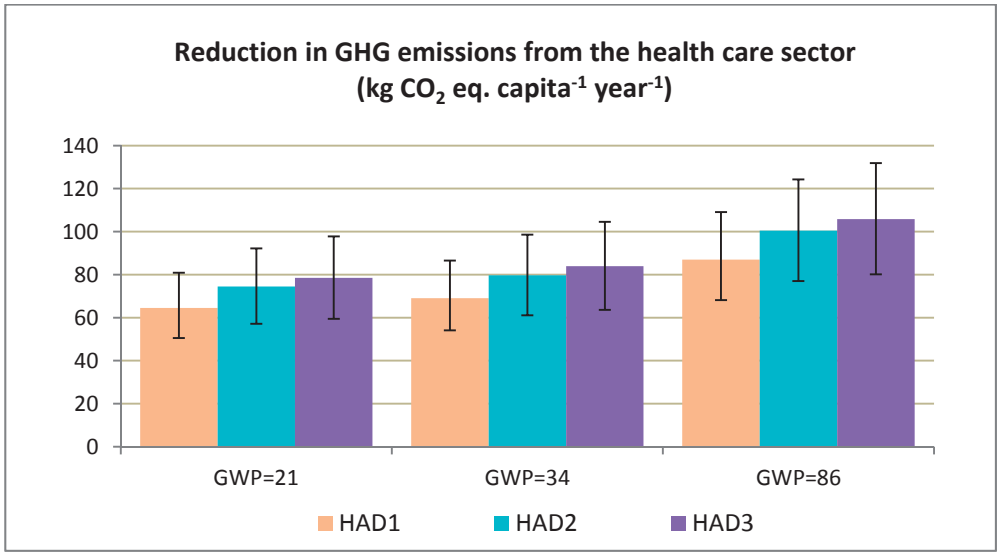

Figure 26. Reduction in GHG emissions in the health-care sector for three diet scenarios relative SAD.

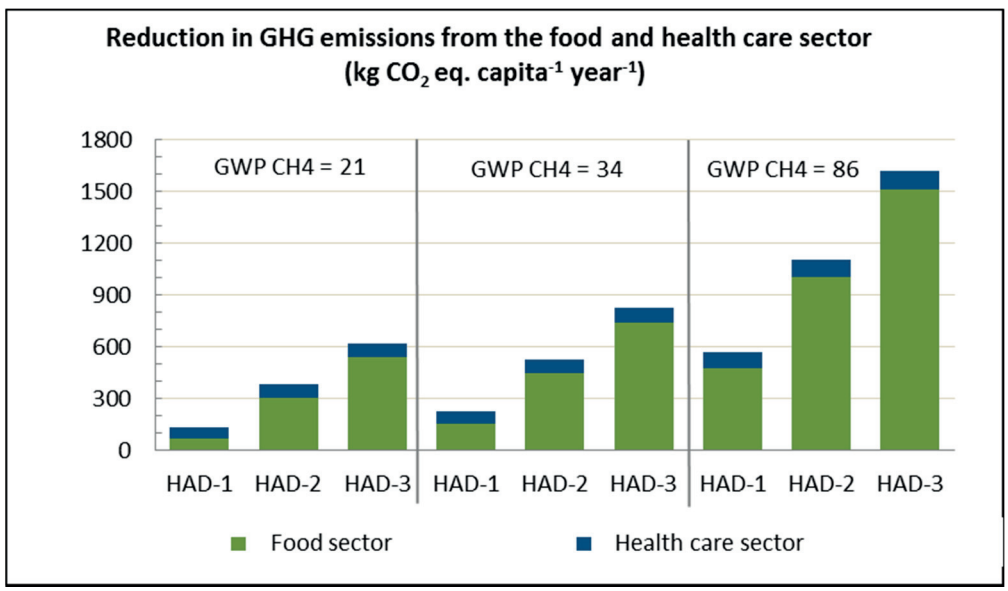

Figure 27. Reduction in GHG emissions in the food and health care sector relative SAD.

6.3.4 Choice of GWP values – Effect on GHG emissions?

The results from **paper III** illustrate how sensitive the estimates of GHG emissions from the diet are to the GWP value for methane. The choice of GWP value has a larger effect on GHG emissions from the food sector, as emissions of methane constitute a small share of total GHG emissions from the health care sector. By using the old 100-year GWP value for methane of 21 instead of the current estimate of 34, per capita GHG emission from the food sector in the SAD is underestimated by 14%. Thus, the use of old and inaccurate GWP values for methane implies a risk for underestimation of both GHG emissions of the diet and the mitigation potential of dietary change. In contrast, using the 20-year GWP value for methane of 86, doubled the GHG emission reductions potential for the HADs, compared with the current 100-year GWP of 34.

6.4 Paper IV

6.4.1 Scientific basis of dietary scenario analyses

In the review, 14 peer-reviewed articles published scientific journals that fulfilled the inclusion criteria, and in total 49 dietary scenarios, were identified and included in the review. Of these, five articles assess the effect of dietary change on both GHG emissions and land use demand; two assess the effect on land use only, and ten assess the impact on GHG emissions only. Five articles were published between 2009 and 2011, and nine articles between 2012 and Febuary 2014 (Table 19). Additional information on the study design and scenarios in the reviewed articles is provided in **paper IV**.

6.4.2 GHG emissions and land use demand of affluent diets

The GHG emissions from the reference scenarios, i.e. the current average diet in the populations studied, ranged from 0.9-1.7 and 1.4-3.2 tons (0.4 tons for Indian diet) of CO_2 eq. per capita per year in the studies accounting for emissions up to farm gate and retail, respectively. The annual GHG emissions for the average EU citizen are around nine tons of CO_2 eq. (EEA, 2012), which means that food consumption is responsible for about 15-35% of the total climate impact (Table 19).

The land use demand of the reference scenarios ranged from 1400-2100 m^2 per capita. This can be compared to the current global per capita availability of agricultural land, which is about 7000 m^2 (divided approximately as 30% arable land and 70% pasture) if global croplands are assumed to be distributed equally across the population (Table 19).

Table 19. Summary of results adapted from articles reviewed in paper IV

SCENARIO	REDUCTION OF GHG EMISSIONS			REDUCTION OF LAND USE DEMAND		
	(%)[1]	(kg CO_2 eq./yr)[3]	(n)	(%)[2]	(m^2/yr)	(n)
Vegan diet	25-55	760 (520-1090)	6	50-60	970 (690-1160)	3
Vegetarian diet	20-35	540 (110-1110)	7	30-50	790 (570-1010)	2
Ruminant meat replaced by monogastric meat	20-35	560	2	-	-	-
Meat partially replaced by plant-based food	+5-0	+20 (+40-0)	2	15	220	1
Meat partially replaced by dairy products	0-5	40 (30-50)	2	-	-	-
Meat partially replaced by mixed food	0-5	80 (40-110)	2	-	-	-
Balanced energy intake	0-10	100 (40-160)	2	-	-	-
Healthy diet	0-35	210 (+40-490)	14	15-50	590 (310-940)	6

[1]Effect of dietary change on GHG emissions from the diet, in % of reduction in GHG emissions of current average diet. [2]Effect of dietary change on demand of land, in % of reduction in total demand of agricultural land of the average diet. [3]Average effect (minimum change-maximum change), n = number of scenarios. "+" indicates an increase in GHG emission or land use demand.

6.4.3 Potential for GHG emission reduction

The impact of dietary change on GHG emissions from diet is summarized in Table 19 and Figure 28. Completely avoiding all animal-based products (vegan) provides the largest potential for reducing GHG emissions from the diet, followed by scenarios of avoiding all meat (vegetarian), replacing ruminant meat with pork and poultry and eating a healthier diet.

The results from **paper IV** suggest that the potential to reduce the total per capita GHG emissions via dietary change is about 4-20% for a transition to a vegan diet, and up to 12% by a transition to either a vegetarian diet, a diet in which ruminant meat has been substituted by monogastric meat or a healthier diet with restricted intake of red and ruminant meat.

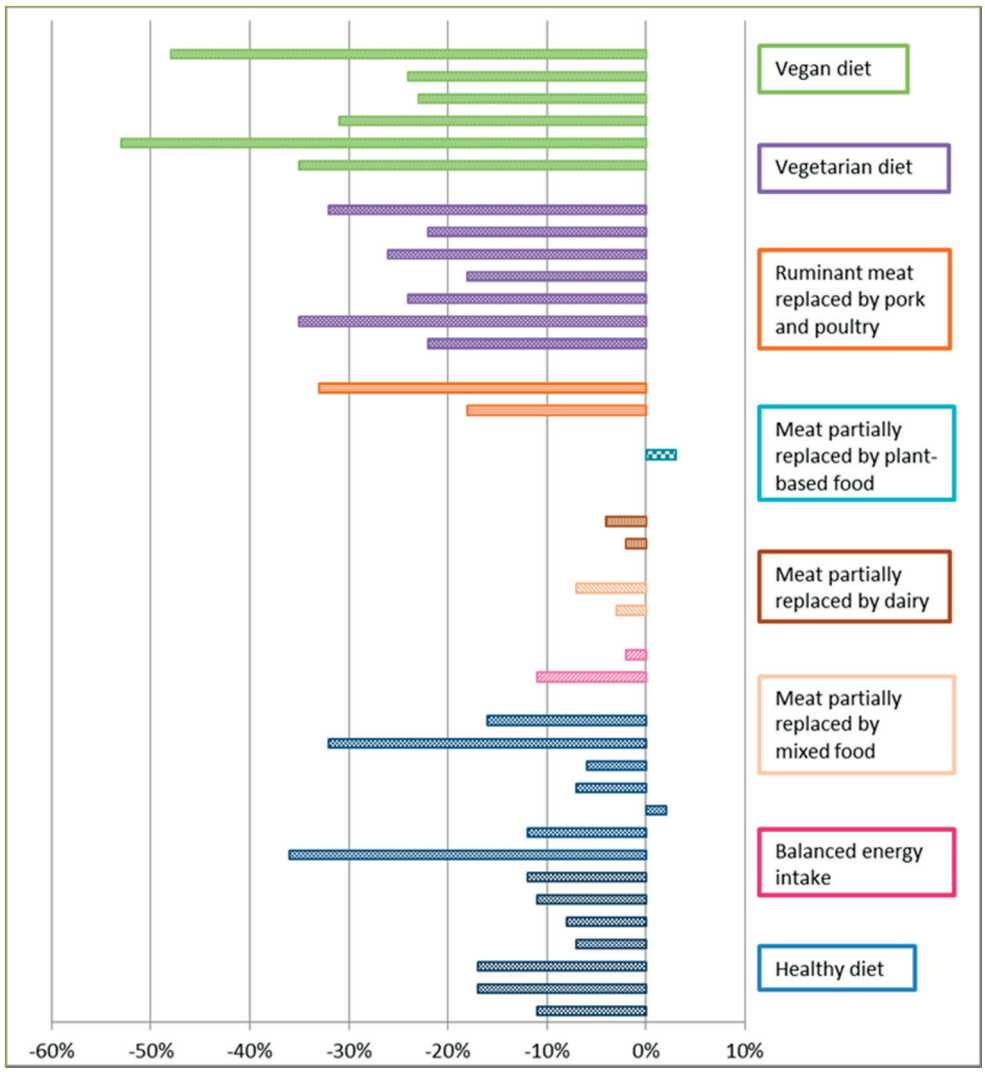

Figure 28. Impact of dietary change on GHG emissions from the diet, in percentage of relative change in GHG emissions compared to the reference scenarios. The data presented are based on the results from 12 articles. References and more information on the dietary scenarios studied are available in paper IV.

6.4.4 Potential for reducing land use demand

The impact of dietary change on land use demand by diets is summarized in Table 19 and Figure 29. According to the results, a change to vegan or vegetarian diets has the largest potential to reduce the demand for agricultural land, followed by changing to a healthier diet and diets in which meat is partially replaced by plant-based food.

The potential of the diet to reduce the land demand appears to be largely dependent on the amount of ruminant meat consumed. Substituting all animal products with plant-based food can, according to the results, reduce the land demand of the diet by 50-60%.

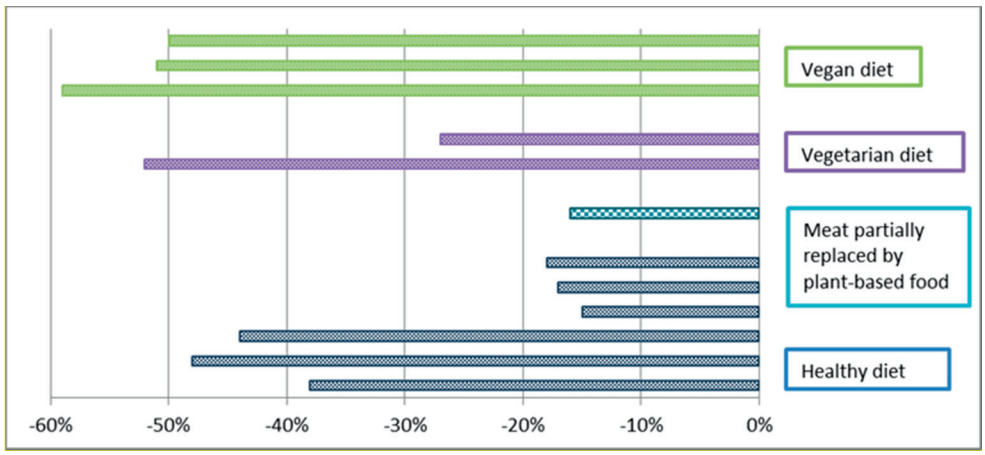

Figure 29. Impact of dietary change on current demand of land from the diet, in % of relative change of land demand compared to the reference scenarios. The data presented are based on results from four articles. References and more information on the dietary scenarios studied are available in paper IV.

6.4.5 Methodological aspects of importance

The methodological approach of scenario analysis can have a decisive effect on its quality and final outcome. Methodological choices identified to be of importance for the quality and outcome of dietary scenario analysis are, i) the method for scenario development, ii) the functional unit, iii) the system boundaries, iv) impact categories, e.g. assumptions on land use change and categorization of land use, and vi) the method for uncertainty analysis. An overview of identified methodological aspects of importance for dietary scenario analysis is provided in Table 20.

i) Scenario development

In the articles reviewed, the reference scenarios are based on data on average per capita consumption of the studied population, i.e. current consumption patterns. The exception is the study by Pathak et al. (2010), in which the reference diet is based on a hypothetical, well-balanced diet consisting of common Indian foods. The reference diets are thereafter modified in order to study the environmental impact of different hypothetical changes in the diet. In some of the articles reviewed (Aston et al., 2012, Berners-Lee et al., 2012, Vieux et al., 2012), not only the reference scenarios but also the studied dietary scenarios are based on registered consumption data, e.g. self-selected diets.

The population studied is in general defined by its nationality. In all the articles reviewed, except for Pathak et al. (2010), the effect of dietary change is studied in European populations characterized by having affluent diets. In a few articles, the reference diets reflect consumption patterns of particular groups of the population, for example, women (Macdiarmid et al., 2012; Temme et al., 2013; van Dooren et al., 2014).

ii) The functional unit

In the articles reviewed, the most common approach to account for the nutritional value of the diet is to use iso-caloric substitution, i.e. that all dietary scenarios contain the same energy content. In addition, some articles design the scenarios so that the diets studied are comparable for other nutrients. Several of the articles also use additional criteria, for example, that the dietary scenarios should be in line with healthy recommendations. In the paper by Temme et al. (2013) the functional unit relates only to the weight of the food. This makes it difficult to evaluate the comparability of nutrient content in the different dietary scenarios. This study, however, quantifies the intake of iron and saturated fatty acids from all the scenarios studied.

iii) System boundaries

Only two of the articles reviewed (Pathak et al., 2010, Tukker et al., 2011) use system boundaries including the production system from primary production to consumer phase. Tukker et al. (2011) also includes emissions from waste disposal. The most common procedure is to set the system boundaries to include emissions produced up to the distribution of the food, e.g. to the stage of retail. In three of the articles (Fazeni & Steinmüller, 2011; Risku-Norja et al., 2009; van Dooren et al., 2014) quantifications of GHG emissions from the diet are limited to emissions taking place in the agriculture phase, i.e. up to the farm gate.

Losses and waste along the food chain are accounted for in various ways in the reviewed articles, for example, by using LCA data that include emissions from all stages up to the retail or consumer level in which emissions from food wasted in upstream processes

are added to the remaining food that becomes available to the consumers. The difference between per capita agricultural supply data and consumption data of actual intake levels is often used as an estimate of the amount of food that is lost and wasted during the lifecycle (Berners-Lee et al., 2012; Hoolohan et al., 2013).

iv) Impact categories

In the articles reviewed, only two articles (Hoolohan et al., 2013; Meier & Christen, 2012) account for GHG emissions from direct land use change, and none for emissions from indirect land use change.

Of the four articles reviewed which include land use demand, only Meier & Christen (2012) report the demand for cropland and pasture land separately. In addition, this article makes a distinction between domestic and foreign production, which is not done in the other articles.

v) Uncertainty analysis

Uncertainty and discrepancy in methodology, data and assumptions, for instance, regarding the aspects mentioned above are identified as key factors affecting the reliability of the results of dietary scenario analysis. In addition, variability in results may be due to geographical, temporal or technological variability in the input data (Björklund, 2002; Huijbregts, 2002).

Of the 14 articles reviewed, Vieux et al. (2012) is the only one to perform an uncertainty analysis (Monte Carlo Analysis) of the results.

Table 20 Methodological overview of the articles reviewed in paper IV

Author(s)	Year of publication	Uncertainty analysis?	Nutritional considerations in function unit?	System boundary	Impact category		
					Environmental indicator	dLUC/iLUC?	Specification of land use?
Van Dooren et al.	2014	No	[1]Rec. intake	Cultivation – farm gate	GWP, LU	No/No	No
Hoolohan et al.	2013	No	Energy	Cultivation – retail	GWP	Yes/No	
Saxe et al.	2013	No	Energy, protein	Cultivation – retail	GWP	No/No	
Temme et al.	2013	No	Mass		LU		No
Aston et al.	2012	No	[2]Rec. intake	Cultivation – retail	GWP	No/No	
Berners-Lee et al.	2012	No	Energy	Cultivation – retail	GWP	No/No	
Macdiarmid et al.	2012	No	[1]Rec. intake	Cultivation – retail	GWP	No/No	
Meier & Christen	2012	No	Energy	Cultivation – retail	GWP, LU	Yes/No	Yes[5]
Vieux et al.	2012	Yes[4]	Energy	Cultivation – retail	GWP	No/No	
Fazeni & Stenmüller	2011	No	[1]Rec. intake	Cultivation – farm gate	GWP	No/No	
Tukker et al.	2011	No	Energy, protein, fat	Cultivation–waste disposal	GWP	No/No	
Arnoult et al.	2010	No	Energy		LU		No
Pathak et al.	2010	No	Energy	Cultivation – consumer[3]	GWP	No/No	
Risku-Norja et al.	2009	No	Energy	Cultivation – farm gate	GWP	No/No	

[1]Healthy diet that meets energy and nutrient recommendations, [2]In the studied scenario the proportions of vegetarians in the population doubled and remaining population adopted a climate friendly diet, low in red and processed meat, [3]Including food preparation in the household, [4]Monte Carlo Analysis, [5]Distinction between cropland and grassland, domestic and foreign production.

6.4.6 Gaps of knowledge

The assessment of key methodological aspects in dietary scenario analysis showed that this can be performed in various ways and that the choice of method can affect the scientific quality and outcome of the study. Although there are still gaps in our knowledge, the increased number of publications in this area has contributed significantly to a better understanding of sustainable production and consumption of food.

In **paper IV**, four aspects are identified as obstacles in research and policy development of sustainable food production and consumption, and on which more knowledge is required: i) distinction between individual, regional and social level, ii) differentiation of plant-based scenarios, iii) differentiation of agricultural land, and iv) improved assessment and incorporation of uncertainty. These, and other research gaps identified in the thesis, are further discussed in chapter 8.

7 Interpretation of results

In this chapter the results from this thesis are further assessed and interpreted to answer the research questions previously presented in section 2.2. The chapter discusses the uncertainty and reliability of the results (7.1), the potential to improve sustainability in the food system via dietary change (7.2), synergies and conflicts between adoption of healthier diets and diets with lower environmental impact (7.3), and integration of health and environmental methodology in sustainability assessments of food (7.4).

7.1 Uncertainty and reliability of results

The methods used in this thesis include simplifications, assumptions and uncertainties that must be accounted for when interpreting the results. The results are largely dependent on the quality of underlying data, for example, on the consumption and production, losses and waste, nutrient content, relative risk of health effects, GHG emissions and land use demand of food. In this thesis, uncertainty in the methods and results are handled by using validated data of high quality whenever possible, by assessing the uncertainty in results by Monte Carlo analysis, by aiming for high transparency in the presentation of the methods, and by incorporating the estimated uncertainty in the interpretations of the results.

7.1.1 Uncertainty in nutrient intake

Main factors affecting the reliability of the assessment of nutrient intake include uncertainties related to the food consumption data used and the data on the nutrient content of the foods analyzed.

To examine and calculate the nutrient intake in an accurate and reliable way requires, primarily, that the estimated amounts of food consumed in the reference scenarios are representative of the average consumption in the studied population. This can be challenging, given the uncertainties associated with food consumption surveys and statistics. Sources of uncertainties related to food consumption data are thoroughly described in **paper I**. In **papers II** and **III**, the nutritional calculations are based on food supply data adjusted for losses and waste prior to consumption. This has the advantage that uncertainties related to under-, over- and mis-reporting of food intake are to a large extent avoided. On the other hand, this approach introduces other uncertainties, for example, related to the calculations of what proportion of the food supply is lost or wasted and actually eaten. Another drawback of using per capita supply data is that they often provide less specific information on the different types of food consumed, and hide large variations between different groups of the population. The choice of using per capita food consumption data and evaluating the estimated intake levels of nutrients against reference values of the recommended dietary intake for the whole population, constrain the assessment to study the effect on nutritional status on a public health level and not in specific groups of the population or in individuals.

In **paper II**, the losses between production and the consumer are assumed to represent on average 5% of the bone-free carcass weight, and are further adjusted to exclude small bones remaining at sale. However, since no data of sufficient quality were available to adjust for household losses, meat wasted at consumer level is not accounted for. Thus, for the quantification of nutrient intake the per capita supply of meat available at household level is assumed to correspond to the actual intake of meat. According to a

previous report, the average percentage of meat wasted in Europe during post-harvest handling and storage, processing and packaging and distribution are 0.7%, 5% and 4%, respectively (percentage of quantity entering each step) (Gustavsson et al., 2011). The same report estimated that on average 11% of all purchased meat in Europe is wasted in the household. Underestimation of the losses between production and consumption would imply a lower actual intake of nutrients per amount of meat produced. In **paper III**, the adjustment for food losses and waste are based on specific waste estimates for each food category provided by the USDA, which also include household waste and plate waste (USDA ERS, 2014).

The precision in nutrient calculations also depend on the level of detail in information on the food consumption patterns between and within food categories. In **paper II**, a detailed assessment is made of how the consumption of different meat products is distributed. Despite the differentiation between meat products, it is uncertain how representative these data are since the data available for use were old (1993) and thus do not capture changes in consumption patterns during the past two decades. The simplification that meat consumption only comes from beef, pork or chicken, adds some further uncertainty to the nutrient calculation. In Sweden beef, pork and chicken represent about 93% of total meat consumption (SBA, 2011). In **paper III**, the proportion of meat consumed as processed meat, and grains consumed as whole grains and refined grains is based on estimates from the literature. The energy content of the scenarios analyzed is based on the energy content of a selection of foods or on averages from each food category included in the analysis, and are thus only approximate estimates. However, the quality of the assessment is deemed sufficient for the purpose of validating the energy content in the analyzed scenarios, and to correspond to the quality of similar previous assessments.

In **paper III**, the approach for replacing meat with plant-based protein is based on a USDA framework in which the nutritional interchangeability of plant-based and animal-based protein is estimated. According to this framework one ounce equivalent of cooked beans and peas (39 g) is nutritionally interchangeable with one ounce equivalent of cooked meat (28 g). However, no further information is found about which nutritional aspects are considered. A rough validation of the framework, based on the content of energy, protein, iron and zinc in five commonly consumed beans and peas and meat products confirm that the nutrient content in the amounts of food previously stated are in the same range, with the exception of a higher protein content in the meat products. However, the results of such a comparison are highly dependent on the food products selected to represent each group.

Additional uncertainties in the nutrient calculation include the quality of data on the nutrient content of foods. As described in section 5.5.1, data on the nutrient content of a food provided by nutrient databases may be impaired by uncertainties. For example, as each food product typically is indicated by one value the potentially large variation in nutrient content within the same food product is usually not captured. In **papers II** and

III, the energy and nutritional calculations are based on data from the Swedish Food Agency´s food data base, which is judged to be a reliable data source. In **paper III**, the use of an American nutrient database would have provided more precise estimations of the energy content in the diets analyzed. However, since all dietary scenarios are analyzed with the same nutrient database this should not affect the standardization of the energy content of the diets. In **papers II**, the nutrient intake is quantified on the basis of the nutrient content of the raw meat product, which means that changes in nutrient composition during preparation and cooking are not taken into account. In general, cooking has a limited effect on the nutrients studied but may influence, for example, fat intake and result in leaching of minerals if the meat is boiled. In **paper III**, no estimation of the nutrient content, besides energy, is made since the uncertainty of such a calculation is judged too large for the results to be of value. Finally, the intake and uptake of nutrients depend on a range of factors including variations between individual eating habits (e.g. whether there is a preference for eating the chicken with or without skin), the combination of food within the diet, and the current nutritional and health status of the individual, which are not accounted for in this thesis.

In **paper II**, the uncertainty intervals for most nutrients are estimated to be in the range of ± 50% but for some they are even larger. No uncertainty assessment is made for the nutrient calculation in **paper III** since its purpose is limited to validate the energy intake in the studied scenarios.

7.1.2 Uncertainty in health effects

The reliability of the estimated health effects of dietary change is mainly affected by the uncertainty related to the relative risk estimates used and the methodology for estimating the combined effect of several changes in the diet.

In **paper III**, the estimated health effects are quantified based on relative risk estimates from epidemiological studies. Because most diseases are caused and influenced by multiple factors, diet-disease relationships are very complex to analyze. Besides the quality of diet, disease development is affected by genetic factors, physical activity, use of tobacco and alcohol and other lifestyle choices (WHO, 2009). Therefore it may be difficult to isolate the effect of the diet. The risk that an observed association is due, totally or in part, to the effects of another factor which correlates with the exposure is referred to as confounding (Ahlbom, 2006). The risk of double counting the health effects is of special concern when combining relative risk estimates as in **paper III.** To minimize this risk, relative risk estimates are selected from meta-analyses that adjusted for influencing confounders, such as other types of food intake, physical activity level and history of disease. Despite these efforts, some risk of double counting remains, meaning that the health effects reported may be overestimated. For increased transparency, the health effects are presented for each dietary factor individually as well as for the combined effect

of all dietary changes. The uncertainty intervals for the reduction in relative risk of disease in the scenarios studied range from approximately – 35% to + 50%.

A major assumption in the estimated changes in health care costs is that the changes in relative risk of disease are directly related to health care costs. In reality, diet change would only affect disease prevalence over time via reduction in incidence. The results from **paper III** should therefore be interpreted as theoretical estimates of the disease prevalence attributable to the adoption of healthier diets over time, or as the health care costs associated with a counterfactual scenario where the healthier diets have always been consumed.

Estimating the effect on public health and on health care costs in counterfactual and future scenarios is very complex as the changes in other influencing parameters are uncertain and thus cannot be estimated with high certainty. For example, interventions for lifestyle changes in the population also demand resources. Furthermore, healthier diets and people would increase the life span of the population, with increasing health care requirements at higher ages. Such factors may reduce the savings in health care costs estimated in **paper III**. Other factors influencing the health care costs include the size and distribution of the population and the present economic situation. Thus, for a more complete assessment of public health and health care costs many more parameters need to be accounted for.

7.1.3 Uncertainty in GHG emissions

Uncertainty factors in LCA can broadly be categorized into uncertainties in data and uncertainties due to methodological choices. Data uncertainties can be due to the use of inaccurate or unrepresentative data, unavailable or missing data, or inherent variability, for example, biogenic emissions of nitrous oxide. Methodological choices that can affect the reliability are, for instance, the choice of functional unit, system boundaries, allocation procedure and characterization method. Variability in results can also be due to geographical, temporal or technological variability in input data (Björklund, 2002; Huijbregts, 2002).

LCA data in general express the environmental impact of food products per amount of food produced. Therefore, the reliability of LCAs of food, in part, depends on the accuracy of the estimated amounts of food required to be produced in order to supply the analyzed diets. A substantial share of total GHG emissions from food production are estimated to be embodied in food which is not eaten but wasted (Heller & Keoleian, 2014). The approach used to estimate food production and losses along the food chain is previously described in sections 5.4.2 and 7.1.1. According to the American waste data (USDA ERS, 2014) used in **paper III**, the proportion of total red and processed meat, vegetables, fruits and grains lost and wasted in the household is estimated to be 25%, 35%, 55%, and 22%, respectively, with large variations within each food category. The

accuracy of these estimates has a large impact on the estimated GHG emissions from the food sector given in **paper III**. In **paper II**, the GHG emission from NUTR-1 and NUTR-2 may be underestimated as household waste is not accounted for. Underestimation of the losses between production and consumer would imply a higher environmental impact per amount of food consumed.

The reliability of estimated GHG emissions from food consumption and production are also highly dependent on the choice and quality of LCA data. The GHG emissions related to specific food items may vary significantly according to different data sources, due to differences in production systems, regional conditions and methods used to produce the data. The overall uncertainty in GHG estimates for food is estimated to be in the magnitude of ± 10-30%, or more (Röös, 2013). The uncertainty in LCA data is accounted for by using LCA data from reliable sources and by estimating the uncertainty in GHG emissions for the food categories analyzed. The method for uncertainty analysis used in **papers II** and **III** is described in section 5.7.2. Based on the Monte Carlo analysis the uncertainty ranges for GHG emissions in the scenarios studied in **paper II** range from approximately -15% to +85%. In **paper III**, the uncertainty ranges for GHG emissions in the food sector range from roughly - 35% and +30%. Based on the uncertainty in RR data, the reduction in GHG emissions from the health care sector may be ± 20-25%.

The quality of GHG emissions estimates also depends on how well the LCA data represent the food production analyzed. In **paper II**, the origin and production method of the meat consumed in Sweden is thoroughly assessed and matched with corresponding LCA data. In total, the assessment distinguishes between nine production systems for beef and two production systems for pork and chicken. In **paper III**, GHG estimates are collected for 32 different foods and food groups. Further differentiation is made between some production and transportation systems. However, the assessment of variation in GHG emissions, due to differences in production system and origin, is less detailed than that in **paper II**. In addition, uncertainty in the GHG emission calculations results from the limited availability of representative LCA data to be found in the literature. For example, because GHG emission data from US production were limited, GHG calculations are based on LCAs of foods produced both in and outside the US. For higher precision, regional or country level LCA data are required, which to a large extent are currently lacking.

In **paper III**, there are also some uncertainties associated with the estimated GHG emissions from the health care sector. The IO-LCA methodology has some limitations, particularly its aggregate-based assignment of GHG emissions for economic activity in a given industry/sector, as opposed to process/product based assignment. Looking at broader aggregates, some factors are adjusted for, such as the decrease in carbon intensity for the overall economy. However, it is not clear that the health care sector would experience the same rate of decrease in carbon intensity as the overall economy. Some components of the LCA, such as pharmaceutical manufacturing, leave out potentially important factors that could add to the GHG emissions of a given health care

expenditure. Uncertainty is also introduced due to the use of proxy data in the case GHG data for an activity was missing. In addition, the risk of rebound effects of health care cost savings being used for equally, or more, carbon intensive activities is not accounted for.

Estimations of GHG emission also largely depend on the choice of system boundaries. In **paper II**, the assessment includes GHGs produced up to the farm gate, hence emissions from pre- and post-slaughter transports and slaughtering, packaging, storage and preparation are not included. The environmental impact per kg of meat is generally small for these stages compared to the agricultural phase (Sonesson et al., 2010). However, including these emissions would result in slightly higher GHG emissions. In **paper III**, GHG emissions are accounted for up to the stage of retail. In addition, the GHG emission from land use changes, e.g. from deforestation, and carbon sequestration in pastures are not accounted for. Emissions of GHGs from direct and indirect land use change are suggested to contribute substantially to the climate impact of agricultural products (Cederberg et al., 2011; Flysjö et al., 2012; Ponsioen & Blonk, 2012; Schmidinger & Stehfest, 2012). Therefore, the emissions of GHGs in **papers II** and **III** are assumed to be underestimated. Inclusion of emissions from land use change may increase the GHG emissions from the diet by a magnitude of 10-30% (Röös et al., 2015a). However, the estimated effect of including land use change varies greatly depending on the assessment method used. Another potential direct and indirect aspect of land use not considered in this thesis is carbon sequestration. As for the quantification of increased GHG emissions from land use change, the methods and assumption used to estimate the positive effect from carbon sequestration vary, as well as its estimated potential for GHG mitigation (Garnett, 2011). In future studies these aspects are important to be considered.

The results from **paper III** show that dietary GHG emissions are highly sensitive to the choice of GWP value for methane. Using the old GWP value for methane of 21, instead of the new estimate of 34, may underestimate GHG emissions from the diet by roughly 15%. The fifth IPCC assessment report, in which the most recent GWP values are reported, was published in 2013 (Myhre et al., 2013). **Paper II** and the articles reviewed in **paper IV** were published before the new IPCC report was published and therefore use the lower GWP estimates for methane in their calculations. This means that the climate impact of the diets analyzed which contain ruminant meat is likely to be higher than what is estimated in the respective papers. **Paper III** further illustrates that GHG emissions from the diet calculated over a 20-year period, instead of a 100-year period, are substantially higher. In **papers II** and **IV**, GHG emissions are only analyzed over a 100-year period.

7.1.4 Uncertainty in land use demand

Uncertainty aspects related to the assessment of land use demand are to a large extent the same as those previously described for GHG emissions. Aspects on uncertainty in the methodology of importance for LCAs, and uncertainty related to the estimated production of food are described in section 7.1.3. As for GHG emissions of food, there is a large variability in land use demand within the same food groups, especially for livestock products, depending on the production system and type of land used. In **paper II**, the uncertainty intervals for land requirement in the scenarios studied range roughly from -25 to +110%. Thus the potential land areas required could be either smaller or reach up to twice the estimated amount.

A limitation in this thesis is that the proportion of agricultural land consisting of cropland and pasture land is not assessed. This information would have contributed to further knowledge about the actual impacts of the land used. A rough estimate suggests that grazing land represents between one third and one fifth of the land use demand of total Swedish meat consumption, in the scenarios studied in **paper II**. However, not all of the land used for grazing is pasture land, as grazing also takes place on cropland.

7.1.5 Other limitations

In this thesis the environmental impact of dietary scenarios is assessed only based on the emissions of GHG and demand of agriculture land. These aspects can often, but not always, serve as indicators of other environmental impact categories such as eutrophication, acidification and loss of biodiversity (Rockström et al., 2009; Röös et al., 2013; van Doreen et al., 2014). For a more complete assessment of the environmental impact of the diet, other environmental impact categories also have to be included. Furthermore, in this thesis, the nutritional assessment is limited to study the effect of making changes to selected food groups in the diet. Thus, the assessment of nutrition and health effects is limited in the selection of nutrients and diseases studied. A more complete assessment of the environmental and health effects of dietary change further requires that the complete diet is analyzed. Within the wide concept of sustainable food production and consumption (section 2.3.1) several other aspects, of ecological, social and economic dimensions are included. These aspects, however, go beyond the scope of this thesis. Thus, future interdisciplinary and holistic assessments of the diet should include more sustainability aspects.

7.2 Potential for improved sustainability via diet change

7.2.1 Dietary change for more sustainable diets

The results from this thesis suggest that reduced intake levels of meat, and in particular of red and processed meat, and increased intake of vegetables, legumes, whole grains and fruits, can improve the sustainability of the food system from the perspectives of health, climate and land use in regions with unrestricted diet. In addition to the food categories studied in this thesis, a reduced intake of empty calories from cookies, candy, sweet drinks etc., and high fat dairy products is suggested, in combination with a balanced energy intake, for more sustainable diets. The suggested changes are in line with existing dietary recommendations from nutrition and health perspectives (NCM, 2014; Reynolds et al., 2014; USDA, 2015). The findings are also in accordance with the general perception of the dietary composition of more sustainable diets (Figure 30) (Bajželj et al., 2015; FAO, 11, Garnett, 2014).

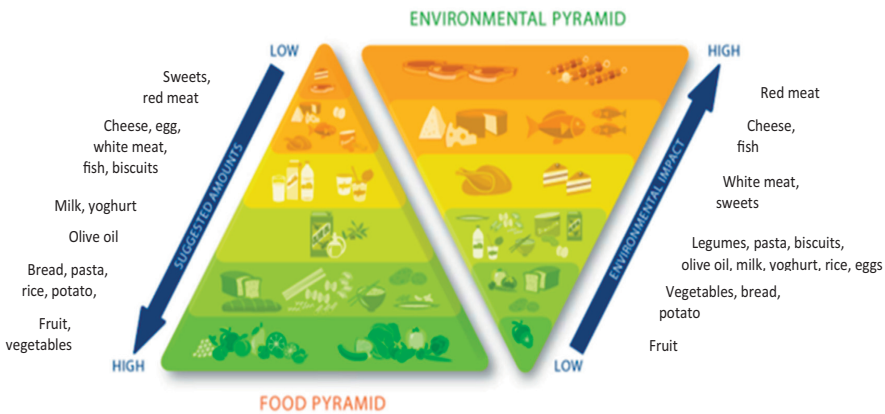

Figure 30. Illustration of dietary changes recommended for healthy diets with less environmental impact. Figure adapted from (Barilla, 2012).

According to the results of **paper II**, current total per capita meat consumption in Sweden is estimated to be 190 g of uncooked boneless meat per day. In cooked weight, this equals a consumption of approximately 135 g of meat per person and day, of which 105 g consists of red meat, and about one third consists of charcuteries. In **paper III**, the US, consumption of red and processed meat (i.e. excluding white meat) is estimated to be about 90 g of cooked meat per day, of which 38% is estimated to be processed meat. The estimated levels of total meat consumption in Sweden and the US, exceed the Swedish and US food-based dietary guidelines of approximately 100 g of cooked meat (total) per

day (Enghardt Barbieri & Lindvall, 2003; USDA, 2010). In addition, current meat consumption levels substantially exceed the public health recommendation of a maximum intake of 300 g of cooked red meat per week (45 g/d), and are well above the recommendation to avoid or limit processed meat as much as possible (WCRF/AICR 2007; 2009; 2011). To meet national and international dietary recommendations for healthy meat intake, current total meat consumption in Sweden should be reduced by approximately 25%. In Sweden and the US, current consumption levels of red meat have to be cut by about half, and the proportion of meat constituted by processed meat must be reduced substantially, to meet dietary health recommendations.

In contrast, consumption of vegetables, fruits and whole grains are often below national and international dietary health recommendations, in populations eating an unrestricted diet. In the US, for example, current consumption of fruit and vegetables is estimated to be 335 g per day, which can be compared to the health recommendation of a minimum of 400-500 g per day (NCM, 2014; WHO/FAO, 2003). Total grain consumption in the US is above the USDA guidelines, and more importantly the proportion between refined and whole grains is skewed in an undesirable manner. In the current US diet, only 10% of the total grain consumption is estimated to come from whole grains, while at least half of the grain consumption is recommended to come from whole grains in a healthy diet (USDA, 2010). Swedish consumption of plant-based food is not estimated in this thesis. However, the latest dietary survey of food consumption patterns among Swedish adults suggests that only 20% of the Swedish population meet recommended intake levels of fruits and vegetables and 90% of the population eat insufficient amounts of whole grains (Amcoff et al., 2012). The estimated dietary patterns in Sweden and the US are characteristic also for other regions where the population has unrestricted diets (Popkin, 2006; WHO/FAO, 2003).

7.2.2 Potential for improved nutritional quality and public health

The results from this thesis indicate that adoption of more sustainable diets offers the potential to improve the nutritional quality and public health in regions with unrestricted diet. Positive effects on the nutritional quality from the adoption of more sustainable diets include a reduced intake of dietary energy, total and saturated fat and salt, as well as an increased intake of fiber, and a range of nutrients from the higher intake of plant-based food. The improved nutritional quality in diet is associated with reduced risk of several non-communicable diseases, for example, coronary heart disease, diabetes type II and cancer.

Over-consumption of energy and resultant overweight is considered the largest individual risk factor for non-communicable disease and death in many regions in the world (WHO, 2009). In Sweden, meat provides about 11-13% of total dietary energy. Total average energy intake in Sweden is estimated to be 8.4-13.7 MJ (Amcoff et al., 2012; SBA, 2013). This can be compared with the estimated energy requirements of 9.7-11.7 MJ and 8.1-

9.4 MJ for adult men and women with sedentary lifestyles, respectively (NCM, 2014). In **paper II**, a 25% reduction of Swedish meat intake is estimated to reduce the total energy intake by 0.7-0.8 MJ per day. For those in the population with an excessive energy intake, the reduction is positive, while the proportion of the population who has a too low energy intake, or a healthy energy intake and wishes to keep weight balance, the reduction in energy needs to be compensated for by an increased intake from other food groups.

Positive health effects are also expected from the reduction in total and saturated fat. In Sweden, fat is estimated to contribute 35% of the total energy, which is in the range of the recommended intake (25-40 E%). On the other hand, 80% of the population is estimated to exceed the recommended upper intake levels of saturated fat (10 E%) (Amcoff et al., 2012). Meat and meat products are estimated to contribute 19-22% of total fat, and 19% of saturated fat in the current Swedish diet (Amcoff et al., 2012; SBA, 2013). Based on the results from **paper II**, adoption of healthier meat consumption in Sweden according to the scenarios studied is estimated to reduce the intake of total fat and saturated fat in the average Swedish diet by approximately 10-25% and 20-25%, respectively. From a health perspective, the reduction in animal fat can be replaced to advantage by plant-based fat containing a higher proportion of mono- and poly-unsaturated fatty acids. Such a change in the diet is likely to also reduce the intake of trans-fatty acids, the intake of which should be kept as low as possible (NCM, 2014). Reduced intake of saturated fat and positive health effects thereof, due to adoption of diets with lower shares of animal-based food and higher shares of plant-based food, have been documented in several other research papers (Friel et al., 2009; Scarborough et al., 2012; Temme et al., 2013; Westhoek et al., 2014).

In this thesis, the nutritional assessment is restricted to a limited number of nutrients. Assessments of the nutritional effect of the adoption of healthier diets which include other nutrients are made, for example, by Röös et al. (2015) and the Nordic Council of Ministers (2012). The findings of these studies suggest that further positive health effects can be expected from reduction of salt and from increased levels of, for example, fiber and folate, if meat is replaced by an increased intake of plant-based food.

In **paper III**, adoption of healthier diets in the US is estimated to reduce the relative risk of coronary heart disease, diabetes type II and colorectal cancer by 20-45%. Altogether, the reduced risk of disease can be translated to potential savings in the magnitude of US$ 54-72 billion per year in health care costs, equivalent to 20-30% reduction of total yearly US health care costs. Positive health effects from adoption of more sustainable diets have also previously been estimated (Aston et al., 2012; Friel et al., 2009; Scarborough et al., 2011; Tilman & Clark, 2014). Based on the findings of these studies, the change from a typical "western diet" to a diet in which meat and dairy is reduced and replaced by an increased intake of plant-based food is predicted to have substantial benefits on prevention of cardiovascular disease, cancer and diabetes type II. The health effects of partial or total exclusion of meat in the diet also been analyzed in several epidemiological

studies (Alexander & Cushing, 2011; Aune et al., 2009; Chan et al., 2011; Ferguson, 2010; Feskens et al., 2013; Micha et al., 2010; Wang, 2015; WCRF/AICR 2007; 2011; Wyness et al., 2011). The results confirm the findings from this thesis of a link between red and processed meat consumption and increased risk for cardiovascular disease, cancer, diabetes type II, and overall mortality. Also the beneficial health effects of plant-based diets, with high levels of vegetables, legumes, fruits, berries, nuts and whole grains are well documented (Aune et al., 2013; Mente et al., 2009; NCM, 2014; WCRF/AICR, 2007; 2011; Wirfält et al., 2013). A few studies have further studied the socio-economic effects of diet change e.g. (Saxe, 2014). In the UK the burden of food-related disease has been estimated to be around £6 billion annually (Scarborough, 2011).

7.2.3 Potential for reduced GHG emissions

The results from this thesis indicate that the adoption of healthier diets offers the potential to reduce GHG emission of the diet. The largest potential for reduced GHG emissions in the diet comes from reduced levels of meat consumption, in particular of ruminant meat.

In **paper II**, changes towards healthier meat consumption patterns in Sweden are estimated to reduce current GHG emission from meat in the diet by 0.2-0.4 tons of CO_2 eq. capita^{-1} year^{-1}, corresponding to a 10-20% and 3-5% reduction of the per capita emissions of Swedish food consumption and total private consumption, respectively.

The effect on GHG emissions of replacing the reduced intake of meat with an increased intake of other food groups is not analyzed in **paper II**. However, this effect has been analyzed in another Swedish study (Röös et al., 2015a). In this study the reduction in meat intake is the same as in **paper II**, but this study also analyses the effects of increasing the intake of plant-based food, and making other dietary changes recommended from a health perspective. The results suggest that changing current Swedish dietary pattern to a healthier diet, based on dietary guidelines, would reduce GHG emissions by 0.5 tons of CO_2 eq. capita^{-1} year^{-1} (from 1.9 to 1.4 tons of CO_2 eq. capita^{-1} year^{-1}). Based on these results, changes in meat consumption account for the largest potential for reducing GHG emissions via adoption of healthier diets.

Paper III showed that adoption of healthier diets in the US reduced the GHG emission from the food system by 0.1-0.7 tons of CO_2 eq. capita^{-1} year^{-1}, mainly depending on the amount of red and processed meat allowed in the diet and the choice of GWP value for methane. If the global warming potential instead is evaluated over a 20-year period, the GHG emission reduction potential increases to 0.5-1.5 tons of CO_2 eq. capita^{-1} year^{-1}. By including the emission reduction estimated from the health care sector, adoption of healthier diet in the US is estimated to have the potential to reduce GHG emissions by up to 1.6 tons of CO_2 eq. capita^{-1} year^{-1} (in a 20-year perspective). Total GHG emissions related to food for the average US citizen has been estimated to be 3.1 tons of CO_2 eq.

capita^{-1} year^{-1} (Jones & Kammen, 2011; Weber & Matthews, 2008). Based on this, the changes in diet analyzed could reduce per capita GHG emissions of US food consumption by about 50%, and the total US per capita GHG emissions by approximately 7% (in a 20-year perspective).

The effect of changing the current US diet to a diet corresponding to the USDA dietary guidelines was also analyzed in a study by Heller et al. (2014). In contrast to the methodology used in **paper III**, where only a selection of food groups are analyzed, this study analyzed all changes in the diet required to meet health recommendations. The results from this study suggested that an iso-caloric shift from the current average US diet to the USDA dietary recommendations would increase the GHG emissions from the diet. However, if the energy content in the healthier alternative diet was reduced to correspond to the average energy requirement of the population, the GHG emissions from the two diets were about the same. The larger GHG emissions estimated from the healthier diet were to a large extent due to the substantial increase of dairy consumption recommended by the USDA. In contrast, adoption of the USDA dietary recommendation, adapted to vegetarian and vegan diets resulted in a reduction of 0.4 and 0.7 tons of CO_2 eq. capita^{-1} year^{-1}, respectively, which is in the same range as the results in **paper II**.

The results from the review analysis performed in **paper IV** indicate that GHG emissions (up to retail) from current average diets in regions with unrestricted diet, are in the range of 1.4-3.2 tons of CO_2 eq. capita^{-1} year^{-1}, corresponding to about 15-35% of total GHG emissions in the EU. The results further suggests that dietary change can reduce current GHG emissions from the diet by up to 55%, with the largest potential coming from completely avoiding animal based-products. The potential to reduce the total per capita GHG emissions via dietary change is estimated to be about 4-20% for a transition to a vegan diet, and up to 12% by a transition to a vegetarian diet, a diet in which ruminant meat has been substituted by monogastric meat or, a healthier diet with restricted intake of red meat.

The review in **paper IV** includes papers published up to February 2014. Since then, several interesting papers that analyzed the effect on GHG emission of dietary change have been published (Green et al., 2015; Heller and Keoleian, 2014; Hendrie et al., 2014; Masset et al., 2014; Pairotti et al., 2015; Röös et al., 2015a; Scarborough et al., 2014; Tilman & Clark, 2014). The systematic review performed by Joyce et al. (2014) provides a good complementary overview of the research field up to mid-2014. The findings of these studies are in agreement with the results from this thesis. The GHG emission reduction potential from adopting a healthy diet is estimated to be in the range of 17-26% (Green et al, 2015; Hendrie et al., 2014; Röös et al., 2015a; Tilman & Clark, 2014), with the exception of the lower estimate previously mentioned by Heller and Keoleian (2014). Furthermore, the GHG emission reduction potential from adopting a vegetarian and vegan diet is estimated to be in the range of 13-55% (Heller & Keoleian, 2014; Pairotti et al., 2015; Scarborough et al., 2014; Tilman & Clark, 2014) and 47-60% (Heller & Keoleian, 2014; Scarborough et al., 2014).

Although the maximum potential to reduce current GHG emissions via dietary change is estimated to be above 50% in regions with affluent diet, the question remains whether such changes in the diet are realistic or not. Complete avoidance of meat and dairy products may not be realistic on a population level if cultural and gastronomic considerations are taken into account. Therefore, the maximum potential to reduce current GHG emission from the diet through dietary changes that can be seen as realistic, has been suggested to be limited to 20-40% instead of above 50% (Green et al., 2015; Masset et al., 2014).

While cultural and gastronomic aspects may constrain the potential to reduce GHG emissions from the diet, the ongoing global warming and severe consequences thereof is a threat that cannot be ignored. As previously mentioned in section 4.2.2, 1.5 tons of CO_2 eq. capita^{-1} year^{-1} have been suggested as a theoretical level of sustainable emissions globally in order to reach international climate goals. The results from **paper II** estimate that current meat consumption in Sweden alone, is responsible for emissions representing about 40% of the total budget for sustainable GHG emissions, that also need to cover emission from other food and activities such as housing, transportation and other consumption. **Paper IV** further suggests that in many regions current per capita GHG emissions from the complete diet correspond to the total amount of GHG emissions allowed in order to meet international climate goals. On the assumption that food can account for half of the total permitted per capita emissions, a sustainable level of GHG emission from food can be considered to be roughly 0.75 tons of CO2 eq. capita^{-1} year^{-1} (Röös et al., 2015a). Of the 49 dietary scenarios reviewed in **paper IV**, only three meet this sustainable level of GHG emissions, even though many of them are designed to be more sustainable. Thus, to achieve sustainable levels of GHGs from the food sector seems to require either more drastic changes in the diet, in combination with improved production systems and reduction of food waste, or that other sectors need to bear a greater share of emission reduction.

As described in section 1.1, the agrifood sector is estimated to be responsible for almost one third of total global anthropogenic GHG emissions. The growing global population and the trend of a nutrition transition towards an increased preference for climate intensive foods, foresees that global GHG emissions from the agrifood sector will continue to grow. For example, Tillman & Clark (2014), estimate that global average per capita dietary GHG emissions from crop and livestock production will increase by 32% between 2009 and 2050, based on current trends and forecasts of future per capita income. This means that GHG emission from the agrifood sector would have to be reduced by some 30% to be kept at the same level as today, a reduction equivalent to the estimated GHG mitigation potential in the agrifood sector via agricultural and technical mitigation options (Hedenus et al., 2014). Thus, to reach global climate goals, dietary change, and reduced meat intake in particular, seems to be crucial, in combination with reduced food loss and waste (Garnett, 2011; Hedenus, 2014; Tilman & Clark, 2014).

7.2.4 Potential for reduced land use demand

The results from this thesis indicate that adoption of healthier diets offers the potential to reduce land use demand for food consumption and production. The largest potential coming from reduced levels of meat consumption.

In **paper II**, changes towards healthier meat consumption patterns in Sweden are estimated to reduce current land use demand for the production of meat in the diet of 0.11 ha capita^{-1} year^{-1} by 36-64%. On a national basis this corresponds to a potential of releasing 0.04-0.07 ha capita^{-1} year^{-1} of agricultural land, or an area equivalent to about 12-20% of the total agricultural areas in Sweden, that could be used for other purposes.

As previously mentioned, **paper II** does not analyze the effect of increased food consumption of other food groups to compensate for losses in nutrients due to the reduction in meat consumption. Röös et al. (2015) estimated the net effect of reducing current Swedish meat consumption, in accordance with NUTR-1, and at the same time making the additional changes required to meet Nordic dietary recommendations. According to the results, average total food consumption in Sweden requires 0.34 ha capita^{-1} year^{-1}, of which meat occupies 0.2 ha capita^{-1} year^{-1}. Adoption of a diet in line with dietary recommendations is estimated to reduce the land use demand by about 20%, to approximately 0.27 ha capita^{-1} year^{-1}. In comparison to the results in **paper II**, Röös et al. (2015) estimates the land use demand of the complete Swedish diet to be about three times higher. Also the land use demand of Swedish meat consumption is estimated to be higher, and the potential to reduce land use demand by adopting a healthier diet is estimated to be about the same or higher, depending on the scenario used for comparison.

The results of the review analysis performed in **paper IV** indicate that the land use demand of current average diets, in regions with unrestricted diet are in the range of 0.14-0.21 ha capita^{-1} year^{-1}. This can be compared with the current availability of global agricultural land which is about 0.7 ha capita^{-1} year^{-1}, of which 70% consist of pasture land. The potential for reducing land use demand via dietary change is estimated to be as much as 60%. The largest potential comes from completely avoiding animal-based products (vegan diet), while the potential is estimated to be up to 50% for a transition to either a vegetarian diet, or a healthier diet based on dietary guidelines.

Since the review in **paper IV** was made, additional papers analyzing the land use demand of diets have been published (de Ruiter, 2014; Meier et al., 2014; Mulik & O´Hara, 2014; Röös et al., 2015a; Thaler et al., 2015; Tilman & Clark, 2014; Van Kernebeek et al., 2015; Westhoek et al., 2014). The results from these studies are in agreement with the results of this thesis. Per capita land use demand in European counties is estimated to be in the ranges of 0.14-0.32 and 0.16-0.19 ha capita^{-1} year^{-1}, based on two separate studies (de Ruiter, 2014; Kastner et al., 2012). Adoption of healthier diets based on dietary recommendations, in regions with unrestricted diet, is estimated to have the potential to reduce current land use demand from the diet by 15-31% (Meier et al., 2014; Röös et al., 2015a; Thaler et al., 2015). Meier et al. (2014) further estimated that

adoption of a vegetarian diet or a vegan diet can reduce the land use demand from the average German diet by 28% and 44%, respectively.

The availability globally of agricultural land may not be as critical as the emissions of GHG in a shorter term, but is largely dependent on the agricultural intensity, soil fertility, changing climatic conditions and on the future demand for agricultural products for purposes other than food. However, the competition for land suitable for agriculture and crop land in particular, is predicted to increase, accompanied with adverse risks for the environment, health and society in general (Bajželj et al., 2014; Foley et al., 2005; 2011; Smith et al., 2010). Tillman and Clark (2014) estimate the global potential to limit future increase of land requirement through changes in the diet. According to their estimates, adoption of healthy plant-based diets may reduce the area of global agricultural land in 2050 by up to 740 Mha, an area equivalent to about half of the current global cropland. Thus, changes towards more sustainable diets may substantially enhance the ability to produce sufficient amounts of food to feed the growing population in a sustainable manner.

7.3 Synergies and conflicts between health and environmental perspectives

The overall results from this thesis suggest that changes in diet, in line with nutrition and health recommendations, as well as improved nutritional quality and public health can provide additional benefits from environmental perspectives via reduced GHG emissions and land use demand. The results confirm the findings in prior work (Bajželj, 2015; Joyce, 2014; Macdiarmid, 2013; Meier & Christsen, 2012; NCM, 2014; Reynolds et al., 2013; van Doreen et al., 2014).

Whereas the health benefits of increased consumption of fruit, vegetables, legumes and whole grains are more or less indisputable, concern has been expressed for the consequences of reduced meat consumption on nutrition, especially for protein, iron and zinc (Geissler & Mamta, 2011; Millward & Garnett, 2010; Tetens et al., 2013). Thus, the risk of inadequate intake of these nutrients, due to reduced meat consumption, requires a further discussion.

Regarding protein, an aspect to be considered is the digestibility (i.e. the proportion of food absorbed from the digestive tract). The digestibility of vegetable protein is somewhat reduced compared to animal protein, which for individuals excluding animal-based food in their diet may have a slightly elevated requirement of additional protein. In addition, the proportions of amino-acids are better in animal protein than in vegetable protein. However, since all essential amino-acids are found in plant-based food the content of amino acids in different food groups (e.g. cereals and legumes) will complement each other. In general, a varied plant-based diet therefore provides adequate protein of good quality (NCM, 2014). The results from **paper II** suggest that a change towards healthier meat consumption patterns in Sweden would have a minor impact on the nutritional status of protein. Average intake of total protein in Sweden is estimated to be 81-109 g per day (Amcoff et al., 2012; SBA, 2013). This is well above the WHO recommendation of approximately 50-70 g per day (WHO, 2002). On the other hand, protein contributes 14-17% of the total energy intake in the current Swedish diet, which is in the range of recommended intake (10-20 E%). Meat and meat products are estimated to contribute 25-33% of the total protein in the average Swedish diet (Amcoff et al., 2012; SBA, 2013). Based on the results from **paper II**, a 25% reduction of current Swedish meat intake according to the scenarios studied is estimated to reduce the daily intake of protein by about 10%. As inadequate levels of protein are not a problem in Sweden, the reduction is not considered a public health risk. This finding is confirmed by another Swedish study in which the protein intake from the current Swedish diet and a counterfactual healthier diet is estimated to exceed the WHO recommendation by more than 43% (Röös et al., 2015a). Similar results were found in a study which analyzed the effect of replacing 25-50% of all animal products in the European Union by plant-based food, and found that

the average protein intake exceeded recommended intake levels by at least 50% (Westhoek et al., 2014).

Iron is of special concern also due to the higher bioavailability of iron from animal-based food (i.e. heme iron), compared to iron from plant-based foods (i.e. non-heme iron). Phytic acid further limits the absorption of iron from some plant-based foods, for example cereals. On the other hand, vitamin C enhances the uptake of iron from plant-based food (NCM, 2014). Reduction of meat intake, and red meat in particular, can have a substantial effect on the dietary content of iron. Meat and meat products are estimated to provide 21-30% of the total intake of iron in the average Swedish diet (Amcoff et al., 2012; SBA, 2013). For men, the current average intake of iron is estimated to be above recommended levels (128% of RDI). Also, for unfertile women, the current Swedish diet is estimated to provide sufficient amounts of iron (106% of RDI). However, for fertile women the average intake of 9.5 mg of iron per day is below (63%) the recommended intake level of 15 mg per day (Amcoff et al., 2012). Thus, a 25% reduction in current average Swedish meat consumption could imply insufficient intake levels of iron, especially among fertile women, if not compensated with an increased intake of other foods containing iron. The effect of reducing meat consumption on the dietary intake of iron has been evaluated in several other studies (Geissler et al., 2011; Röös et al., 2015a; Temme et al., 2013; Tetens et al., 2013). The results from these studies suggest that replacement of a portion or the whole intake of meat by plant-based food can provide the same, or larger, amounts of iron, although the iron will be from less bioavailable sources. For a sufficient iron intake, a balanced diet including a variety of foods containing iron is therefore emphasized, while the importance of iron from meat sources specifically is considered to be of less importance (Geissler et al., 2011; NCM, 2014).

Also in the case of zinc, the absorption is better from animal-based foods than from plant-based food and is further impaired by the intake of phytic acid (NCM, 2014). In Sweden, meat and meat products are estimated to provide 30% of total intake of zinc (Amcoff et al., 2012). Thus, reduction in meat intake may have a considerable impact on the overall intake of zinc. However, the current average Swedish diet is estimated to be above the recommended intake levels among both men and women (138%, 136% of RDI) (Amcoff et al., 2012). In **paper II**, a 25% reduction of current Swedish meat intake according to the scenarios studied is estimated to reduce the total average zinc intake in the Swedish diet by 20-30%. According to a modelling study carried out in the UK, red meat makes a greater contribution to the overall intake of zinc than of iron (Geissler et al., 2011). The same study further indicates that a reduction of red and processed meat intake in the section of the UK population with high intake levels, to an average of 80 g (cooked weight) per day would not imply a risk of inadequate levels of zinc. However, a further reduction to an average intake of 70 g of cooked red and processed meat per day could imply an increased risk for inadequate intake of zinc among men. In the study by Röös et al. (2015), the intake of zinc was well above recommended levels (150% of RDI) in the current average Swedish diet, as well as in the alternative diet based on Nordic dietary recommendations.

Besides the nutrients analyzed in this thesis, meat and meat products are good sources, and/or provide a large proportion of the total dietary intake of niacin, vitamin B_{12}, vitamin A, vitamin D, phosphorus, natrium, and selenium (Millward & Garnett, 2010; SFA, 2015a). In a varied and balanced diet these nutrients can be compensated for from food groups other than meat. However, it should be kept in mind that some groups of the population (e.g. children, elders, the sick), may have special nutritional needs and pre-conditions for meeting those needs (Millward & Garnett, 2010). In Sweden, low intake levels of vitamin D, and over-consumption of salt is reported as a general problem in the population (Amcoff et al., 2012). Reduced intake of meat could potentially further reduce per capita intake levels of vitamin D, if not compensated by other foods containing the vitamin.

From a climate perspective, it should be emphasized that a healthy diet including a large proportion of plant-based food will not always necessarily have low GHG emissions (Macdiarmid, 2013; Vieux, 2013). As for all food categories, there is a large variation in GHG emissions between different plant-based foods. Some plant-based food, such as air-transported fruits and vegetables, may have GHG emissions as large as, or larger than, some types of meat (Carlsson-Kanyama & Gonzalez, 2009). In a French study, for example, the replacement of meat in the diet by self-selected fruits and vegetables resulted in a small (5%) increase of GHG emissions if the caloric content of the diet was maintained (Vieux et al., 2012). This result illustrates the importance of taking into account the nutritional content when comparing the environmental impact of food. Depending on the composition of the diet, other food groups such as cheese and some types of seafood may also cause high emission levels (Carlsson-Kanyama & Gonzalez, 2009; Sonesson et al., 2010). The magnitude of GHG emissions embodied in the diet will ultimately depend on a range of factors, including the type of product, production method and how the product is handled until it gets eaten by the consumer.

In this thesis land use demand is assessed as the total amount of agricultural land occupied by food production. To fully understand the effect of diet on land use demand requires differentiation between different types of agricultural land, and inclusion of more environmental impact categories. Reduced intake of ruminant meat is found to have the largest potential to reduce the total land use demand of the diet. Due to the substantial land use demand of ruminants, a reduction in land use demand will occur regardless of whether the ruminant meat is replaced by vegetables, or any other type of meat. On the other hand, grazing on land non-suitable for cropping, and livestock production systems, e.g., based on feed from waste and/or other by-products, have been put forward as resource efficient ways of producing food of high nutritional value. In some areas, grazing animals can also contribute to increased biodiversity by keeping landscapes open (Garnett, 2009; Röös et al., 2015b). In addition, monogastric animals depend on cultivated feed which is also suitable for humans. Thereby, replacing beef with pork and poultry may increase the total demand of cropland and the land use competition between humans and animals (Audsley et al., 2010; de Vries & de Boer, 2010). A net gain in cropland is also not obvious if consumption of dairy products is replaced by plant-based

food, or when monogastric meat is replaced by processed vegetarian meat substitutes (Audsley et al., 2010; Stehfest et al., 2009).

The synergies between diets that improve health and reduce environmental impact suggest that diet change can bring great benefits to society. The discussion above highlights some potential conflicts that should be bourne in mind in future analyses, and further illustrates the need of using a broad perspective, including many different aspects when assessing the sustainability of food.

7.4 Integration of health and environmental methodology

For the assessments included in this thesis, methods originating from the traditional fields of nutrition and environmental science, such as nutrient calculation, nutrition epidemiology and life cycle assessment, have been used and integrated. The interdisciplinary approach employed has contributed to an improved understanding of the possibilities and limitations of dietary change to simultaneously improve human health and reduce environmental burden and resource demand, but has also contributed to method development.

Introduction of nutrient calculation and nutrition epidemiology to sustainability assessments of food and diet has successfully proved to enhance the analysis by including the perspective of public health. Inclusion of the public health perspective can contribute knowledgeably on the impact of diet on nutritional status as well as the risk of developing diet-related diseases. In this thesis the system boundaries are further broadened to include the effect of dietary change on health care costs and GHG emissions from the health care sector.

Integration of nutrition and health aspects in sustainability assessments is a new, but growing, research fields. Nutrient calculation is, for example, increasingly being used to standardize the nutritional content of dietary scenarios and/or to estimate the effects of dietary change on nutritional status (Meier & Christen, 2012; Röös et al., 2015a; van Doreen et al., 2014; Van Kernebeek et al., 2014). In a more limited number of studies, nutrition epidemiology and LCA are combined to study the co-effects of dietary change on disease burden and environmental impacts (Aston et al., 2012; Biesbroek, 2014; Scarborough et al., 2012; Tilman & Clark, 2014; Yip et al., 2013)..

A methodological challenge identified when integrating health and environmental methodology is the development of dietary scenarios. For example, estimation of the produced amounts of food required to supply an amount of food consumed, and vice versa, requires a good understanding of food consumption, supply and production statistics and/or access to information, such as data on the proportion of food loss and waste for different food categories, which enables conversion between the two data sets. As previously described, food consumption and production statistics, as well as data on food losses and waste, are often impaired by uncertainty, and presented in a manner neither easily accessible nor straightforward.

Another challenge is the matching of consumption and production data with the corresponding impact data (e.g. LCA data). The choice of functional unit and system boundaries determines which data that should be combined for a correct assessment of the impact associated with food consumption and production. For example, if the functional unit is defined as "1 kg of food purchased" this generally implies that the system boundaries include all emissions produced along the food chain to supply 1 kg of food at retail including the proportion of food wasted before the food reaches the retail.

As an example, if 50% of the produce is lost and wasted between the farm gate and retail, 2 kg of carrots need to be produced in order to supply 1 kg of carrots to retail stage. Depending on the aim of the study, the objective may be to analyze the emissions embodied in 1 kg of carrots purchased, with or without accounting for emissions associated with the carrots wasted before and after purchase. Likewise, to not overestimate the nutritional intake from food consumption, the amount of food wasted prior to intake must be subtracted.

Some of the methodological challenges in diet sustainability assessments are common to the assessment of nutritional-, health- and environmental effects. For example, there may be large variations in both the nutrient content, GHG emissions and land use demand due to differences in agricultural practice, regional conditions and handling along the food chain. The uncertainty in calculations due to these variations is receiving increasing attention in LCA assessments of food but is rarely mentioned in nutritional and health assessments. Today, it is increasingly common to standardize, for example, the energy content of diets the environmental impact of which are being compared. Such calculations are to a large extent dependent on the foods chosen to be representative of each food group. The method used to standardize the energy or nutrient content in dietary scenarios is rarely specified, which is why it can be difficult to assess its quality and reliability. In order to evaluate the nutritional effects of dietary change, extensive nutritional calculations, such as in **paper II**, are increasingly demanded. However, reliable estimations of the nutritional intake require data on how the consumption is distributed between, and within, food categories. In the assessment of both nutritional, health and environmental effects of food intake it is of significant importance that the uncertainty in data and methods is assessed and presented so that the results can be interpreted based on the quality and reliability the assessment.

Well-established methods for performing sustainability assessments of food and diet are available in both the fields of environmental and nutrition science. The challenge of conducting interdisciplinary research is to handle the methods properly for which, generally, expertise in both areas is required. Interdisciplinary research, for example, places high demand on the understanding of the data in order to decide which data to use for which calculation, how data should be processed for use in subsequent calculations, and how to assess and manage uncertainties to evaluate the overall reliability of results and conclusions. Transparent presentation of data and methods facilitates the performance and increases the chances of good quality in interdisciplinary research, as do cross-border collaborations and use of external expertise.

8 Future outlook

This chapter describes some areas identified to be in need of further research. Research requirements identified are summarized in section 8.1 and further described in sections 8.2-8.7.

8.1 Research requirements – a summary

Below are listed areas in need of further research:

- Improved knowledge on how much meat, especially red meat, can be included in a sustainable diet.
- Improved knowledge of the environmental impact of substitutes for and complements to meat.
- Improved knowledge of the effect of dietary change in specific groups of the population, and in different geographical regions with different habits, cultures and conditions.
- More research focusing on ways to prevent the development of unhealthy and resource-intensive food habits in developing countries/regions.
- Further analysis on effects of land use changes in food production and consumption which distinguishes between types of agricultural land and specifies its geographical location.
- Research in which availability of land and conflicts concerning land use are analyzed by the application of integrated, top-down (global/regional/national) and bottom-up (local) perspectives.
- Improved knowledge on the impact of uncertainty and variability in dietary scenarios.
- Improved knowledge of policy instruments for more sustainable food consumption and production and their effectiveness in different regions.
- For a more complete understanding of how sustainability can be improved in the food system, holistic assessments which include more perspectives and further broaden the system boundaries are required.

8.2 Sustainable protein

A growing body of literature suggests that reduced meat consumption can offer multiple benefits, including reduced environmental impact and improved public health, in populations with unrestricted diets. However, the level to which meat consumption needs to be limited in order to be sustainable, and what should replace it, is still under discussion.

Products that can replace or supplement meat in the diet include plant-based meat substitutes (legumes, cereals, vegetables, nuts), processed vegetarian meat-substitutes (quorn, tofu, tzai, tempeh etc.), and animal-based meat substitutes (dairy, eggs, fish, insects, cultured meat). In addition, there are alternatives such as nutrient fortification and supplementation.

The majority of studies analyzing the sustainability of diet have so far focused on the effect of replacing parts of meat consumption with plant-based foods. More limited knowledge is available on the environmental and health effect of increasing the intake of less traditional foods, such as processed vegetarian meat-substitutes, insects and cultured meat. The potential and limitations for increasing sustainability of diets with consumption of alternative and sources of protein requires further analysis. There is also a need for more interdisciplinary and holistic analyses of other options to increase sustainability in the diet, such as replacing meat with other animal-based foods, e.g. healthier meat alternatives, fish, eggs or dairy products, or including fortified foods or supplements in the diet to meet the requirements of specific nutrients.

8.3 Differentiation on individual, regional and social levels

The general approach in dietary scenario analysis is to base the reference scenario on the average per capita consumption in the population studied. The diet registered in the reference scenarios is thereafter modified to examine the impact of different hypothetical dietary changes on the environment. As previously discussed in this thesis, per capita consumption statistics conceal information about different characteristics between groups of the populations. Since consumption patterns and nutritional requirements differ depending, among other things, on gender, age and physical activity level, it would be of value to do more research on specific groups in the population.

To analyze the effects of diet change in a particular group of the population has the advantage that the risk of, for example, malnutrition can be focused on the group with the highest risk (e.g. fertile women and iron). Men generally eat more meat and their diet has a higher environmental impact than women´s (Meier & Christer, 2012). Due to the higher consumption of meat and the lower requirements for iron, meat consumption could probably be reduced more in men´s diet than in women´s. This example shows that, from an environmental, health, and, not least, policy perspective, it may be of interest to adapt dietary scenarios to specific groups of a population to a higher degree than at present.

Dietary scenario studies are in general designed to study the impact of dietary change in European countries/regions, characterized by having an unrestricted diet. In many parts of the world trends of rapid changes in diet have had negative (and positive) implications from both health and environmental perspectives. This trend is predicted to continue as the living conditions of more people improve. To understand the impact of dietary change in a broader perspective, and not only to study how to implement change but how to prevent unhealthy and resource-intensive food habits, dietary scenario studies are increasingly required, also in developing and transitional regions with different habits, cultures and conditions.

8.4 Differentiation of land use and land use impacts

When discussing the availability of agricultural land it is necessary to distinguish between different types of land, as this will affect the possibilities and consequences of how the land can be used and how its use can be optimized for food production.

Pressure on global agricultural land is especially intense on cropland, on which the majority of global food supply is produced (Johansson, 2005). However, as discussed in **paper IV**, dietary change has, above all, the capacity to release agricultural land currently used for pasture. Since not all pasture land is suitable for cultivation, a distinction between different types of land is required for an improved understanding of the potential to reduce the demand of land use by dietary change. The review of dietary scenario analyses in **Paper IV** identified a lack of differentiation of land use in the literature. To avoid a situation in which demand for agricultural land is exported to other countries and regions where it might lead to deforestation and other negative impacts associated with increased pressure on land use, it may also be of use to distinguish between domestically produced food and imported food in future dietary scenario analysis.

The use to which a piece of land is best suited is affected by a variety of factors, several of which go beyond geographical and ecological considerations. Two important questions in this context are: who owns the land? and what is its current use? Ecological and social consequences of land use change may differ depending on whether a top-down or bottom-up perspective is applied. Today much of the research is done with a top-down perspective by which the use and availability of land is analyzed from a national, regional or even global perspective. However, to answer the two previous questions a change of focus is required from a top-down to a bottom- up perspective, which would account for local effects at the place where the land use change is actually taking place.

How the livelihood of the people in developing regions is affected by land use changes is of special concern, as it is in these regions (e.g. Africa) the largest potential for increased intensity in agriculture and expanding land under cultivation is estimated to be. Poor people in these areas also often rely more heavily on local ecological resources and surrounding ecosystem services, which makes them particularly vulnerable to food and energy insecurity as a result of changes in land use.

As a top-down perspective may lead to a loss of information about the effects at the local level, while a bottom-up perspective may be insufficient to understand the dimensions of a problem at a higher level, research from both perspectives is required, as well as research in which land availability and conflicts are analyzed from an integrated top-down and bottom-up perspective.

8.5 Accounting for uncertainty

Despite the knowledge of the uncertainty related to LCA data for food and dietary scenario analysis, the review in **paper IV** shows that most articles report the environmental impact of dietary scenarios in precise numbers without any uncertainty intervals or sensitivity assessment, making it difficult to determine the reliability of the results.

According to the ISO standard, the interpretation phase in LCA should include an evaluation of the completeness, sensitivity and compliance of the analysis (ISO, 2006b). This would help the reader to determine which conclusions can be drawn from the results, and is critical also in dietary scenario analysis.

To improve the knowledge of how to reduce the environmental impact of food consumption and production, research studies in the future should, to a greater extent, include reports of the uncertainty when analyzing, evaluating and reporting methods and results.

8.6 Sustainable nutrition – how to get there?

From public health-based interventions it is well known that the implementation of changes in diet is difficult, and that despite knowing what a healthier diet implies people often find it difficult to change their eating habits (Davies, 2011). Besides increased knowledge of how consumption (and production) of food can be improved, a better understanding of how to implement these changes in society is required in order to increase the sustainability of the food system.

Policies for influencing the sustainability of food systems can be categorized into information-based (e.g. product labels, marketing, education, campaigns), market-based (e.g. subsidies, taxes, fees) and regulatory instruments (e.g. laws, policies, certifications, public procurement) (Reisch, 2011). Information-based instruments are the cheapest and most widely used measures to influence the eating habits of the general public. Several countries have also developed national polices or guidelines for sustainable food consumption. However, to meet environmental goals by raising awareness is unlikely alone to be sufficient. It is more likely that a combination of several different interventions will be required (Garnett et al., 2015; Reisch, 2011).

Besides research on how diets can be improved in populations which have already undergone the nutrition transition towards "western diets", and health problems linked to over-consumption of food, more studies are needed to analyze how a sustainable nutrition transition can be promoted in those populations which have not undergone this transition. The challenge is to bring about changes that reduce the health problems related to undernutrition without introducing new problems linked to the over-consumption of food.

Which policy instruments that are most effective from a holistic perspective, for a development towards more sustainable food patterns, and how these can be successfully implemented in different regions and populations, is an area of research that requires further exploration.

8.7 The holistic perspective

Figure 31 provides an overview of the system studied in this thesis, and the interrelation between some components in the system identified to be of significant importance. The components identified are categorized as either challenges (in red), measured indicators (in purple), drivers (in yellow) or potential opportunities (in green).

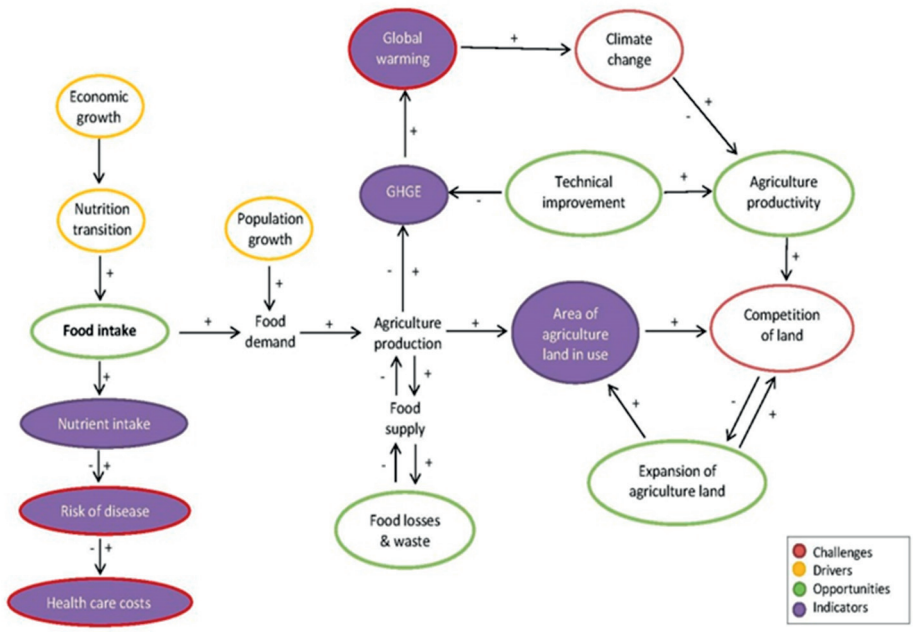

Figure 31. Schematic illustration of the system studied in this thesis

Figure 31 illustrates how the complexity of the system increases as more perspectives are added. The different components of the food system are interrelated and influence each other through a variety of both known and unknown mechanisms. Changes in the diet can thereby have consequences in different parts of the system. Many of these consequences are not covered in this thesis, which is why future studies must be further complemented and refined. Thus, a more complete understanding of how sustainability can be improved in the food system requires more holistic assessments including additional aspects.

9 Concluding remarks

The world is facing of a number of great challenges as the population is growing and living habits are getting more resource demanding. To provide enough food in a sustainable way is a dilemma today and not least for the future. The current food system is not sustainable, it is a key driver of environmental pressures and is linked to health problems both related to inadequate and excessive consumption of food.

This thesis shows that there are good opportunities to improve the sustainability of current food systems. Dietary change is one of many suggested measures for reducing global GHG emissions and land use pressure. However, in contrast to most other measures suggested, dietary change can bring the co-benefit of improving dietary quality, which is necessary to curb the global epidemic of diet-related, chronic diseases. The positive synergies suggest dietary change to be an effective measure for tackling several major challenges simultaneously.

Assessments of environmental- and health effects of food consumption and production are hampered by uncertainty and variability, and therefore awareness of the limitations in the quality of data and methods is crucial. Transparent presentation of data and methods is necessary to enable a proper interpretation of the reliability and significance of the results. Integration of nutrition and health aspects into environmental assessments of food is an exciting development of this research field contributing to important new knowledge. To further broaden the perspectives and deepen the knowledge of sustainable food systems more aspects need to be covered.

There is an urgent need to reduce environmental pressures from the food sector and to improve global health. Although there are still gaps in our knowledge, the advances in research have significantly improved the understanding of sustainable food systems. This thesis shows that dietary change can play an important role in reaching future environmental and health goals. To achieve a positive development towards more sustainable food systems, and thereby avoid the risks associated with current trends, now is the time for knowledge to be implemented in practice and transformed into action.

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Paper I

Sustainability: Science, Practice, & Policy
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ARTICLE

Meat-consumption statistics: reliability and discrepancy

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Interest in meat consumption and its impact on the environment and health has grown markedly over the last few decades and this upsurge has led to greater demand for reliable data. This article aims to describe methods for producing meat-consumption statistics and discuss their limitations and strengths; to identify uncertainties in statistics and to estimate their individual impact; to outline how relevant data are produced and presented at the national (Swedish), regional (Eurostat), and international (FAOSTAT) levels; to analyze the consequences of identified discrepancies and uncertainties for estimating the environmental and health effects of meat consumption; and to suggest recommendations for improved production, presentation, and use of meat-consumption statistics. We demonstrate many inconsistencies in how meat-consumption data are produced and presented. Of special importance are assumptions on bone weight, food losses and waste, weight losses during cooking, and nonmeat ingredients. Depending on the methods employed to handle these ambiguous factors, per capita meat-consumption levels may differ by a factor of two or more. This finding illustrates that knowledge concerning limitations, uncertainties, and discrepancies in data is essential for a correct understanding, interpretation, and use of meat-consumption statistics in, for instance, dietary recommendations related to health and environmental issues.

KEYWORDS: meat production, food consumption, statistical analysis, environmental effects, public health

Introduction

Quality national and international data are essential for understanding social dynamics that are often the foundation for scientific research and policy development. Recent decades have given rise to growing interest in meat consumption and its effects on the environment and health, leading to a greater demand for reliable meat-consumption data. Such statistics are used in research to assess present and historical nutrient intake and environmental impacts, but also to predict future trends. Data on meat consumption are also used to develop guidelines, policy programs, and strategic interventions regarding health, climate change, and land-use issues.

Methodologies for producing consumption statistics suffer from a number of limitations and uncertainties that affect the overall reliability of the data. Lack of harmonization of definitions and regulations concerning how data are obtained and presented further complicates the combination and comparison of data from different countries and regions (e.g., EU by Eurostat) and globally (e.g., by FAOSTAT). Examples of factors that influence meat-consumption data are whether bones are included in weight calculations, waste is accounted for at different stages along the food chain, weight refers to raw or cooked meat, and whether ingredients of nonmeat origin in mixed processed-meat products and ready meals are included. Awareness of inclusions and exclusions in the data, and of its limitations and uncertainties, is essential for correct understanding, interpretation, and use of such statistics. This article seeks:

- To describe the methods for producing meat-consumption statistics and discuss their limitations and strengths.
- To identify uncertainties in statistics and estimate their individual impact.
- To outline and compare how meat-consumption statistics are produced and presented at the national (Swedish), regional (Eurostat), and international (FAOSTAT) levels.
- To analyze consequences of identified uncertainties and discrepancies for assessments investigating the environmental and health impacts of meat consumption.
- To suggest improvements in the methodology for producing, presenting, and handling meat-consumption statistics.

Methodology and Assessment Approach

This study relies on data and other information from scientific articles, statistical reports, online databases, and personal communication with authorities in the field. We analyzed, processed, and categorized information and data as outlined below to formulate relevant comparisons and conclusions.

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The categorization is based on the following:

- Type of survey methods (see Methods for Producing Meat-Consumption Data): whether data are based on agricultural supply, household-budget surveys (HBSs), or individual dietary surveys (IDSs). This section describes ways of producing meat-consumption data, limitations and strengths of existing methods, and appropriate usage of data produced with different methods.
- Type of meat data (see Uncertainty Factors in Meat-Consumption Data): whether meat-consumption data i) refer to carcass weight or bone-free weight, ii) are adjusted for food losses and waste, iii) refer to raw or cooked meat, and iv) include or exclude nonmeat components in mixed-meat products and prepared meals. This section describes factors contributing to discrepancies in meat-consumption data, explains how different types of survey methods deal with this variability, and estimates their individual impact.
- Type of statistical sources (see Meat-Consumption Statistics on National, Regional, and International Levels): whether meat-consumption data are provided at the national (Sweden), regional (Eurostat), or international (FAOSTAT) level. This section describes how meat-consumption data are produced and presented, discrepancies among statistics at the different levels, and factors affecting accuracy and reliability.

Results from previous research are used to discuss and illustrate the consequences of variability in data for assessments investigating the environmental and health impacts of meat consumption.

Results

Methods for Producing Meat-Consumption Data

There are several methods of producing data on meat consumption. The specific method should reflect the purpose for which the data will be used and will influence how they should be interpreted. Data on food consumption can be derived from agricultural supply, HBSs, or IDSs (Naska et al. 2009; SFA, 2011a). Table 1 provides a summary of methods used for generating meat-consumption data and the factors that determine their correct use and interpretation.

Food-Consumption Data Based on Agricultural Supply

Per capita consumption data are generally based on agricultural and trade information and provide insight into the average quantity of the commodity in question available for use within a country or region (FAO, 2001; SFA, 2011a). Food-balance sheets (FBSs) at regional (e.g., Eurostat) and global (e.g., FAOSTAT) levels provide standardized supply data and represent an important knowledge base that permits comparative analyses over time.

In agricultural statistics, meat refers to the flesh of animals used for human food and hence excludes meat unfit for human consumption (EC, 2009; FAO, 2011a). The available supply of meat in a country is typically calculated as (national production + import + opening stocks) – (exports + usage input for food[1] + feed + nonfood usage + wastage + closing stocks). Per capita supply data are obtained by dividing the national available supply by the number of inhabitants (FAO, 2001; EC, 2011a). Although the data in agricultural statistics only provide information on the available per capita supply of meat, these data are often used, due to economic constraints and lack of other data, as a proxy for per capita meat consumption.

Agricultural supply data can either be presented as the available supply of raw material per person (i.e., cereals, milk, sugar), or as the available supply of food per person (i.e., bread, cheese, candy) (Eidstedt & Wikberger, 2011; SFA, 2011a). Depending on how data are presented, adjustments for food losses (beginning of the food chain) and waste (end of the food chain) may or may not be accounted for. Factors affecting the reliability of agricultural supply data include the risk of incomplete and/or inaccurate underlying national statistics (e.g., in certain developing countries), limited information on losses and waste along the food chain, and incomplete reporting of noncommercial products (e.g., game) (FAO, 2001; Hawkesworth et al. 2010).

Agricultural supply data makes it possible to study consumption trends over time and to compare consumption across different regions and countries. The data are useful for evaluating a country's agricultural situation (and thus to projecting future demand and supply of food), setting targets for agricultural production and trade, and evaluating national food and nutrition policies (FAO, 2001). As the data are based on the available supply per person, they are not completely accurate in describing what people actually eat (SFA, 2011a). The available supply of food thus represents only the quantities reaching the consumer (after losses and waste during harvest, storing, processing, distribution, and retail) and it does not take into account household wastage during storage, preparation, and cooking. Furthermore, agricultural supply data provide no specific insights

[1] Usage input for food refers to the amount of originating meat required for obtaining an output of a derived meat product.

Table 1 Appropriate use of meat-consumption statistics and factors of importance for the correct use and interpretation of data produced by different methods.[a]

Method	Appropriate Use	Important Factors for Correct Use and Interpretation
Data based on agricultural supply (e.g., FBSs)	• For description of the average quantity of meat available for use within a country. • For studying consumption trends over time and for comparing consumption in different countries and world regions. • Less accurate for describing what people actually eat and consumption characteristics in different national populations groups and regions.	• Is consumption of noncommercial meat accounted for? • Have the data been recently updated? • Are food losses and waste accounted for? • How is meat content in processed products and prepared meals reported?
Household Budget Surveys (HBSs)	• For comparison of consumption between different regions and socio-economic groups. • For monitoring changes in consumption patterns over time. • More appropriate for studying food intake in a population than for individuals.	• Is the selection of participants representative of the population studied? • How good/bad is the participation rate? • Is there a risk of under-, over- or mis-reporting? • Has the method been internally or externally validated? • Are the food categories used comparable? • Do the data account for meat consumed outside the household? • Is food waste in the household accounted for?
Individual Dietary Surveys (IDSs)	• For description of individual consumption. • Provides information about the amount of meat actually eaten. • For mapping dietary habits, studying the relationship between diet and health, and quantifying determinants and consequences of food choices.	• What survey method has been used to obtain the data? • Is the selection of participants representative of the population studied? • How good/bad was the participation rate? • Is there a risk of under-, over- or mis-reporting? • Has the method been internally or externally validated? • Has food waste in the household been accounted for? • Does the consumption refer to raw or cooked weight?

[a] For references, see Methods for Producing Meat-Consumption Data

about consumption characteristics in different populations, regions, socioeconomic groups, or among individuals in households (FAO, 2001). Despite these limitations, agricultural supply data are often used to describe food consumption.

There is currently no international regulatory framework for how statistics on agricultural supply should be produced (Eidstedt, 2011), although global recommendations and guidelines exist for appropriate approaches to obtain and present data (FAO, 2001; De Henauw et al. 2002; EC, 2011b; European Statistical System, 2011). This means that national data on agricultural supply from different time periods may not be comparable if methods used to produce the data changed over time and that accurate comparisons across different countries may be difficult (Serra-Majem et al. 2003; Eidstedt, 2011; SFA, 2011a).

Food-Consumption Data Based on Household Budget Surveys

HBSs are generally conducted by national statistics offices and provide information on how much money is spent on different foods per household and sometimes also on the quantity of food purchased per household (Naska et al. 2009; SFA, 2011a). The data can either be obtained from trade-sales figures or from self-reported household expenditures.

Statistics based on these surveys are useful for comparing expenditures on different foods and consumption across different regions, populations, and socioeconomic groups. These data generally provide no information about what happens to food after purchase (i.e., whether the food is eaten or not, or how consumption is allocated among individuals in the household) (Hawkesworth et al. 2010; SFA, 2011a). Consumption data based on HBSs are therefore more appropriate for studying food intake in a population than in individuals (Naiken, 2003; Serra-Majem et al. 2003). Furthermore, data based on HBSs are often expressed as food categories rather than individual foods, which may cause difficulties due to lack of harmonization of categories in different surveys (Serra-Majem et al. 2003).

Like other self-reporting methods, HBSs are challenged by various uncertainties, such as recall and reporting errors. To study food consumption using HBSs, data should ideally be collected both on

food consumed in the household and away from home. If household expenditure is used to estimate food intake, there is a risk that food eaten outside the home will be excluded. Other reliability issues arising from the use of these surveys include the difficulty of accounting for food consumed by guests in the household and of adjusting for food that is purchased and stored without being consumed during a recall period (as well as the reverse, if food is consumed that was purchased prior to the recall period) (Smith, 2003; Hawkesworth et al. 2010). The representativeness of data based on HBSs further depends on the participation rate and whether the sampling consists of a uniform distribution between, say, urban and rural areas, poorer and wealthier households, and single and multi-individual households (Hawkesworth et al. 2010). The reliability of HBSs can be increased by covering longer recall periods and by conducting multiple rounds, as well as by collecting complementary information about food habits in the household (Smith, 2003).

Food-Consumption Data Based on Individual Dietary Surveys

IDSs provide data on the amount of food actually eaten by individuals and groups and are one of the most accurate (and costly) methods for obtaining data on food consumption (Naska et al. 2009). These data are typically used to map dietary habits, to study the relationship between diet and health, and to quantify determinants and consequences of food choices (Naska et al. 2009; SFA, 2011a).

There are several methods for studying eating habits. The most common approaches are 24-hour recall, dietary history interviews (DHIs), food-frequency questionnaires (FFQs) (retrospective methods), and dietary records (prospective methods). Twenty-four hour recall and DHIs entail interviewing participants about the amount and type of food previously eaten. In a 24-hour recall, only food eaten during the past day is reported, while DHIs typically are used to inventory food consumption over a longer period. Reported intake that deviates from the person's average consumption is presumed to average out across the whole sample. FFQs are based on instruments in which participants report information on the quantity and type of food eaten. Dietary record methodology can vary, but the basis is that the participants report in writing all food eaten during a specified period. The amount of food consumed can either be estimated or weighed and the survey period can vary. The reported intake in dietary records is assumed to be representative of the person's average food consumption.

Statistics based on IDSs offer the possibility of studying food consumption on an individual level and of matching consumption patterns with specific characteristics such as gender, age, employment, and cultural and ethnic background. Unlike data based on agricultural supply and HBSs, IDSs usually refer to food intake after adjustment for household waste and may also provide insight on methods of preparation.

It is well known that existing methods used to assess dietary intake on an individual level are hampered by various limitations and inherent errors that affect reliability. Sources of errors can be divided into random and systematic errors. Random errors refer to problems such as day-to-day variability of food intake and seasonal variations (i.e., that reported food intake in dietary surveys is not representative of average consumption). The risk of random errors can be reduced by increasing the number of surveyed days and subjects. Systematic errors, such as problems with under-, over-, and misreporting, are common in retrospective methods that require a good memory and sincerity (Ferro-Luzzi, 2003; SFA, 2011b). To overcome uncertainties in data and to enhance the quality of data on food consumption based on IDSs, internal or external validation of the method used is recommended (Ferro-Luzzi, 2003).[2] A more detailed summary of recommendations for the improvement of the quality of IDSs can be found in De Henauw et al. (2002).

Uncertainty Factors in Meat-Consumption Data

The previous description of existing methods for producing meat-consumption data makes clear the various factors that may affect reliability and accuracy. To be aware of the factors that contribute to uncertainty and to know their individual impact facilitates accurate interpretation and use of the resultant data. Based on the previous description of methods and the summary in Table 1, we have identified four main uncertainty factors affecting the accuracy and reliability of meat-consumption statistics. Table 2 provides an overview of these issues and their estimated impacts.

Weight of Bones

Agricultural statistics on meat consumption and production are generally presented as carcass weight or as bone-free carcass weight. The carcass weight typically refers to the total weight of the slaughtered animal's body after removal of inedible body parts (e.g., skin, offal, slaughter fats, head, feet, tail, and genital organs) and body parts used for nonfood pur-

[2] Examples of validation methods are comparing data against results produced by another method, using biomarkers, or relating energy-intake levels with measurements/estimations of energy expenditure.

Table 2 Uncertainty factors in meat-consumption data and their individual impact.[a]

Uncertainty Factor	Estimated Uncertainty Interval (%)
Weight of bones	23-41[b]
Food losses and waste	15-21[c]
Weight loss during cooking	20-50[d]
Meat content in processed meat	41-53[e]

[a] For references, see Uncertainty Factors in Meat-Consumption Data.
[b] Average bone weight of the carcass for beef, pork, and chicken.
[c] Post-farm food losses and waste throughout the entire production, depending on region.
[d] Weight loss during cooking of raw meat.
[e] Based on a Swedish study (SBA, 1998) concerning the meat content of mixed charcuterie products and prepared meals.

poses (e.g., feed, soap, and cosmetics) (EC, 2009; FAO, 2011a). The definition of carcass weight is not homogeneous and parts of the animal included in the definition vary depending on the country and type of livestock (FAO, 2011a). According to previous research, bone-free meat represents on average 70% (Cederberg et al. 2009), 59 %, and 77% (Sonesson et al. 2010) of the carcass weight in beef, pork, and chicken, respectively. Statistics based on HBSs and IDSs refer to the amount of meat purchased at retail (or in restaurants) and/or the amount actually eaten, and thus are indicative of bone-free meat. Statistics normally report meat products found at retail establishments that contain smaller pieces of bones (e.g., chicken wings) as bone-free meat (Eidstedt, 2011).

Food Losses and Waste

Food losses and waste can occur at virtually any stage along the food chain. At the global level, it is estimated that between one third and one half of all food produced is spoiled before or after it reaches the consumer (Lundqvist et al. 2008; FAO, 2011b). The magnitude of these estimates illustrates the importance of accounting for food spoilage in consumption statistics. Where in the supply chain food losses occur and how large proportions are lost vary by commodity as well as by country and region. For example, in developed countries food waste at retail and consumer levels accounts for a sizable fraction of food losses, whereas losses in the early stages of the supply chain (e.g., storage and distribution) are more common in developing countries (FAO, 2011b).

Various sources, including enterprises and manufacturing surveys, provide information on losses occurring along the food chain. Food losses refer to "the decrease in edible food mass along the supply chain leading to edible food for human consumption," and thus exclude meat intended for nonfood uses (e.g., feed and industrial uses) and inedible parts (FAO, 2011b). Products originally intended for human consumption that end up being used for a nonfood purpose may, however, be categorized as food loss. Food waste refers to losses occurring at retail or consumer levels. The estimated losses and waste of meat after agricultural production (e.g., postharvest handling, storage, processing packaging, distribution, retail, and consumption) ranges between 15–21% of the total production, depending on global region (FAO, 2011b).

Postfarm losses and technical losses in processing up to the retail stage may or may not be accounted for in meat-consumption data based on agricultural supply, depending on the country concerned. In addition, household-food waste is not taken into account in agricultural supply data, such as that provided by FAOSTAT. Furthermore, in consumption statistics based on HBSs and IDSs, waste at the household level is in general not accounted for. However, this depends on the specific design of the method used to obtain the data. Food waste in the household is estimated to account for about 25% of all food purchased (by weight) in the UK (WRAP, 2009), for 8–11% of the meat purchased in industrialized regions, and for 2–6% in developing regions (FAO, 2011b).

Raw or Cooked Weight

Data on meat consumption can either be presented as raw weight or weight after cooking. One kilogram (kg) of raw meat is roughly equivalent to 700 grams (g) of cooked meat. However, the conversion factor may vary between 0.5 and 0.8 depending on, for example, the cut of meat, proportions of lean to fat, as well as method and extent of cooking (KF & ICA Provkök, 2000; WCRF/AICR, 2007). The weight difference is due to the water content, which partially evaporates during cooking. The water content in beef, pork, and chicken is approximately 58–73%, 65–75%, and 53–75%, respectively (Amcoff, 2011).

Meat-consumption statistics based on agricultural data and HBSs in general refer to the raw weight of meat, whereas statistics based on IDSs, as well as nutritional recommendations, can be reported either as raw or cooked weight, depending on the design of the method.

Mixed-Meat Products and Prepared Meals

Meat is often eaten in the form of mixed-meat products, such as sausages and other mixed-charcuterie products and prepared meals. A Swedish Board of Agriculture (SBA) (1998) study examined

Table 3 Methods and assumptions used to produce meat-consumption statistics in Sweden, Eurostat, and FAOSTAT.[a]

	Weight of Bones	Food Losses and Waste	Raw or Cooked Weight	Mixed-Meat Products and Prepared Meals
Swedish Agricultural Statistics	• Total meat consumption, including bone weight. • Direct meat consumption, excluding bone weight in beef and pork (25% and 15.2% of carcass weight), including bone weight in poultry.	• Total meat consumption: no deduction for losses/waste between slaughter and consumption. • Direct meat consumption: assumed losses/waste between slaughter and retail corresponds to 5% of bone-free carcass weight.	Raw weight	• Total meat consumption: exclude weight for nonmeat content of processed products. • Direct consumption: total weight of the processed product.
Eurostat	• Consumption refers to carcass weight, i.e., including bones. • No information found on assumed bone weight in relation to carcass weight.	• Losses/waste between slaughter and retail is adjusted for. • No information found on assumptions for food losses and waste.	Raw weight	• Mixed products made up of meat from several species are included in the "other meat" balance.
FAOSTAT	• Consumption refers to carcass weight, i.e., including bones, unless otherwise stated. • No information found on assumed bone weight in relation to carcass weight	• Meat-supply data are adjusted for food manufacture and losses/waste up to the stage of retail. • No information found on assumptions for food losses and waste.	Raw weight	• Food-supply data include both primary commodities and processed-food products.

[a] For references, see Meat-Consumption Statistics on National, Regional, and International Levels

the meat content in processed meat products on the Swedish market and showed that the average meat content in mixed-charcuterie products and prepared meals was 53% and 41%, respectively. Despite the fact that the meat content is considerably less than 100% in these products, the total weight of such products is commonly reported in meat-consumption statistics (Eidstedt, 2011).

In statistics based on agricultural supply, only the meat content in mixed-meat products and prepared meals is generally included in figures for the total consumption of different types of meat. By contrast, in the cases where meat consumption is reported as products of a higher degree of processing (e.g., "direct consumption" in Swedish statistics), consumption often refers to the total weight of the mixed products and prepared meals. Furthermore, in consumption statistics based on IDSs, mixed-meat products and prepared meals in general refer to the total weight of such products. If meat-consumption statistics are based on data that do not distinguish between meat and nonmeat components in mixed-meat products and prepared meals, consumption risks being overestimated (Riley & Buttriss, 2011).

Meat-Consumption Statistics on National, Regional, and International Levels

Statistical institutes at national, regional, and international levels publish per capita meat-consumption statistics. Consumption statistics at different levels are produced by various methods based on nonstandardized assumptions and thus vary in reliability and accuracy. Being aware of the procedures used to collect data, the assumptions on which they are based, and the discrepancies between meat-consumption data at different levels will improve the prospects for correct understanding, interpretation, and use of this information. The following sections describe how per capita meat-consumption statistics from Sweden, Eurostat, and FAOSTAT are produced and presented. The statistics are based on agricultural data and thus refer to the available supply for human consumption, i.e., excluding meat for nonfood purposes. Table 3 provides a summary of methods and assumptions used to produce meat-consumption statistics at national, regional, and global levels.

Swedish Statistics on Meat Consumption

Sweden has three different sources of meat-consumption statistics: agricultural data provided by SBA, HBSs developed by Statistics Sweden, and dietary surveys carried out by the Swedish Food Agency (SFA, 2011a). This section describes meat-consumption statistics distributed by SBA, which are used to calculate Swedish per capita meat consumption.

Swedish meat-consumption statistics are either presented as "total meat consumption" or "direct meat consumption." "Total meat consumption" refers to the overall supply of raw meat (including bones) available for human consumption at the farm gate

(plus estimated quantities of noncommercial meat), after correction for imports and exports (Eidstedt & Wikberger, 2011). In mixed products, such as charcuterie products and prepared meals, "total meat consumption" refers to meat excluding the weight of nonmeat ingredients (Eidstedt, 2011).

"Direct meat consumption" refers to the total supply of food available for private households and catering (Eidstedt & Wikberger, 2011). The data exclude the weight of bones in beef and pork (25% and 15.2% of carcass weight, respectively) but includes bone weight in poultry. Losses due to inputs in processing and losses up to the stage of retail are adjusted for, and assumed to correspond to, 5% of the bone-free carcass weight. "Direct meat consumption" statistics present data on the consumption of charcuterie products and frozen prepared meals as separate categories. For charcuterie products and frozen prepared meals containing meat, data refer to the total product weight (i.e., including nonmeat contents) (Eidstedt, 2011).

Swedish per capita consumption statistics are based on several sources, such as data from agriculture and trade statistics, SBA statistics, Swedish Statistics, Swedish Food Agency, and so forth and production information is obtained from specific commercial producers. The reliability of Swedish-consumption statistics largely depends on the quality of the underlying data. National consumption statistics are presented annually and give statistics referring to data that are about one-and-a-half years old. Continuous changes in methodology and classification may mean that statistics from year to year are not fully comparable, and therefore consumption changes seen over short time periods should be interpreted with caution (Eidstedt & Wikberger, 2011). Other factors affecting the accuracy and reliability of national agricultural statistics have previously been described in Methods for Producing Meat-Consumption Data.

European Statistics on Meat Consumption

The European Statistical System (ESS) provides European statistics on meat consumption in different countries, which are freely available via the online database Eurostat.[3] ESS is based on a partnership between the Commission (Eurostat), the national statistical institutes (NSIs), and the national authorities in individual member states. The main role of ESS is to harmonize statistics to provide comparable information at the European level (EC, 2011d). Eurostat presents statistics on consumption of commodities in FBSs as the gross human apparent consumption, which is "a proxy indicator for the availability of a commodity to the consumer." Human consumption is defined as "the quantity of products placed at the disposal of human consumption in all forms: quantities consumed without modification and processed quantities," and is quantified as "the balance between production, imports and exports, stock changes, and by its uses as food, waste, food manufacture, and others" (EC, 2011a).

Meat-consumption data in Eurostat are expressed in carcass weight (i.e., in raw weight excluding offal and hide but including weight for bones). Information on which body parts are included in the carcass weight of different animals is described in the manual for compilation of supply-balance sheets in Eurostat (EC, 2009), according to which food-supply data are adjusted for losses during stocking, transport, processing, and packing (i.e., up to the stage of retailing). No information has, however, been found on assumptions regarding the proportions assumed for bone weight in relation to carcass weight or on the magnitude of losses and wastage at different stages along the food-supply chain. Mixed products made from meat of several species (e.g., sausages) are included in the "other meat" balance. Further information on how consumption statistics for mixed-meat products are presented could not be found.

Statistics presented in Eurostat are mainly based on data collected by statistical authorities in individual member states, but also on data from unpublished national contributions, subsets of national contributions, and specifically designed European statistical surveys (ESS, 2011). Factors affecting the accuracy and reliability of national agricultural statistics have previously been described. As consumption statistics from different countries are based on various methods and assumptions, they may not be consistent and are thus not appropriate for direct comparison. After collection, data must therefore be harmonized to provide comparable statistics at the European level (EC, 2011d). To increase the quality, comparability, and reliability of European statistics, data should be produced, developed, and disseminated according to the uniform standards in the European Statistics Code of Practice (ESS, 2011).

Global Statistics on Meat Consumption

Global statistics on meat consumption in different countries and regions are provided on a yearly basis by the Food and Agriculture Organization of the United Nations (FAO) and are freely available via the online database FAOSTAT.[4] FAO's FBSs provide statistics on domestic supply quantities, defined as "the total quantity of the foodstuff produced in a

[3] See http://epp.eurostat.ec.europa.eu/portal/page/portal/eurostat/home.

[4] See http://faostat.fao.org.

country added to the total quantity imported and adjusted to any change in stocks (from production to retail and all actors holding a stock of a meat-based commodity) that may have occurred since the beginning of the reference period." In addition, the domestic use of each commodity is presented, with a distinction made between quantities fed to livestock (feed), used for seed (seed), processed for food and nonfood uses (processed), lost during storage and transportation (waste), available for human consumption at the retail level (food), and other use (other utilization). Food-supply data, expressed in metric tons or as kilograms per capita per year (kg/capita/year), are provided in FAOSTAT both under the category of "food supply" and in the FBSs (food-supply quantity, food). The food-supply data refer to the domestic supply quantity after a deduction for feed, seed, food manufacture, and waste (FAO, 2001; Jacobs & Sumner, 2002; FAO, 2012a).

In FAO's FBSs, the supply of meat is expressed as carcass weight (i.e., weight including bones, unless otherwise stated, excluding pieces unfit for human consumption as well as inedible offal and unused fats) (FAO, 2011a; Westhoek et al. 2011). Food-supply data are adjusted for losses and waste at all stages between the level of production and household (i.e., during storage, transportation, processing, and retail). Data on average carcass weight in relation to live weight and on waste of supply of crops and derived products are presented in FAO's publication on technical conversion factors for agricultural commodities (FAO, 2012b). However, no information has been found regarding the assumed proportions of bone weight in relation to carcass weight or assumed magnitude of losses and wastage of meat supply. FAO food-supply data include both the supply of primary commodities and processed foods derived therefrom, expressed in amounts of the original farm commodity. The amount needed to produce a processed food product is quantified based on technical conversion factors provided per commodity and country (FAO, 2012b).

Underlying data in the FBSs are based on a wide variety of sources of varying quality, including both official and unofficial documentation such as national trade and agriculture statistics, sample surveys, questionnaires, censuses, administrative records, and best estimates. Missing data are often estimated on the basis of surveys as well as technical expertise available at FAO (2001; 2012a).

The accuracy of data in FAO's FBSs to a large extent depends on the quality of the underlying data in official national statistics. Factors affecting accuracy and reliability of national agricultural statistics have previously been described. As supply data from different countries may be produced via different methods and based on different assumptions, they may not be appropriate for direct comparison. To allow for international comparisons, data are adjusted by FAO before being disseminated (FAO, 2001; 2012a; Jacobs & Sumner, 2002).

Discussion

The purpose of this article is to identify uncertainties and discrepancies in meat-consumption statistics and to discuss their potential impact on assessments of environmental and health effects of dietary patterns. The results show various uncertainty factors in how these data are produced and highlight issues that encourage more subtle understanding and interpretation. We also find the transparency of information pertaining to methods and assumptions in the generation of food-consumption statistics deficient on Swedish, European (Eurostat), and international (FAOSTAT) levels.

The importance of accounting for uncertainties in consumption statistics has previously been noted and discussed. Serra-Majem et al. (2003), for example, identified significant differences in consumption data produced by different methods. One example was that the quantity of food consumed, indicated by data based on HBSs, in general is lower than that from FBSs, and that FBS data often overestimate food and nutrition intake compared to data based on IDSs. Other studies, which confirm that quantities indicated by data based on HBSs are typically lower compared to those predicated on FBSs, suggest that the differences are at least 20% (Sekula, 1993; Serra-Majem et al. 1993; Naska et al. 2009). Naska et al. (2009) compared food-consumption statistics from FBSs and HBSs in eighteen countries and demonstrated that the correlations between data derived from FSBs and HBSs are quite strong for vegetables, fruit, fish, and oil (+0.69 – +0.93), whereas the correlation is lower for meat and meat products (+ 0.39).

To interpret consumption statistics correctly, one must remember that per capita data refer to the average intake level in a population. Per capita consumption data thereby conceal considerable inherent variation among different groups, such as between men and women, adults and children, and across socioeconomic groups (especially in developing countries). To draw general conclusions and formulate recommendations within a population based on per capita figures can thus lead to certain errors of interpretation. For example, it is well known that men generally consume more meat than women (Beardsworth et al. 2002; Kubberød et al. 2002; Prättälä et al. 2007). Results from Swedish nationwide nutrition surveys have shown that average meat intake among men is 35% higher than among women (Becker &

Pearson, 1998). Differences in meat intake, as well as in nutrient requirements among population groups, has implications for the quantity of meat that can be considered healthy and should thus be taken into account when designing recommendations for meat consumption.

An incomplete understanding of meat-consumption data entails the risk that the statistics will not be used appropriately, which could have widespread implications for research findings and recommendations. One direct effect of the several existing definitions of meat consumption, which are based on varying methodologies and assumptions, is that meat-consumption data vary depending on the statistical source. For example, the official figure for per capita meat consumption in Sweden is 83 kg per year (data for 2009) (Eidstedt & Wikberger, 2011). However, estimates based on IDSs show that annual per person meat consumption in the country is only 66 kg (Lagerberg-Fogelberg, 2008) and according to FAO's FBSs the figure is 75 kg (data for 2009) (FAO, 2012c). The calculation of nutrient intake and environmental impact from Swedish meat consumption will thus be heavily dependent on which data set is employed. Furthermore, meat-consumption statistics do not include information on production systems, which will have a significant impact on the environmental performance of various types of meat (i.e., from grass-fed cattle or indoor grain-fed cattle) (De Vries & De Boer, 2010).

When using meat-consumption statistics, it is thus important to know what the data actually represent, specifically whether i) consumption refers to agricultural supply, purchased amount, or actual intake, ii) consumption refers to bone-free weight, iii) food losses and waste are accounted for, iv) consumption refers to raw or cooked weight, and v) weight of nonmeat ingredients in mixed-meat products are accounted for. Methodological descriptions providing this information are, however, often difficult to find and to interpret. An accessible and clear presentation of meat-consumption data, which outlines the procedures used to generate the information and documents the underlying assumptions, would facilitate appropriate usage and interpretation.

The factors that contribute to discrepancies in meat-consumption data may individually affect the data by 15–50%. A simple quantitative example illustrates how these factors can influence meat-consumption data. As previously mentioned, annual per capita meat consumption in Sweden was 75 kg in 2009 according to FAO supply data. However, the actual intake of meat, after adjustment for bone weight, household-food losses and waste, and weight reduction in cooking, is markedly lower. If the intake were adjusted for bone weight, annual per capita meat consumption in the country would be reduced to approximately 53 kg (assuming that bone-free weight represents 70% of carcass weight). If adjusted for household-food spoilage and waste (assumed to represent 11% of purchased weight) (FAO, 2011b) and weight reduction by cooking (assumed to be 30% of raw weight), the actual annual per capita intake of cooked, bone-free meat in Sweden would be approximately 33 kg.

On one hand, in environmental assessments, meat-supply data, expressed as raw meat including bones, are often used as the basis for calculations. When studying health effects or nutritional intake, on the other hand, data on actual consumption, expressed as uncooked or cooked meat, are generally employed. The example above shows that annual per capita meat consumption may differ by a factor of two or more depending on the data series. There is an obvious risk of mixing these data when consumption statistics are used for subsequent calculations of, for example, environmental and health effects. In environmental assessments of meat and in dietary recommendations, it is thus crucial to specify the functional unit and to define whether it refers to meat, including or excluding bones, and whether it is after weight reduction by cooking, as well as if losses in distribution and consumer level are included. The choice of meat-consumption data should further correspond to the functional unit in calculations used to formulate recommendations and policy decisions.

Conclusions and Recommendations

We have discussed the reliability of meat-consumption statistics with the aim of identifying limitations, strengths, and uncertainties in methods and data. The results show various discrepancies regarding how meat-consumption data are produced and presented, awareness of which is important for a correct understanding and interpretation of the statistics. Increased attentiveness to these issues, in turn, will have a significant impact on diet recommendations and policy tools related to health and environmental issues, such as climate change and land use.

We advance several recommendations to improve the production, presentation, and use of meat-consumption statistics. First, the definitions of meat consumption and supply on national, regional, and international levels should be standardized and harmonized to the greatest extent possible. Second, methods for obtaining meat-consumption data should be of the highest possible quality to ensure high statistical validity. Third, relevant national, regional, and international statistical agencies should enhance the transparency of meat-consumption data. Fourth, assumptions regarding weight of bones and other

inedible body parts of the animal, food losses and waste in the stages up to and after retail sale, weight reductions due to cooking, and nonmeat components in mixed-meat products and prepared meals, should be presented in a more accessible and straightforward manner. Finally, limitations, uncertainties and discrepancies in meat-consumption data should be addressed for correct utilization in subsequent calculations of, for instance, the environmental and health effects of meat.

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Paper II

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Sustainable meat consumption: A quantitative analysis of nutritional intake, greenhouse gas emissions and land use from a Swedish perspective

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ABSTRACT

Background: Food consumption is one of the most important drivers of environmental pressures. Adoption of healthy diets is suggested to be an option for less environmentally intensive food habits and improved public health. In particular, changes in meat consumption are believed to bring potential benefits.
Objective: To quantify the impact of changes in meat consumption on the dietary contribution of nutrients, GHG emissions and on land requirement.
Design: Scenario analysis is performed for three scenarios representing different variants of meat consumption in Sweden. The reference scenario is based on average Swedish meat consumption while NUTR-1 and NUTR-2 are hypothetical scenarios in line with prevailing dietary guidelines. The results are evaluated in relation to the recommended daily intake of nutrients, international climate goals and global capacity for sustainable expansion of agricultural land. Uncertainties and variations in data are captured by using Monte Carlo simulation.
Results: Meat consumption in line with nutritional guidelines, implying an approximate 25% reduction of Swedish average intake, reduces the contribution of total and saturated fat by 59–76%, energy, iron and zinc by about half and protein by one quarter. Restrictions in meat consumption are most critical for the intake of iron and zinc, whereas positive effects on public health are expected due to the reduced intake of saturated fat. Aligning meat consumption with dietary guidelines reduces GHG emissions from meat production from 40% to approximately 15–25% of the long-term (2050) per capita budget of sustainable GHG emissions and the share of per capita available cropland from 50% to 20–30%.
Conclusions: This quantitative analysis suggests that beneficial synergies, in terms of public health, GHG emissions and land use pressure, can be provided by reducing current Swedish meat consumption.
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Introduction

Diet and nutrition are major determinants for maintaining health and preventing non-communicable diseases (WHO/FAO, 2003). During the past decades, a transition towards energy-dense diets and sedentary lifestyles have resulted in a global epidemic of overweight and obesity, affecting a fifth of the world's adult population (Finucane et al., 2011; WHO, 2012). Apart from the impact on public health, food consumption has been identified as one of the most important drivers of environmental pressures (UNEP, 2010a). Food is estimated to be responsible for 20–60% of environmental impact, including greenhouse gas (GHG) emissions, eutrophication, acidification and eco-toxicity from European household consumption (Weidema et al., 2008) and account for about 30% of global anthropogenic GHG emissions (Garnett, 2011). To avoid dangerous and irreversible effects of climate change on the ecosystem it is argued that global GHG emissions need to peak within the coming 10–15 years (IPCC, 2007a), which will require substantial mitigation efforts on all fronts not least in the food sector.

A growing body of literature suggests that the adoption of healthy diets could offer multiple benefits, including improved

Abbreviations: GHG, greenhouse gas; REF, reference scenario; NUTR-1, nutrition one scenario; NUTR-2, nutrition two scenario; RDI, recommended daily intake; LCA, life cycle assessment; CO_2e, carbon dioxide equivalent.

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public health and potentially reduced environmental impact (McMichael et al., 2007; Saxe et al., 2013; Scarborough et al., 2012; Briggs et al., 2013). A reduced consumption of meat in favour of plant based foods has in particular been demonstrated to bring potential benefits in regions with affluent diets (McMichael et al., 2007; Saxe et al., 2013; Scarborough et al., 2012; de Boer and Aiking, 2011; González et al., 2011). The knowledge of how nutrient intake from different diets is affected by changes in meat consumption is, however, still limited (Westland and Crawley, 2012). Meat is a good source of many minerals (iron, zinc and selenium) and vitamins (vitamin D, riboflavin, B_{12}) and contains all the essential amino acids (Millward and Garnett, 2010). A high intake of meat has, however, been associated with an excessive intake of energy, cholesterol and saturated fat, which are known risk factors for coronary heart disease (Baxter et al., 2006; Micha et al., 2010). Red (beef, lamb and pork) and processed meat (bacon, salami, sausages, hot dogs, etc.) have in addition been associated with an increased risk of certain cancers (Ferguson, 2010; WCRF/AICR, 2007). Food of animal origin is in general also more climate and land intensive compared to food of vegetable origin (Garnett, 2009; González et al., 2011; Wirsenius et al., 2010).

In a future development of holistic guidelines and policy tools promoting more sustainable food consumption it is essential to consider both health and environmental aspects. Nutritional aspects have previously been included in environmental analysis in various but often limited ways, for instance, as a determinant of the functional unit in life-cycle assessments (LCA) (González et al., 2011; Smedman et al., 2010; Vieux et al., 2012), in scenario analysis (Tukker et al., 2011; Wolf et al., 2011; Temme et al., 2013) and in qualitative discussions (Garnett, 2008; Millward and Garnett, 2010). However, there is currently a lack of studies that have analysed the effect of dietary change in a broader perspective and of studies in which the effect on both nutrition and environment has explicitly been quantified (Hallström et al., 2011).

The objective of this study is to quantify the impact of changes in meat consumption on the dietary contribution of nutrients, GHG emissions and on land requirement, in order to identify beneficial synergies (and potential conflicts/drawbacks) for more sustainable food consumption patterns. This, in turn, could be used as a motivation for more integrated policies within the food, climate and agriculture sector.

Methodology and assumptions

Scope of the study

Scenario analysis is performed for three scenarios representing different variants of meat consumption in Sweden. The reference scenario (REF) is based on current average Swedish meat consumption while NUTR-1 and NUTR-2 are hypothetical scenarios in which meat consumption is based on criteria from the perspectives of nutrition and health. In NUTR-2 also criteria for efficient use of resources in the production system are considered. The scenarios are developed to represent Swedish conditions in a near-term perspective but are also applicable to countries where meat consumption is based on similar production systems. The results are presented per capita and are evaluated in relation to the recommended daily intake of nutrients, international climate goals and the global capacity for sustainable expansion of agricultural land.

This paper analyses solely the effects of changes in meat consumption while the composition of the remaining diet is assumed to be unchanged. The study design is chosen, firstly, to account for the total potential of reducing GHG emissions and land use by changing meat consumption according to the studied scenarios, and secondly, to analyse to what extent meat in the Swedish diet needs to be replaced by other foods from a nutritional perspective.

Per capita supply of meat differs from the actual intake due to losses and wastage along the chain of supply and handling. Hence, quantification of nutrient intake needs to be based on consumption data while production data, which refer to the available agricultural supply, is used to quantify environmental impacts. In this study quantification of nutrient intake is based on the per capita supply of bone-free, uncooked meat available for human consumption, including wastage during production and retail. The results of this study should thus be interpreted as the supply of nutrients that is theoretically available for consumption if no meat is wasted at the consumer level. The effect of wastage at consumer level on nutrient intake and environmental impact from meat is further discussed in Section 'Limitations and uncertainties'.

To capture the uncertainty and variation in nutrient content, GHG emissions and land use, Monte Carlo simulation was used (Rubinstein and Kroese, 2007). In Monte Carlo simulation, parameters are described by a probability distribution, rather than a single deterministic value, and the calculation is repeated a number of times, here 10,000; each time randomly drawing a parameter value from the probability distribution. The result of a Monte Carlo analysis consists of a number of possible outcomes of the calculation, hence giving a representation of the probability of different results depending on the uncertainty and variation in the input data.

For this article a section of complementary materials is available, in which the methodological approach, made assumptions and an extensive literature review are described in detail.

Scenario description

REF

This reference scenario (REF) is designed to reflect current average per capita consumption of meat in Sweden. Total meat consumption amounts to 169 g uncooked, pure meat (i.e. excluding bones and non-meat ingredients in charcuteries) per day, with beef, pork and chicken accounting for 30%, 47% and 24% of the total intake, respectively (Table 1). These amounts are based on data from national statistics (data for 2009), which refer to the per capita supply of meat available for consumption after adjustment for losses between the production and household level (i.e. amounts purchased at retail and "away-from-home consumption") (SBA, 2011). A detailed description of assumptions made in the development of the reference scenario is found in the section of complementary materials.

NUTR-1

The amount of meat consumed in this nutrition one scenario (NUTR-1) is based on prevailing dietary guidelines. Total meat consumption is limited to 126 g uncooked, pure meat per day (excluding 47% of non-meat content in mixed charcuteries), as suggested by the Swedish Food Authority (Enghardt Barbieri and Lindvall, 2003). Consumption of red meat is restricted to 60 g (uncooked weight) per day (50% beef, 50% pork) and consumption of charcuteries is reduced to zero, which corresponds to the public health recommendation by the World Cancer Research Fund (e.g. max 300 g cooked, equivalent to 400–450 g uncooked, red meat per week, avoid processed meat) (WCRF/AICR, 2007).

NUTR-2

As in NUTR-1, the total meat consumption in this nutrition two scenario (NUTR-2) is limited to 126 g uncooked meat per day and the intake of charcuteries is reduced to zero. In this scenario, the beef comes entirely from production systems that produce both milk and meat, which are more resource efficient than systems producing only meat, since the emissions from enteric fermentation, feed production, etc. can be split between the milk and the meat. As a co-product from combined meat and milk production,

Table 1
Meat consumption and production in the scenarios studied.

Scenario	Beef	Pork	Chicken	Mixed charcuteries	Unmixed charcuteries	Total
Meat consumption[a,d] (g/capita and day)						
REF[b]	50	79	40	–	–	169
REF[c]	44	47	40	48	14	193
NUTR-1	30	30	66	0	0	126
NUTR-2	13	0	113	0	0	126

	Beef		Pork		Chicken		Total	
	kg/yr	g/d	kg/yr	g/d	kg/yr	g/d	kg/yr	g/d
Meat production[a] (kg/capita and year, g/capita and day)								
REF	19	53	30	83	15	42	65	178
NUTR-1	12	32	12	32	25	69	48	133
NUTR-2	5	14	0	0	43	119	48	133

[a] Amounts refer to uncooked weight.
[b] Refers to the consumption of pure meat excluding non-meat ingredients in charcuteries. Mixed and unmixed charcuteries account for 15% and 8% of the pure meat intake, respectively.
[c] Refers to the consumption of meat expressed as the total weight of the final meat products (including non-meat ingredients in charcuteries).
[d] A waste percentage of 5% is assumed between farm gate (production) and household (consumption).

one kg of milk produces 0.028 kg of beef meat from the culled cows and their raised offspring (based on a recruitment rate of 38% and a milk yield of 8300 kg ECM/year). The supply of beef at farm gate is restricted to 14 g per day, an amount corresponding to the Swedish dietary guidelines for dairy consumption (0.5 l of milk equivalents per person and day) (SFA, 2012a). The remaining amount of meat is assumed to be consumed as chicken since chicken has a high efficiency converting feed to meat (i.e. feed conversion ratio), and thereby uses cropland more efficiently than beef and pork.

Analysis of nutrient intake

Meat consumption

Table 1 illustrates the amounts of meat consumed in the scenarios studied. Estimated consumption of pure meat is specified as well as the proportion of meat consumed as mixed and unmixed charcuteries.

Nutrient intake from meat

The nutrient intake from meat is quantified for energy, protein, total fat, saturated fat, iron and zinc, all of which are nutrients largely provided by meat in regions with affluent diet (Millward and Garnett, 2010). The quantification of nutrient intake is based on the amounts of meat consumed as presented in Table 1. The quantification of nutrient intake from charcuteries is based on the total weight of the final meat products, i.e. including the non-meat content, representing 47% and 3% of the total consumption of mixed and unmixed charcuteries, respectively (Eidstedt, 1998). As the nutrient content in meat also varies depending on the cut of meat, the consumption of meat in the scenarios studied is further categorised as specific meat products. The proportions between the different meat products are based on Swedish trade statistics after correction for small bones remaining at retail, i.e. they correspond to the average sale of bone-free meat in Sweden (data from 1993) (SFA/SBA, 2011). For chicken the corresponding proportions have been estimated in communication with the Swedish Poultry Meat Association. Consumption of beef, pork, chicken, mixed and unmixed charcuteries are divided among 12, 9, 11 and 5 different meat products, respectively. In total, meat consumption was divided among 37 different meat products of varying nutrient content (more details in the section of complementary materials). The nutrient content of the different meat products is calculated for uncooked meat, based on data from the Swedish Food Agency's food data base, version 2012/01/06.

Uncertainty in nutrient composition

The nutrient composition can vary in meat products originating from the same type of animal. For example, for chicken the nutritional value depends largely on whether the meat considered includes or excludes the skin. Depending on the cut of meat the proportions of lean to fat tissue will vary, which also affects the composition of other nutrients.

To account for variability in nutrient composition a random value for the nutrient intake in the Monte Carlo simulation was determined by drawing a value from a discrete distribution containing the percentage of consumption of the different meat products and the nutrient intake they correspond to.

Recommended intake of nutrients

The average contribution of nutrients from meat in the scenarios studied is evaluated in relation to the mean recommended daily intake (RDI) of nutrients (NCM, 2004). The energy requirements for women and men are based on the mean reference energy requirements for adults (18–60 years) of average weight and level of physical activity. The RDI of total fat, saturated fat and protein is estimated based on the recommendation that they should provide 25–35%, ⩽10% and 10–20% of total energy intake (E%), respectively (NCM, 2004). RDI of total fat is set to 30 E%, which is the population goal. RDI of protein is set to 10 E%, which corresponds to the WHO recommendation of a daily intake of 0.8 g protein per kg body weight (WHO, 2002). RDI for iron and zinc are based on recommended intake levels for adults. This represents an intake that is likely to meet the nutrient requirement of 97.5% of the population and is calculated by adding a safety margin of two standard deviations to the average nutrient requirement of the population (NCM, 2004). An overview of RDI levels used in the analysis is provided in the section of complementary materials.

Analysis of greenhouse gas emissions and land requirement

Meat production

The production of bone-free meat required to meet the amounts consumed in each scenario (Table 1) is calculated by assuming a percentage of waste (input for processing and food waste) of 5% of the bone-free meat between farm gate and household, which is consistent with the estimate of post-farm losses in Swedish consumption statistics (SBA, personal communication).

Greenhouse gas emissions and land requirement from meat in different production systems

Table 2 provides information on the origin and production system of meat available for consumption in Sweden, as well as amounts of GHG emissions and land use per meat type, including their uncertainty intervals (more details in the section of complementary materials).

The processes contributing to major GHG emissions in meat production are the production of feed, enteric fermentation from the digestive system of mainly ruminants, manure handling and energy requirements for the housing of livestock. GHG emissions are calculated using LCA methodology (ISO, 2006a,b), in which emissions of all relevant GHG gases, carbon dioxide, methane and nitrous oxide, from all phases in the life-cycle of meat are quantified. GHG emissions of meat are commonly expressed as kg of carbon dioxide equivalents (CO_2e), in which all GHGs have been recalculated to the global warming potential of CO_2 and summed, per kg of carcass weight or bone-free meat. Effects on the per kg GHG emissions due to carbon sequestration in pastures or from land use changes, e.g. deforestation, is not included.

The area needed for feed production in the different meat production systems (Table 2) are calculated from feed intake data (Cederberg et al., 2009) by dividing the amount of feed needed per kg of meat by the yield per ha for different types of feed (forage, pasture, grain and protein feed) (Flysjö et al., 2008).

GHG emissions and land requirements for meat production are reported per bone-free weight at the farm gate, hence, pre- and post-slaughter transports and slaughtering, packaging, storage and preparation are not included. The environmental impact per kg meat is generally small for these stages compared to the agricultural phase (Sonesson et al., 2010), thus, these potential emissions are not included in this study.

Uncertainty in greenhouse gas emissions and land requirement

Previous research has shown that the environmental impact of meat production can vary considerably due to inherent differences in production system (de Vries and de Boer, 2010). The intensity in production, amounts and types of feed, slaughter age and manure handling have proven to be factors of importance for the overall environmental performance, including GHG emissions (Röös et al., 2013). In addition, determining the emissions arising from biological processes that are difficult to measure and model, such as methane from enteric fermentation in ruminants and nitrous oxide emissions from soils used for feed production, is associated with large uncertainties (IPCC, 2006). Methodological aspects, such as the determination of system boundaries, add to the total uncertainty of the environmental impact of meat.

Table 2
GHG emissions and land requirement for different production systems.

		Percentage of total consumption (%)	GHG emissions per kg bone-free meat[a] (kg CO_2e)	Uncertainty range[b] (kg CO_2e)	Land use per kg bone-free meat[a] (m2)	Uncertainty range[c] (m^2)
Beef						
System:	Sweden					
B.1	– Culled cows from milk production	11	20	12–34	31	21–62
B.2	– Bulls from milk production	13	22	13–37	31	20–61
B.3	– Steers from milk production	5	28	17–47	45	30–89
B.4	– Heifers from milk production	1	31	18–52	47	32–95
B.5	– Suckler production (cows and off-spring)	17	31	19–53	53	35–106
	Ireland					
B.6	– Suckler production (cows and off-spring)	18	31	19–53	53	35–106
	Germany, Poland and Denmark					
B.7	– Bulls from milk production	18	21	13–36	31	20–61
	Uruguay					
B.8	– Suckler production (cows and off-spring)	1.5	31	19–53	53	35–106
	Brazil					
B.9	– Extensive suckler production	0.5	41	25–70	250	150–350
	Other imported					
B.10	– Calves from milk production, extensive	15	25	15–42	40	27–79
Pork						
	Sweden					
P.1	– Intensive indoor	65	5.5	3.2–8.7	10	5–19
	Import					
P.2	– Intensive indoor	35	5.5	3.2–8.7	10	5–19
Chicken						
	Sweden					
C.1	– Intensive indoor	70	2.6	1.2–5.2	7	3–15
	Import					
C.2	– Intensive indoor	30	2.6	1.2–5.2	7	3–15

[a] The results have been recalculated from "carcass weight" to "bone-free meat" using a yield factor of 75%, 62% and 76% for beef, pork and chicken, respectively.
[b] The uncertainty range corresponds to min and max values calculated using a uncertainty importance analysis with the most influential parameters (more details in the section of complementary materials).
[c] The uncertainty ranges were set by using minimum and maximum values of the amount of feed used in production and the yield levels (more details in the section of complementary material).

To account for variability in GHG emissions and land requirement uncertainty ranges, used in the Monte Carlo simulation, are determined based on an uncertainty importance analysis (see section of complementary materials) (Björklund, 2002) and matched against estimates.

Sustainable level of greenhouse gas emissions and land requirement

GHG emissions from meat production in the scenarios studied are evaluated in relation to a theoretically sustainable level of emissions, here set to 1.5 tonnes of CO_2e per person and year. Due to human activities global GHG emissions have increased by 70% during the past 40 years. Scientific evidence indicates that a rise in global mean temperature will result in adverse effects including serious impact on the environment and future availability of food and water (IPCC, 2007b; Stern, 2006). To realise international climate targets, the global GHG emissions for 1990 will have to be cut by 50% by 2050, which requires that global emissions are reduced to levels of 1–2 tonnes of CO_2e per person and year (EC, 2006; UNEP, 2010b).

Land requirement for meat production in the scenarios studied is evaluated in relation to a theoretical limit for sustainable expansion of global agricultural land until 2050. Agricultural land covers somewhat more than a third of global land areas (approx. 4900 Mha). About 12% of the global land surface is currently used for cropping (approx. 1500 Mha) and the remaining agricultural land is covered by permanent meadows and pastures (approx. 3400 Mha) (FAO STAT, 2012). According to Rockstrom et al. (2009) no more than 15% of the global land surface should be converted to cropland for a land system change operated within the planetary boundaries. On the assumption that the world population will be nine billion by 2050, as projected by the United Nations (2010), available land areas for sustainable cropping will be about 0.22 ha per person, which is the figure used in this analysis. The global availability of total agriculture land (cropland and pastures) is currently around 0.7 ha per person and will be reduced to 0.54 ha per person by 2050 if agricultural land does not increase further.

Results

Nutrient intake

The daily contribution of nutrients from meat consumption in the scenarios studied is shown in Table 3. A dietary change in line with prevailing guidelines for a healthy meat intake (NUTR-1, NUTR-2) would have the strongest effect on the contribution of total fat and saturated fat which would be reduced by 59–76% compared to the reference scenario. The contribution of energy, iron and zinc would be reduced by about half (42–62%) and the contribution of protein by about a quarter (23–26%). For most nutrients the uncertainty intervals are in the range of ±50% but for some they are even larger. In Fig. 1 the contribution of nutrients is illustrated in relation to mean RDI levels for men and women. Current Swedish meat consumption (REF) contributes to about 14% of the recommended daily energy requirement, about one third of the maximum RDI of total fat and saturated fat (28%, 34%) and between approximately one and two thirds of the RDI of protein, iron and zinc (54%, 32%, 65%). In NUTR-1 and NUTR-2 the contribution of energy, protein and iron is, on average, equivalent to 7%, 40% and 14% of the RDI, respectively. The corresponding contribution of total fat, saturated fat and zinc is between 8–11%, 8–14% and 29–38% of RDI, respectively.

Greenhouse gas emissions

In Fig. 2 GHG emissions from meat production in the scenarios studied are illustrated in relation to the sustainable level of total emissions per capita and year. The production of meat currently consumed in Sweden (REF) is responsible for GHG emissions in the range of 0.6 tonnes of CO_2e per capita and year, representing approximately 40% of the total budget for sustainable emissions. A dietary change towards healthier meat consumption would reduce the GHG emissions to approximately 0.4 and 0.2 tonnes of CO_2e per capita and year in NUTR-1 and NUTR-2, respectively. Meat consumption would in these scenarios account for some 13–26% of the yearly per capita GHG emission budget. Uncertainty intervals for GHG emissions in the scenarios studied range from approximately −15% to +80% from the average value.

Land requirement

In Fig. 3 land requirement for meat production in the scenarios studied is illustrated. The results are put in relation to the

Table 3
Daily contribution of nutrients from meat consumption in the scenarios studied (uncertainty intervals[a]).

	REF	NUTR-1	NUTR-2
Energy (MJ/d)	1.5 (1.1–2.0)	0.8 (0.6–1.1)	0.7 (0.6–0.9)
Total fat (g/d)	24 (13–40)	9.7 (2.6–21)	6.6 (1.7–14)
Sat. fat (g/d)	9.6 (5.0–16)	3.9 (1.0–8.2)	2.3 (0.6–4.9)
Protein (g/d)	34 (29–37)	25 (22–28)	26 (23–29)
Iron (mg/d)	3.8 (2.2–10)	1.8 (1.3–2.2)	1.4 (1.1–1.9)
Zinc (mg/d)	5.2 (4.1–6.4)	3.0 (2.3–3.8)	2.3 (1.4–2.9)

[a] The uncertainty range corresponds to min and max values obtained from the Monte Carlo Analysis.

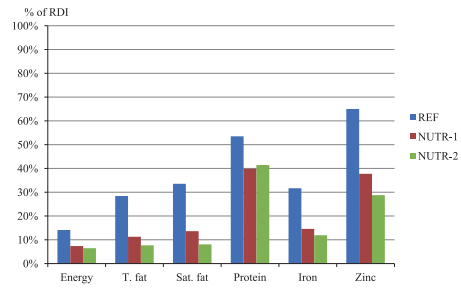

Fig. 1. Daily contribution of nutrients from meat as a percentage of RDI. RDI of total fat, saturated fat and protein corresponds to 30%, 10% and 10% of recommended daily energy intake (11 MJ/d), respectively. For more information, see text.

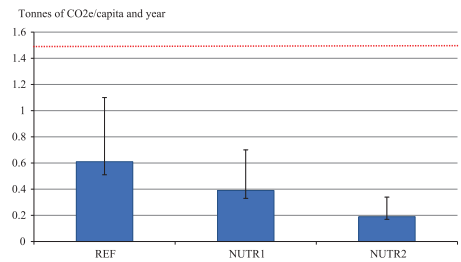

Fig. 2. Greenhouse gas emissions from meat production (tonnes of CO_2e/capita and year). Sustainable level of total GHG emissions is marked by the dotted line.

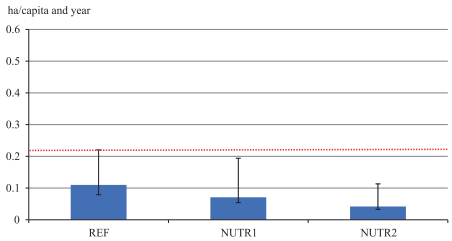

Fig. 3. Land requirement for meat production (ha/capita and year). The theoretical availability of cropland in 2050 is marked by a dotted line. The theoretical availability of total agricultural land (including pasture land) in 2050 is 0.54 ha per capita.

theoretical availability of global cropland, as meat consumed in Sweden is mainly produced on cropland. The production of meat consumed in Sweden (REF) demands 0.11 ha per capita and year, representing half of the theoretically available cropland in 2050. A dietary change towards healthier meat consumption (NUTR-1, NUTR-2) would reduce the demand for agricultural land to 0.04–0.07 ha per capita and year, which would reduce the share of global per capita available cropland for meat production to 19–32%. The uncertainty intervals for land requirement in the scenarios studied range roughly from −25% to +100% from the average value.

Discussion

Interpretation of the results on nutrient intake

In relation to RDI levels reduced meat consumption has a minor effect on the overall nutritional contribution of energy and protein, whereas the contribution of total fat, saturated fat, iron and zinc is influenced more strongly (Fig. 1).

To evaluate the effect of changes in meat consumption on nutritional status, both the nutrient contribution of meat in relation to other food groups and the current nutritional status of the Swedish population must be taken into consideration. The latest Swedish dietary survey among adults, performed in 2010–2011 (Amcoff et al., 2012), confirmed that meat and meat products make an important contribution to several nutrients in the average Swedish diet (11% of energy; 25% of protein; 19% of total fat, 19% of saturated fat, 21% of iron, 30% of zinc). According to the same survey average intake of protein, zinc and iron among Swedish adults was well above (28–38%) RDI levels. The exception was iron intake among fertile women which was lower than recommended. This indicates that a substantial share of the male population would meet RDI of these nutrients even if meat consumption was reduced in line with the studied scenarios. For women the need to replace nutrient losses from a reduced meat intake appears to be greater, in particular to meet the requirements of iron. Further analysis is however required to confirm these indications.

The decrease of saturated fat in NUTR-1 and NUTR-2 is assumed to bring positive health effects for the majority of the Swedish population. In Sweden average intake levels of saturated fat are higher than recommended (Amcoff et al., 2012), which is undesirable due to the associations with an increased risk for cardiovascular disease (WHO/FAO, 2003). The reduced intake of saturated fat in the studied scenarios corresponds to a 20–25% reduction of total saturated fat in the average Swedish diet. Case studies in the UK and Brazil suggest that a 30% reduction in intake of saturated fat and cholesterol from animal sources could decrease the burden of ischaemic hearth disease by about 15% (Friel et al., 2009).

Among individuals whose nutritional intake would fall below RDI levels in NUTR-1 and NUTR -2, the diet would need to be complemented by an increased intake of other foods. Desirable changes towards a healthier diet in Sweden include an increased intake of dietary fibre, whole grains, fruit and vegetables (SFA, 2012b). Substituting meat with an increased intake of plant based foods in general also reduce GHG emissions and land use requirement (Scarborough et al., 2012; Stehfest et al., 2009). From a resource perspective, intake of the critical nutrients could alternatively be improved by increasing the consumption of food produced from by-products such as blood or offal, by fortification or supplementation. As previously discussed, the intake of iron among women of fertile age is of particular concern at reduced meat intake, but the intake of zinc should also be given extra consideration (Geissler and Singh, 2011; Millward and Garnett, 2010; Temme et al., 2013). Iron and zinc are of extra concern also due to the lower bioavailability in plant based foods. Examples of plant based foods rich in iron and zinc are cereals, pulses, nuts and seeds. The effect of substituting meat with different, e.g. plant based, foods on nutritional intake, health and the environment requires further analysis. The results on nutrient intake are in agreement with findings in previous research (NCM, 2013).

The REF scenario showed that Swedish average per capita consumption of bone-less, uncooked meat is about 62 kg per year (170 g/d), or 70 kg per year (190 g/d) if the whole weight of charcuteries, including the non-meat content, is accounted for. Red meat from pork and beef constituted three quarters of the total meat consumption (129 g/d) and one third of the total meat was consumed as charcuteries (64 g/d, if expressed as the total weight of the charcuteries). According to our results the consumption levels of red meat in Sweden are twice as high as the recommended levels from a public health perspective (e.g. 60 g/d of uncooked red meat), even after accounting for a 10% deduction for household wastage. Also the intake of charcuteries is above recommended levels (e.g. avoid processed meat) (WCRF/AICR, 2007). The results from this study are in line with previous findings where Swedish consumption of uncooked total, red and processed meat have been estimated to 120–205 g/d (119–134 g/d for women, 164–205 g/d for men, adolescents and children), 60–135 g/d (60–68 g/d for women, 100–135 g/d for men, adolescents and children) and 30–60 g/d (30–35 g/d for women, 37–60 g/d for men, adolescents and children), respectively (NCM, 2013). The NUTR-2 scenario shows that a per capita consumption of 5 kg of beef per year, or 13 g/d, is required to provide the amount of milk corresponding to guidelines for healthy milk consumption in Sweden (e.g. 0.5 l of milk equivalents per person and day) (SFA, 2012a), assumed that all milk comes from combined milk and meat production systems. From this perspective, current average beef consumption of approximately 50 g/d could be reduced by 70%.

Interpretation of the results on GHG emissions

A dietary change towards healthier meat consumption would cut the emissions of GHG by about half (40–70%) but meat consumption would still account for some 15–25% of the required emission target. Furthermore, this target must also cover emissions from other foods, especially, and importantly, from dairy products which also cause substantial GHG emissions (Weidema et al., 2008), and from other activities such as housing, transportation and other consumption.

A previous study quantified the climate impact of Swedish meat consumption in 2005 to 0.7 tonnes per capita (Cederberg et al., 2009) which is of the same magnitude as the finding of this study. Estimates from developed countries indicate that GHG emissions involved in the total diet are in the range of 2–3 tonnes of CO_2e per capita and year (Berners-Lee et al., 2012; Pradhan et al.,

2013; SEPA, 2011), equivalent to about 15–28% of overall national emissions (Garnett, 2011). Swedish per capita emissions amount to about 10 tonnes of CO_2e per capita and year. The private consumption is responsible for 8 tonnes per capita and year of which food is estimated to account for about 2 tonnes per capita and year (excluding emissions from imported goods and services and land use change) (SEPA, 2011). According to this study healthier meat consumption, equivalent to a 25% reduction of current consumption, could reduce Swedish per capita emissions from private consumption by 3–5%. Corresponding GHG emissions from Swedish food consumption would be reduced by 11–21%.

For a complete analysis of the mitigation potential of dietary change also GHG emission from a potential increase in consumption of complementary foods, needed to compensate the intake of critical nutrients, should be included. Dietary change towards healthy meat intake, as defined in this study, would thus have a rather limited effect on the total GHG mitigation potential, whereas the impact on the emission from the diet, depending on the substitute, could be large. To achieve sustainable levels of GHG emissions would require either more drastic changes in diet, in combination with improved production systems, or that other sectors need to bear a greater share of emission reduction.

Interpretation of results on land requirement

According to the results of this study current Swedish meat consumption require an area representing half of the theoretically available cropland in 2050. By aligning meat consumption with prevailing dietary guidelines the share of global per capita available cropland needed to produce the meat consumed is reduced to between 19% and 32%. However, the uncertainty is large due to the large variety of production systems and type of land used, and the land areas required could reach twice this amount. In the REF scenario this means that all available cropland in 2050 is used for meat production, assuming that all fodder is produced on cropland.

When discussing availability of agricultural land it is necessary to distinguish between different types of land. Meat consumption in the scenarios studied claims a minor share of the 0.54 ha per capita of total agricultural land assumed to be available in 2050. However, only some 30% of agricultural land available consists of cropland. Meadows and pastures, which represent the remaining agricultural area, are only to a limited extent suitable for cultivation. As the majority of current global food supply is dependent on cultivated land (Johansson, 2005) pressure on agricultural land is especially intense on cropland. As a rough estimate, beef consumed in Sweden uses on average approximately 17 m^2 of land for grazing per kg bone-free meat (calculations based on Cederberg et al. (2009)). By applying this average, grazing land represents between one third and a fifth of the total requirement of agricultural land for meat consumption in the scenarios studied. However, not all of the land used for grazing is pasture land, as grazing also takes place on cropland sown with ley. Current production systems for meat consumed in Sweden is thus to a large extent dependent on cropland. A transition towards more pasture-based beef production could reduce the competition for cropland but would result in a larger absolute requirement of agricultural land compared to the results in this study, since extensive ruminant production requires substantially more land than intensive ruminant production. An increased share of meat consumption coming from monogastric animals (i.e. pork or chicken), as in NUTR-2, would instead further increase the pressure on cropland for feed production (e.g. soy). Such a development could be a driving force for increased deforestation in some areas (Nepstad et al., 2006) and is competing for land with crops for direct human consumption.

The total land area required for the production of animal products consumed in Sweden was previously estimated to be 0.3 ha per capita by Johansson (2005). This area refers to the land needed for production of all animal products consumed in Sweden, including dairy products and eggs, which partly explain why it is notably larger than the result in this study. In the study by Johansson the domestic land use for animal production was further determined based on national agricultural statistics in contrast to this study where it is calculated by multiplying the amount of meat consumed by a land area determined by the average demand of feed and corresponding yields. The observed difference in results indicates that current agricultural systems are used less intensively compared to average production systems represented in LCA data. This in turn would mean that there is a potential for increased production of biomass for food or other purposes (e.g. biofuels and biochemicals) on existing Swedish agricultural land. However, to determine whether this hypothesis is correct, further research is required.

Limitations and uncertainties

As described previously, data on the nutritional composition of foods and the environmental impact of meat production are challenged by a number of uncertainty factors which will affect the overall reliability of further calculations. Accounting for all meat products sold on the Swedish market provides a great challenge due to differences in nutrient content and the diversity of products coming from production systems with great variability in connection to GHG emissions and land use. Therefore, in this study the uncertainties in data and variations in production systems are accounted for by using Monte Carlo analysis to establish realistic uncertainty intervals for nutrient content, GHG emissions and land use.

Factors contributing to uncertainty in the analysis of nutrient intake data include the lack of updated information on how consumption of beef, pork and charcuteries is distributed among different meat products. The proportions used are based on old data (from 1993) and thus do not capture changes in consumption patterns today. For example in this study ham, cutlet and bacon represented 60% of the total consumption of pork, whereas fillet and minced meat represented only 5% and 2% of pork consumption, respectively. During the two past decades it is probable that the preferences for meat products have changed, which would also affect the nutrient intake. Another limitation of this study is that the nutrient intake is quantified on the basis of composition of the raw meat product, i.e. changes in nutrient composition during preparation and cooking have not been taken into account. Cooking in general has a limited effect on the nutrients studied but may, for example, influence fat intake and result in leaching of minerals if the meat is boiled. The contribution of nutrients may also be influenced by the consumer's eating habits. In this study it was, for example, estimated that 40% of all chicken is eaten with the skin, based on estimates of the current proportions of chicken products sold on the Swedish market. Whether the skin is eaten or not has a large impact on the nutrient intake, especially on the energy and fat intake. Finally, in this study the analyse of nutrient intake is in this study limited to a few nutrients identified to be of importance. To fully understand the nutritional and health effects of reduced meat consumption requires that the total intake of nutrients is considered.

A methodological difficulty in studies in which both environmental impact and nutritional intake are quantified is the conversion between produced and consumed amounts of food. In this study the losses between production and the consumer were assumed to represent 5% of the bone-free carcass weight. According to a previous report the average percentage of meat wasted in Europe during post-harvest handling and storage, processing

and packaging and distribution are 0.7%, 5% and 4%, respectively (percentage of quantity entering each step) (FAO, 2011). The same report estimated that on average 11% of all purchased meat in Europe is wasted in the household. In this study the amount of meat consumed is assumed to equal the supply of meat available for consumption and vice versa, i.e. waste at consumer level is not accounted for. Underestimation of the losses between production and consumer would imply a lower actual intake of nutrients per amount of meat produced and a higher environmental impact per nutrient intake. The effect of household wastage must therefore be considered when comparing the nutrient intake levels with RDI levels. By including a waste percentage of 10% at consumer level the contribution of nutrients in REF would be 10% lower whereas GHG emissions and land requirement in NUTR1 and NUTR2 would increase by 7–16%. When interpreting the results it should be considered that meat consumption in REF is based on per capita consumption statistics, which hide large variations between different groups of the population. Because per capita statistics include women and elderly whose consumption is lower than average (Amcoff et al., 2012), it can be assumed that meat consumption in some groups of the population is higher than the levels in REF. That in turn would mean that meat consumption in these groups could be further reduced in order to be in line with health recommendations. This applies particularly to adult men who eat more meat and have lower iron requirements than women (Amcoff et al., 2012).

An increasingly important issue to consider in LCA of food is the effect of direct and indirect land use change (Ponsioen and Blonk, 2012). In recent studies the effects from land use change are often accounted for. However, since this article is based on previous LCA data, these effects have not been accounted for here. In future studies these aspects should also be considered.

The scenarios NUTR-1 and NUTR-2 are hypothetical scenarios developed based on health, climate and land-use perspectives. In these scenarios there is no consumption of charcuteries and pork (NUTR-2) and instead a higher consumption of chicken. NUTR-2 can thus be interpreted as a best-case scenario from the three perspectives studied. Whether this is a realistic scenario from a consumer perspective has not been considered in this study.

To broaden the understanding of the effects of dietary change the effect of different types of meat substitutes on nutritional intake, health and the environment needs to be analysed further. Further studies also need to include impacts on other environmental areas, such as the biodiversity, and impacts on social and economic aspects of sustainability.

Conclusions

The overall conclusions from this study can be summarised as follows:

- In relation to RDI, a reduction of average Swedish meat consumption to levels corresponding to Swedish dietary guidelines has a minor effect on the overall contribution of energy and protein, whereas the contribution of total fat, saturated fat, iron and zinc will be reduced considerably.
- Due to overconsumption, there may be scope for reducing meat consumption without any need for nutritional compensation in parts of the Swedish population. To what extent and by which foods meat should be substituted in a sustainable diet requires further analysis.
- Reduced meat consumption is expected to have a positive effect on public health due to the reduced intake of saturated fat.
- From a nutrition perspective restriction in meat consumption is most critical for the intake of iron and zinc.
- Swedish per capita consumption levels of red meat are twice as high as public health recommendations and also exceed recommended intake levels for processed meat.
- Average Swedish meat consumption is estimated to account for some 40% of the long-term per capita budget of sustainable GHG emissions and to occupy half of the available per capita cropland (2050).
- Meat consumption in line with dietary guidelines, implying an approximate 25% reduction of current intake, reduces GHG emissions from meat production to approximately 15–25% of the emission budget and the share of available cropland to 20–30%.
- The availability of global agricultural land may not be as critical but is largely dependent on the agricultural intensity, soil fertility, changing climatic conditions and on future demand for agricultural products for food and other purposes.
- The choice of meat products, variations in production systems and uncertainties in the methodology of calculation may affect the results for nutrient intake, GHG emissions and land use from meat consumption considerably.
- More research is needed to develop recommendations for alternative dietary patterns with lower environmental impact which also satisfy nutritional requirements.

Conflicts of interests

The authors have no conflicts of interest.

Acknowledgements

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Appendix A. Supplementary material

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Paper III

A healthier US diet could reduce greenhouse gas emissions from both the food and health care systems

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Abstract

The standard US diet contributes to greenhouse gas emissions (GHGE) from both the food system and, through its contribution to the high prevalence of non-communicable diseases, the health care system. To estimate the potential for diet change to reduce GHGE in the US, we created three model healthy alternative US diets by changing foods in the standard American diet linked to three non-communicable diseases. We then calculated the differences in GHGE between the standard and alternative diets. We found that adoption of healthier diets reduced the relative risk of diseases by 20-45%, US health care costs by US$B 54-72 per year, and GHGE per capita, per year by 133-1618 kg total (65-106 kg from the health care system, 68-1512 from the food system). Using the more relevant 20-year global warming potential for methane increased emissions substantially, especially from the food system. Emission reductions were equivalent to a maximum of 37% of the US Climate Action Plan's target of a 17% reduction in 2005 GHGE by 2020, and 150% of California's target of 1990 GHGE levels by 2020. Promoting diet change could be an effective and efficient strategy for reaching near-term climate change mitigation goals and simultaneously improving health.

1. Introduction

There is increasing scientific consensus that significant reductions in anthropogenic greenhouse gas emissions (GHGE) will have to be achieved in the next decade or two to limit global warming and avoid dangerous climate change (1-4). Our individual and collective diets, and the food system that creates and supports them, contribute about one third of total emissions globally (5). However, compared with other mitigation strategies, such as alternatives to fossil fuel energy sources and carbon capture and sequestration, the food system has received relatively little attention (6).

Diet change has the potential to reduce GHGE in the food system because the types and relative amounts of foods generate a wide range of emissions (7, 8). The climate impact of protein sources in our diet can, for example, vary by up to a factor 100 between the most and least GHGE intensive sources (9) Changing dietary patterns can reduce GHGE embodied in the diet by up to 50% (10). Diet change is an attractive mitigation option as it requires no new technologies, minimal new infrastructure, and has important positive externalities, including improved health. In addition, because agricultural production is the largest contributor of anthropogenic methane (CH4) (e.g. 34% of total US CH4 emissions (11)) which has a short life span in the atmosphere and a 20-year global warming potential (GWP) 86 times that of CO2 (12), diet change could especially contribute to reducing GHGE over the critical short term.

Diet change also has the potential to reduce GHGE in the health care system. The average intake of foods by the US population, the standard American diet (SAD), has become markedly less healthy in recent decades, and in combination with an increasingly sedentary lifestyle, has resulted in an epidemic of non-communicable diseases (NCD) (13). Since the late 1950s, caloric intake per capita increased by almost 40%, while the share of energy coming from refined carbohydrates, fatty meats and added fats increased along with a decreased intake of whole grains, fruit and vegetables (13). About half of all US adults have one or more NCDs, and about two-thirds are overweight or obese (14); the prevalence of diabetes was 9% in 2012 (15, 16); the prevalence of cardiovascular disease in the population over 20 years old in 2010 was 35% (17) and ~40% of the population are predicted to be diagnosed with cancer during their lifetime (18). NCDs are important contributors to increasing U.S. health care costs to almost \$3 trillion yr^{-1}, representing 18% of the total US GDP in 2014, and a projected 20% by 2022 (19). NCDs can to a large extent be prevented by adopting a healthy lifestyle, including healthy diets (20, 21).

The link between diet, health, and climate is a new but growing research field. By using life cycle assessment (LCA) (22, 23), the climate impact of individual foods, meals and whole diets have been calculated. Health aspects have been included in LCAs by expressing the climate impact of the food with a functional unit that relates to the nutritional content (e.g. energy or protein) (24), or by comparing the climate impact of the average food consumption patterns in a population with a hypothetical healthy diet based on dietary recommendations (25-28). So far there have been only a few studies using epidemiological data to study the relationship between diets' impact on both GHGE via the food system, and on health (29-32). The results from these studies show that dietary change offers a large potential to simultaneously improve health and reduce GHGE from the food system, but have not documented the potential to reduce GHGE in the health care sector via decreases in health care expenditures. Dietary changes most often suggested to bring both health and environmental benefits are limiting total energy intake, reducing intake of red and processed meat and empty calories, and increasing intake of fruit, vegetables, and whole grains (5, 33).

The overall goal of this study was to estimate the combined net effect of the adoption of incrementally more healthy diets by the US population on GHGE from both in the food system, and in the health care system due to reduction in the three diet-related NCDs for which adequate data existed. To our knowledge this has not been attempted by other researchers. Our results provide new insights into the relationship between food, health and climate that have important policy implications in the development of more sustainable food systems and the mitigation of anthropogenic climate change.

2. Methods

2.1 System Boundaries

The spatial boundary of our study was the US, although we extrapolated LCA data from other countries when US data were not available, and data for disease risk were from various countries. The reference year for our study was 2013, and we used either data for 2013, extrapolated data to 2013 based on trends, or data for the closest year when there was no basis for extrapolation. The system boundaries for GHGE in the health care sector were the

components of the health care sector associated with the studied diet related diseases. The system boundaries for GHGE in the food system were inputs to production at the distal end, through retail at the proximal end; thus emissions from land use change, retail to consumer transport, storage and preparation at the consumer stages, and food waste disposal were not included. However, in calculating the food intake levels of the studied scenarios, we began with food availability at the primary (farm gate) level and adjusted this using estimates of food losses through the consumer stage, so that the upstream GHGE from production through retail for the food wasted at the consumer and postconsumer stages was included.

2.2. Overview of Methodology

Our method for estimating the effect of dietary change on GHGE can be described in six steps (Fig. S1).

In step 1, we defined as the reference diet the standard American diet (SAD), using data on loss-adjusted food availability at the consumer level in the US (34) and created three counterfactual healthy alternative diets (HADs) by adjusting a selection of the foods in the SAD. In step 2, we estimated the changes in disease prevalence from dietary change, based on RR (relative risk) estimates found in published meta-analyses. In step 3, we estimated changes in health care expenditures from dietary change, based on changes in disease prevalence from step 2, using the most recent reliable expenditure data. In step 4, we estimated the changes in GHGE in the health care sector (ΔGHGE-H) from dietary change, based on the per unit cost of health care from step 3, and GHGE US$$^{-1}$ from the Economic Input-Output Life Cycle Assessment (IO-LCA) at Carnegie Mellon University (35). In step 5, we estimated the changes in GHGE in the food system (ΔGHGE-F), from dietary change, Δ(SAD – HADs), using GHGE from LCA data found in the literature. In step 6 we estimated the net total change in GHGE from dietary change, ΔGHGE-T = (ΔGHGE-F + ΔGHGE-H).

2.3. Step 1. Developing Dietary Scenarios

To analyze the effect of dietary change on health and GHGE in the US, we used the standard American diet (SAD) as our reference, and compared it with three counterfactual healthy alternative diets (HAD-1, -2, and -3) for the foods changed. We calculated dietary intake levels in SAD based on per capita loss-adjusted food availability data by weight for 2012, the most recent year for which data were available (34). These data estimate the average food intake in the US in cooked weights, based on amounts available at farm gate, adjusted for losses from farm gate through post-consumer stages, including household waste and plate waste (36) In order to distinguish between unprocessed and processed meat, and between whole grains and refined grains, which are aggregated in the data provided by the USDA, we assumed that processed meat accounted for 22% of total meat intake (37) and consisted of 90% red meat, and that refined grains and whole grains accounted for 90% and 10% of total grain consumption, respectively (38, 39).

To create the HADs, we adjusted SAD only for foods for which (i) USDA dietary recommendations were consistent with international nutrition and health authorities (20, 38), (ii) there were documented GHGE estimates, and (iii) there were high quality data correlating them with disease. The dietary recommendations identified were: (i) eat no more calories than needed to maintain a healthy body weight, (ii) increase the proportion of calories coming from plant-based food, (iii) reduce the consumption of meat (especially coming from red and processed meat), and (iv) reduce the consumption of foods with low nutritional value (5). Creation of the HADs thus involved only a portion of the total SAD; we did not change any other food groups (e.g. sugar sweetened beverages, unprocessed white meat, fish, dairy, eggs).

In HAD-1 we increased the amount of fruits, vegetables and whole grains, and reduced the amount of red and processed meat, and refined grains from the levels in the SAD to the USDA recommended levels ("USDA Food Pattern" adjusted to 2000 kcal day^{-1}) (38) (Table 1). Processed meat was limited to 20% of total red meat, based on the recommendation by the WCRF that processed meat should be avoided or limited as much as possible (20), and was assumed to come from red meat (i.e. no consumption of white processed meat). Whole grains and refined grains contributed 60% and 40% respectively, of total grain intake, based on the USDA recommendation that at least half of the grain consumption should come from whole grains (38). We limited fruit juice to 20% of total fruit consumption based on the USDA recommendation that the majority of fruit intake should come from whole fruits (38). By using whole food-based recommendations (e.g. vegetables), as opposed to nutrient-based recommendations (e.g. fiber), we reduced the risk of double counting health effects from nutrients found in various food groups.

We converted recommended food consumption levels provided by the USDA (38) into grams day^{-1} using serving size weights given in (34). According to these data one ounce equivalent of meat and grains equals 28.3 g (cooked weight) and 22 g, respectively (40); one cup of vegetables, beans and peas, and fruits including juices is equivalent to 123 g, 73 g (cooked weight) and 187 g, respectively.

Food quantities in HAD-2 and HAD-3 (Table 1) are the same as in HAD-1, with the exception that consumption of red and processed meat was further reduced and replaced by increases in beans and peas. According to public health recommendations from the World Cancer Research Fund, red meat consumption should be limited to maximum 300 g of cooked meat week^{-1}, or about 45 g of cooked red meat day^{-1}, and processed meat should be avoided as much as possible (20). In order to meet this recommendation, red and processed meat was reduced to 25 g of cooked meat day^{-1} (36 g day^{-1} of raw meat) in HAD-2 and to zero in HAD-3. The approach for replacing meat with plant-based protein was based on a framework developed by the USDA in which the nutritional interchangeability of plant-based and animal-based protein is estimated. According to this framework one ounce equivalent (one quarter of a cup) of cooked beans and peas (39 g) is nutritionally interchangeable with one ounce equivalent of cooked meat (28 g) (38). Our estimate of the energy content of the changed foods in all four diets (SAD and HADs) was 1000-1130 kcal day^{-1}.

Table 1. Foods in SAD that were changed in HADs.

Food	g capita^{-1} day^{-1} [a]			
	SAD[b]	HAD-1	HAD-2	HAD-3
Total red & processed meat	92	51	25	0
Unprocessed red meat	58	41	25	0
Processed meat	34	10	0	0
Total fruits and vegetables	335	672	707	741
Fruits	74	299	299	299
Fruit juices	60	75	75	75
Vegetables without beans and peas	194	283	283	283
Beans and peas	7	15	50	84
Total grains	167	131	131	131
Whole grains	17	79	79	79
Refined grains	150	52	52	52

[a]Intake levels in cooked weights. Basis for RR calculations.

[b]SAD based on loss adjusted food availability (34).

2.4. Step 2. Changes in Relative Risk of Disease with Changes in Diet

We based the selection of diseases to be included on a literature review of the NCBI Pub Med database in March 2014, using as keywords the selected food groups (e.g. "vegetables") and CHD, hypertension, T2D and a range of cancers. We selected peer reviewed meta-analyses of prospective cohort and randomized controlled trial (RCT) studies, published between 2005 and 2014, that provided RR with 95% confidence intervals (CI).

We judged the evidence for the food-disease relationship as: *convincing* if at least two meta-analyses supported the relationship and all meta-analyses showed significant reductions in disease risk with changes in the food; *probable* if at least one meta-analysis supported the relationship and the most recent one with significant results showed a reduced risk from changes in the food; and *insufficient* otherwise. We chose estimates for reduced risk of disease conservatively by only including RR estimates where the evidence was convincing or probable, which limited the diseases studied to CHD, T2D and CRC (Table S1).

The health effects of changing the diet from SAD to HAD were estimated by calculating a revised RR (RR_{re}) for each food-disease RR, assuming a log-linear dose response relationship between food intake and health outcome, as reported in the meta-analyses (Eq. 1):

$$RRre = RR((x-y)/u) \qquad \text{Eq. 1}$$

Where RR is the original RR obtained from meta-analyses for food f (e.g., processed meat) and disease d (e.g., CHD), x is the level of f in the HAD, y is the level of f in the SAD, and u is the unit increase reported in the meta-analysis identified for disease d. The reductions in RR for a unit change in food consumption were assumed to follow a log-linear dose-response relationship across the whole range of intake levels in the SAD and HADs. When there was more than one meta-analysis RR for a food-disease combination, we used an arithmetic average of the RR_{re}s. For the relationship between whole grains and CHD, no dose-response RR estimates were located; therefore, a RR value based on the comparison of a high vs. low consumption was used to estimate this health effect. This was considered valid due to the large difference in intake levels of whole grains between the SAD and HADs.

We then calculated the combined relative risk (RR_{cd}) of the changes in all of the foods contributing to the RR for each disease by multiplying them, based on the assumption that the effect of each food was independent (Eq. 2) (41).

$$RR_{cd} = RR_{re1} * RR_{re2} * RR_{re3} * \ldots RR_{ref} \qquad \text{Eq. 2}$$

Where RR_{re1}, RR_{re2}, RR_{re3}, and RR_{ref} are the revised RR values for each of the individual foods changed in the diet. Finally, to construct the 95% confidence intervals around the relative risk estimates for the HADs, we conducted a Monte Carlo simulation (42) with 5000 iterations in which the individual RR estimates for each disease were allowed to vary randomly according to a lognormal distribution. The result of a Monte Carlo simulation consists of a number of possible outcomes of the calculation, hence giving a representation of the probability of different results depending on the uncertainty and variation in the input data.

2.5. Step 3. Changes in Health Care Costs from Changes in Disease Prevalence

In calculating the reductions in health care costs due to reductions in the RR of the three NCDs for each of the three HADS, we assumed that reduction in these costs in the US economy for each disease were directly proportional to reduction in the *RRcd* for each disease. The Medical Expenditure Panel Survey (MEPS 2011) is a standard source for health care cost data, but has methodological limitations, so we used the most recent data for expenditures for the three diseases from alternative sources. Expenditures for CHD and CRC were from (43, 44), with spending category percentages assigned by percentages for all heart conditions by MEPS (45). For T2D, which accounts for 90% of all forms of diabetes mellitus, we used (46) for costs and spending category assignments, assuming that category expenditure distribution across both forms of diabetes would remain constant for T2D.

Expenditures for each disease were then adjusted for inflation to 2013US$. Because health care spending in the United States increases at a rate different than the standard rate of inflation, we used the Bureau of Labor Statistics consumer price index for medical care (47) to adjust for inflation of medical expenditures.

2.6. Step 4. Change in GHGE Due to Changes in Health Care Costs (ΔGHGE-H)

In this step we calculated per capita reduction in GHGE for the US population based on established relationships between different types of health care costs and their GHGE in the US. In order to allocate GHGE to health care expenses for each disease, we identified subcategories of expenses, since the types of services vary substantially (e.g., diabetes requires more prescription medications than heart disease). Subcategories were assigned in alignment with the relevant Carnegie-Mellon IO-LCA (35) categories of medical expenditures: hospitals, pharmaceutical manufacturing, physician's offices, and home health services. CRC was assumed to have the same general economic activity in those categories as all forms of cancers, CHD was assumed to have the same percentage of economic activity in categories assigned to all heart conditions, both taken from MEPS category spending assignment percentages. The same method was used for T2D in the broader category of diabetes mellitus.

We then used the IO-LCA to determine an initial GHGE-H for each disease. However, because the IO-LCA uses CO2e based on global warming potential (GWP) values from older IPCC assessments of various GHGs, we adjusted the GWP for CH4 from 21 to the most recent GWP estimates of 34 for a 100-year time frame, and 86 for a 20-year time frame (12). We did this only for CH4 in order to be consistent with estimates of GHGE-F. While the GWP for N2O changed in the most recent IPCC assessment, the difference was much smaller, and would also have been more difficult to adjust for in the estimates of GHGE in the food system.

Since the Carnegie-Mellon assessments were based on 2002 emissions levels, we adjusted for the measured decrease in carbon intensity in the US economy from 2002-2011 (48) and projected this to 2013. We assumed that the decrease in carbon intensity experienced by the US economy was the same as that in the health care sector.

2.7. Step 5. Changes in GHGE in the Food System (Δ GHGE-F)

We quantified the effect on GHGE-F from changes in diet based on estimates of the climate impact of the specific foods included in the diets, provided from LCAs found in the literature. The system boundaries for the GHGE-F were between the primary production and retail stages, i.e. including emissions from primary production, processing, packaging, transportation and distribution through retail, but excluding emissions from post retail transportation, refrigeration, cooking and waste management in households, restaurants and institutions. However, these calculations do include emissions of food subsequently wasted (not consumed) after retail purchase (see section 2.1).

GHGE-F data were collected for 28 different food categories (Tables S2, S3), and the median, lowest, and highest amount of GHGE-F kg^{-1} food available at the retail were determined (Table S4).

In order to address uncertainty and variability in our GHGE-F results due to differences in production systems, regional conditions and methods used by different researchers (9, 49, 50) we conducted a Monte Carlo simulation (42). Because of the limited number and range of estimates in the literature, we used a triangular distribution model for each food category changed in the HADs. The 5000-iteration Monte Carlo simulation then produced a most 303 likely value and confidence intervals for GHGE-F for each diet.

2.8. Step 6. Net Change in GHGE from Dietary Change (ΔGHGE-T)

The net change in GHGE from dietary change was calculated as the total emission reductions in the food system (ΔGHGE-F) and in the health care sector (ΔGHGE-H).

3. Results and Discussion

3.1. Reduction in Relative Risk of Disease and Health Care Costs

HAD-3, in which all red and processed meat was replaced with legumes, provided the greatest reduction in disease prevalence of the HADs. The combined RR (RR_{cd}) for CHD, T2D and CRC for all foods changed in the HADs was reduced by 20-45%, with the largest reduction for CHD, followed by T2D and CRC (Table S5).

For CHD, the increased intake of whole grains had the greatest effect on the RR reduction in the transition from the SAD to the HAD-1, followed by the reduced intake of processed meat and increased intake of fruits and vegetables. Adoption of the HAD-2 and HAD-3 further reduced the RR of CHD by 5% compared to the HAD-1, mainly due to the reduced intake of processed meat. For T2D and CRC, the increased intake of whole grains accounted for the greatest effect on RR reduction in the transition from the SAD to the HAD-1, followed by the reduced intake of processed meat, whereas the effect from reducing unprocessed red meat was more limited. Adoption of HAD-2 and HAD-3 further reduced the RR of T2D and CRC by 6-8% and 5-9%, respectively, due to the reduced intake of processed meat 325 and unprocessed red meat.

The potential annual savings in US health care costs with reduction in prevalence of CHD, T2D and CRC, assuming that the entire US population made a transition from the SAD to the HADs, was US$54, US$65 and US$72 billion yr^{-1} for HAD-1, -2, and -3, or 20-30% of the total expenses for these disease of US$219.5 billion yr^{-1} (19).

Reduction in GHGE

Reduction in GHGE-H (Table 2) was estimated from the reduction in health care expenses resulting from the reduction in RR (Table S5), and thus shows a similar pattern.

Table 2. Reduction in GHGE-H with change to HADs (kg CO_2e $capita^{-1}$ yr^{-1}).

Diet		HAD-1			HAD-2			HAD-3		
CH_4 GWP		21	34	86	21	34	86	21	34	86
NCD	CHD	15	16	20	16	18	22	17	18	23
	T2D	47	50	63	55	58	73	58	62	78
	CRC	3	3	4	3	4	5	4	4	5
	Total	65	69	87	75	80	100	78	84	106

[a]Calculated from revised RR (RR_{re}) estimates for each food-disease relationship (Table S5), the associated reduction in health care costs, and the GHGE generated by these costs.

Reductions in GHGE-F were strongly affected by the incremental reduction in red and processed meat (Table 1), since they account for a large proportion of GHGE and of methane, resulting in reductions from 6% to 70% (Table 3). The relatively small emission reduction potential for HAD-1 was also due to the additional emissions from the large increase in fruits and vegetables required to meet the USDA guidelines relative to the reduction in red and processed meat (Table 1). Greater reductions in GHGE-F were achieved by HAD-2 and -3 due to additional reductions in red and processed meat, with no further increase in fruits and vegetables.

Table 3. GHGE-F for different methane GWP for foods changed in HADs, and reductions from SAD.

Diet	SAD	HAD-1	HAD-2	HAD-3
CH_4 GWP[b]	GHGE-F for foods changed in HADs[a] (kg CO2e capita^{-1} yr^{-1})			
21	1,199	1.130	893	656
34	1,397	1.245	951	655
86	2,168	1.689	1.163	655
	Reduction in GHGE-F with change to HADs from SAD (kg CO_2e capita^{-1} yr^{-1})			
21		68	306	543
34		153	446	742
86		478	1.005	1.512
	Per cent reduction SAD-HADs			
21		6%	26%	45%
34		11%	32%	53%
86		22%	46%	70%

[a]GHGE based on amounts of foods purchased (Table S.1), and GHGE for foods (Table S.2) from the literature (Table S.3).
[b]Different methane GWPs used for calculating CO_2e emissions for beef. Of the seven source studies, six used a GWP of 21 and one used 23.

Our results illustrate how sensitive estimates of GHGE-F from the diet are to the GWP for methane. While the IPCC states that the choice of one GWP time horizon over another is a value judgment with no scientific basis (12), most LCAs use the 100-year GWP. Most LCAs also use pre-2013 IPCC estimates of the 100-year GWP of methane (21 in 1996, 23 in 2001, 25 in 355 2007), while the 2013 estimate is 34 (12) due to increased understanding of methane's absorption 356 properties as well as a more comprehensive integration of indirect carbon cycle feedbacks (51). However, since CH4 has a lifespan of 12.4 years (12) using a 20-year GWP more accurately reflects its climate impact over the critical short term (52) especially for foods with high CH4 emissions, like beef and other ruminant foods. Using the 20-year GWP for CH4 doubled the GHGE reductions for the HADs compared with the current 100-year GWP of 34.

Fig. 1 shows the combined reduction from the food system and the health care sector, in annual capita^{-1} GHGE with a transition from the SAD to the HADs and how these are affected by using different GWP assignments for methane. Reductions from activities in the food system dominated the GHGE reduction potential in all HADs, and accounted for 51-93% of total potential, and increased with increasing GWP values for methane (Table S6). By including emissions in the health care sector, the GHGE mitigation potential increased a maximum of about 90% for HAD-1 using a GWP for CH4 of 21, but decreased with increasing GWP for CH4, and for HAD-2 and -3 because of the greater contribution of CH4 to emissions from the food system.

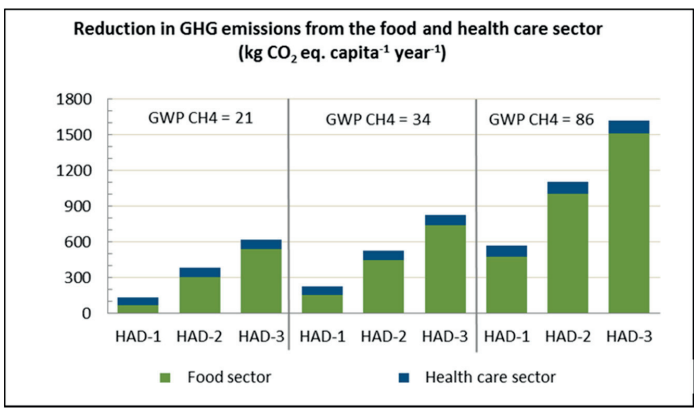

Fig. 1. Reduction in GHGE from the food system and health care sector in HADs for different methane GWPs.

3.3. Limitations and Uncertainty

The results of this study are largely dependent on the quality of underlying data we used for food consumption, food losses, RR, health care expenses, and GHGE. In order to minimize, account for and illustrate the overall uncertainty in our results, we used validated data of high quality, aimed for high transparency in presenting our methods, and estimated the uncertainty of both GHGE and RR with Monte Carlo simulation.

A major assumption in our calculations of the health care costs was that the changes in RR are directly related to health care costs. In reality, diet change would only affect disease prevalence over time via reduction in incidence. Our results should therefore be interpreted as theoretical estimates of the disease prevalence attributable to the HADs over time, or as the health care costs associated with a counterfactual scenario where the HADs have always been consumed.

Of special concern when combining RR estimates as we did, is the risk of double counting the health effects. To minimize the risk of double counting we only used RR estimates coming from meta-analyses that adjusted for influencing confounders, such as other types of food intake, physical activity level and history of disease. Despite these efforts, some risk of double counting remains, meaning that the health effects from the studied dietary change may be overestimated. This is because the RRs are drawn from meta-analyses of prospective cohort studies which have measured usual diet imprecisely, allowing for residual confounding. For increased transparency, we presented the health effects of the HADs both for each dietary factor individually as well as for the combined effect of all dietary changes (Table S5).

There are also some uncertainties associated with the GHGE-H. The IO-LCA methodology has some limitations, particularly its aggregate-based assignment of GHGE for economic activity in a given industry/sector, as opposed to process/product based assignment. Looking at broader aggregates, we adjusted for some factors, such as the decrease in carbon intensity for the overall economy. However, it is not clear that the health care sector would experience the same rate of decrease in carbon intensity. Some components of the LCA, such as pharmaceutical manufacturing, leave out potentially important factors that could add to the GHGE of a given health care expenditure. Our calculations were to some extent based on proxy data as GHGE data were not available for all specific activities.

The uncertainty in the GHGE-F calculation is mainly due to the limited availability of representative LCA data in the literature. Because LCA data were not available for all food items, the GHGE-F for each food category were calculated based on a limited selection of foods for which data were found. Another major challenge is that life cycle data on GHGE-F related to the same food item can vary significantly according to different sources due to differences in production systems, regional conditions and methods used to produce the data (9, 49). For higher precision, regional or country level LCA data are required which are currently lacking.

There are, however, also reasons to believe that the potential GHGE reductions, both in the food system and in the health care sector, from healthier diets were underestimated. First, estimates of reductions in GHGE-F were limited because only a portion of the foods in the SAD was changed in the HADs (red and processed meat, beans and peas, fruit and vegetables, grains), while other foods with high emissions (e.g. dairy and other meats) were not. For example, an estimate for the US of 133.6 Tg CO2e yr^{-1} dairy GHGE in 2008 (53) is equivalent to 439 kg CO2e $capita^{-1}$ yr^{-1}.

Second, due to lack of adequate RR documentation, estimates of reductions in GHGE-H were limited because we did not include many potential diseases (e.g. overweight and obesity, hypertension, stroke and other forms of cancer) associated with the foods we changed in HADs. In addition, there are also potential diet-disease links for foods we did not change in HADs, so that the total estimated health care expenses for the diseases studied accounted for just a small part of the total US health care spending in 2013 of US$2.9 trillion (19). For example, the high sugar intake in SAD is believed to be associated with increased body weight (54), which in turn is a risk factor for CHD, T2D and CRC (55-58) Recent research in Britain has shown that an increased body mass index is associated with increased risk for 17 of 22 cancers, with several at higher risks than CRC (59). The effect of sugar consumption, however, was not included in our study because we found no basis for directly relating it to the risk of these diseases.

3.4. Significance of Results

In accordance with several previous studies (10, 28) our results support the hypothesis that healthier eating habits can contribute to reduced GHGE. However, to our knowledge, this is the first study of the GHGE due to diet to model counter factual diets by making incremental changes based recommendations only for foods for which there are high quality RR data for non-communicable diseases, to estimate reduction in GHGE from the health care sector resulting from a change to healthier diets, and to use the current and more relevant 20-year GWP for methane.

Our results showed that in terms of the total US $capita^{-1}$ GHGE, the maximum potential reduction (with emissions adjusted to CH4 GWP = 86) was less than 3% with HAD-1, up to ~7% with HAD-3 (Table S7), but is

equivalent to removing ~9 to 108 million vehicles from US roads 443 (based on 4.75 MT CO2e vehicle^{-1} yr^{-1} (60)). Compared with previous estimates of GHGE in the food system embodied in the diets of industrialized countries (of which most use a methane GWP of 21), which are in the range of 1400-3200 kg CO2e capita^{-1} yr^{-1} (61), HAD-1 to -3 would reduce emissions ~4-44% (for a CH4 GWP of 21).

However, the real significance of the potential of HADs is in comparison with mitigation targets. For example, the range of mitigation targets potentially achievable for HADs (from HAD-1, CH4 GWP = 21 to HAD-3, CH4 GWP = 86) is 4-37% for the US President's Climate Action Plan goal of a 17% reduction below US 2005 net GHGE levels by 2020, and 27-151% for California's AB 32 goal of reaching 1990 emission levels by 2020 (assuming 2020 emissions 452 equal to 2012 emissions) (Table S7). This suggests that promoting diet change could be an effective and efficient strategy for reaching near-term GHGE mitigation goals, with relatively minimal investment in new research, technology and infrastructure.

Our results show that it is possible to estimate the climate impact of changing to healthier diets with a high level of probability for the small proportion of foods and related diseases for which adequate data exist. However, such a transition, including a sharp reduction in the intake of red and processed meat, would be challenging economically and socially because of the adjustments needed in US food systems and in diets. While major changes are needed in the American diet just to meet the USDA dietary recommendations, the challenges for achieving the greater changes in our healthier and lower GHGE diets would be even greater. Food consumption statistics from the past ten years indicate both positive and negative trends of American dietary patterns. On the one hand, per capita consumption of red meat has declined and whole grain consumption increased, on the other hand, fruit and vegetable consumption has decreased (34). The social and cultural factors that affect these trends need to be further explored in order to implement effective policy instruments, which could range from increasing empirically based and value-based consumer information about the health and environmental impacts of different foods, to government regulations and economic incentives such as Pigouvian taxes.

The positive synergistic effects of diet change could motivate greater efforts in the future to develop and implement such changes, both at the individual and policy levels. Our results suggest that efforts to do so could be cost effective. Given the urgency of mitigating GHGE over the short term, diet change could play a much more prominent role in national, state and local climate policies.

Supporting Information

In the SI we provide detailed information on methods, including sources of data for GHGE and RR; and further detailed discussion.

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Paper IV

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Review

Environmental impact of dietary change: a systematic review

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ABSTRACT

Global food production is identified as a great threat to the environment. In combination with technical advances in agriculture, dietary change is suggested to be necessary to reduce the environmental impact of the food system. In this article a systematic review assessing the environmental impact of dietary change is performed. The aims are to i) evaluate the scientific basis of dietary scenario analysis, ii) estimate the potential environmental effects of dietary change, iii) identify methodological aspects of importance for outcome and iv) identify current gaps in knowledge. The review includes 14 peer-reviewed journal articles assessing the GHG emissions and land use demand of in total 49 dietary scenarios. The results suggest that dietary change, in areas with affluent diet, could play an important role in reaching environmental goals, with up to 50% potential to reduce GHG emissions and land use demand associated with the current diet. The choice of functional unit, system boundaries and methods for scenario development and accounting for uncertainties are methodological aspects identified to have major influence on the quality and results of dietary scenario analysis. Further understanding of dietary change as a measure for more sustainable food systems requires improved knowledge of uncertainty in dietary scenario studies, environmental impact from substitutes and complements to meat and the effect of dietary change in different groups of populations and geographical locations.

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1. Introduction

Global food production occupies more than a third of the world's land surface, accounts for around 30% of total anthropogenic greenhouse gas (GHG) emissions (Garnett, 2011), and is identified as a great threat to the environment (EC, 2006). In combination with technical advances in agriculture, changes towards more sustainable eating patterns are suggested to be necessary to reduce the environmental burden of the food system (Garnett, 2011).

Knowledge of sustainable food consumption is increasing with the growing number of environmental assessments of foods, meals, and complete diets. A method commonly used to assess the impact of different dietary patterns is dietary scenario analysis. This method can be used to estimate the consequences of dietary choices (forecast scenarios), e.g. varying in amount and/or content of food, or the measures required to reach a set target (backcast scenarios) (Alcamo, 2009). By combining food consumption or production data with environmental, economic, or nutritional data of individual food items, the impact of changes in diet can be quantified (Risku-Norja, 2011). The environmental impact of diets is in general quantified based on data provided from life cycle assessments (LCAs). Life cycle assessment is a standardized methodological framework for calculating the environmental impact of a product, process, or service throughout its lifecycle (ISO, 2006a,b).

The methodological approach of dietary scenario analysis can have a decisive effect on the quality and results of the analysis. Due to differences in study design and uncertainty in the methods and data used, results from dietary scenario analyses may differ. Thus, drawing general conclusions on which dietary changes can promote more sustainable food consumption requires that results from several studies are compiled, analyzed, and compared. Previous work provide compilations of LCA data for different foods and food groups (de Vries and de Boer, 2010; Nijdam et al., 2012; Pelletier et al., 2007; Roy et al., 2009) and/or discussions of methodological issues affecting LCAs on food, meals, and diets (Heller et al., 2013; Hospido et al., 2010). However, so far, few quantitative syntheses of the environmental impact of the entire diet have been performed.

This paper provides a systematic review of studies that assess the environmental impact of dietary scenarios. The objectives are to i) evaluate the scientific basis of scenario analyses assessing the

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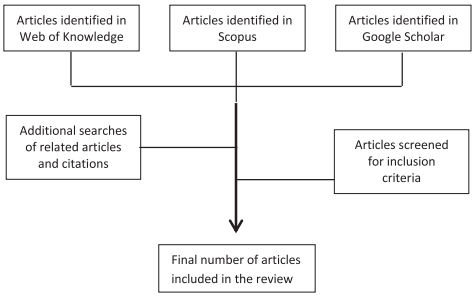

Fig. 1. Literature search and selection of articles in the review.

environmental impact of human dietary change, ii) estimate the potential of reducing GHG emissions and land use demand through dietary change, iii) analyze the study design of existing dietary scenarios to identify methodological aspects of importance for improving quality of research, and iv) identify current gaps in knowledge. This paper can be used as summary of the state of knowledge of sustainable food consumption in 2014 and as a guide for the performance and evaluation of dietary scenario analyses.

2. Method

2.1. Literature search strategy

In order to ensure scientific quality and minimize the risk of bias, the study design and analysis of this review follows the PRISMA Statement protocol (Moher et al., 2009).

The literature search was performed in February 2014 with the use of Web of Knowledge (ISI), Scopus and Google Scholar. To assess the effect of human dietary change on GHG emissions and land use demand, the terms: 'diet' or 'food' and 'scenario' were combined with the terms 'climate' or 'greenhouse gas' or 'land' or 'sustain'. In addition, related and relevant articles found in reference lists were reviewed. Articles included in this review meet the following six inclusion criteria: i) English-language publications; ii) published between 2005 and February 2014; iii) dietary scenario analysis is performed for a complete diet; iv) quantitative estimates of the effect on GHG emissions and/or land use demand of human dietary change are provided; v) published in peer-reviewed scientific journals; vi) results are compared against reference scenarios of current (1990–2010) average food consumption of a population. The inclusion criteria were set to increase the comparability between studies, to capture the effect of dietary change in the current food system and to ensure that included articles were of acceptable quality. Determination of articles that meet the inclusion criteria was made based on information available in titles and abstracts of the articles. In total, 14 articles that fulfilled the inclusion criteria were identified (Fig. 1).

2.2. Synthesis of results

Depending on the dietary composition, scenarios were categorized into healthy diets, diets in which meat is partially replaced by plant-based foods/mixed foods/dairy products, diets in which ruminant meat is replaced by pork and poultry, vegetarian diets, vegan diets and finally diets with balanced energy intake. To only capture the effect associated with changes in dietary composition, scenarios with additional differences, such as in production method (e.g. organic or local food production), were not included in the review.

Diets categorized as healthy are in this paper defined as omnivorous diets based on different dietary guidelines; in diets where meat partially is replaced the proportion of all meat or a specific type of meat is reduced in favor of either plant-based foods, a mix of non-meat food groups or dairy products; in diets where ruminant meat is replaced by pork and poultry all ruminant meat is substituted by monogastric meat; in vegetarian diets all meat is replaced by non-meat food groups; in vegan diets all animal-based products are replaced by plant-based food; and in diets with balanced energy intake the composition of the diet is unchanged and the caloric content is reduced to recommended levels (more details in Table A1, A2).

The potential to reduce environmental impact is reported as the relative and absolute change in GHG emissions, expressed as kg or tons of carbon dioxide equivalents (kg/tCO_2e) per person per year, and land demand, expressed as square meter (m^2) per person per year, compared to the reference scenarios used in the respective studies.

Methodological aspects are assessed based on the approach of scenario development, choice of functional unit, system boundaries, impact categories, and method for uncertainty analysis of used data and results. These aspects are chosen as they are identified as having major impacts on the LCA results on GHG emissions and land use demand of the diet (Cederberg et al., 2011; Heijungs and Huilbregts, 2004; Hospido et al., 2010; Ponsioen and Blonk, 2012; Schau and Fet, 2008). A more in-depth and detailed analysis of the reviewed scenarios is based on: i) the choice of underlying consumption data, ii) the choice of functional unit based on whether nutritional considerations are made in the dietary scenarios, iii) the choice of system boundaries based on the LCA stages

Table 1
Effect of dietary change on GHG emissions and land use demand.

Scenario	Reduction of GHG emissions			Reduction of land use demand		
	(%)[a]	(kg CO_2e/yr)[c]	(n)	(%)[b]	(m^2/yr)[c]	(n)
Vegan diet	25–55	760 (520–1090)	6	50–60	970 (690–1160)	3
Vegetarian diet	20–35	540 (110–1110)	7	30–50	790 (570–1010)	2
Ruminant meat replaced by monogastric meat	20–35	560	2	–	–	–
Meat partially replaced by plant-based food	+5–0	+20 (+40–0)	2	15	220	1
Meat partially replaced by dairy products	0–5	40 (30–50)	2	–	–	–
Meat partially replaced by mixed food	0–5	80 (40–110)	2	–	–	–
Balanced energy intake	0–10	100 (40–160)	2	–	–	–
Healthy diet	0–35	210 (+40–490)	14	15–50	590 (310–940)	6

[a] Effect of dietary change on GHG emissions from the diet, in % of reduction in GHG emissions of current average diet.
[b] Effect of dietary change on demand of land, in % of reduction in total demand of agriculture land of the average diet.
[c] Average effect (minimum change – maximum change), n = number of scenarios. "+" indicate an increase in GHG emission alt. land use demand.

Table 2
Summary of methodological choices and study design in the articles reviewed.

Article	Uncertainty analysis	Nutritional considerations in functional unit	System boundary	Impact category		
				Environmental indicator	Including emissions from dLUC/iLUC	Specification of land use
van Dooren et al. (2014)	No	[a]Rec. intake	Cultivation – farm gate	GWP, LU	No/No	No
Hoolohan et al. (2013)	No	Energy	Cultivation – retail	GWP	Yes/No	
Saxe et al. (2013)	No	Energy, protein	Cultivation – retail	GWP	No/No	
Temme et al. (2013)	No	Mass		LU		No
Aston et al. (2012)	No	[b]Rec. intake	Cultivation – retail	GWP	No/No	
Berners-Lee et al. (2012)	No	Energy	Cultivation – retail	GWP	No/No	
Macdiarmid et al. (2012)	No	[a]Rec. intake	Cultivation – retail	GWP	No/No	
Meier and Christen (2012)	No	Energy	Cultivation – retail	GWP, LU	Yes/No	Yes[e]
Vieux et al. (2012)	Yes[d]	Energy	Cultivation – retail	GWP	No/No	
Fazeni and Steinmüller (2011)	No	[a]Rec. intake	Cultivation – farm gate	GWP	No/No	
Tukker et al. (2011)	No	Energy, protein, fat	Cultivation–waste disposal	GWP	No/No	
Arnoult et al. (2010)	No	Energy		LU		No
Pathak et al. (2010)	No	Energy	Cultivation – consumer[c]	GWP	No/No	
Risku-Norja et al. (2009)	No	Energy	Cultivation – farm gate	GWP	No/No	

[a] Healthy diet that meets energy and nutrient recommendations.
[b] In the studied scenario the proportions of vegetarians in the population doubled and remaining population adopted a climate friendly diet, low in red and processed meat.
[c] Including food preparation in household.
[d] Monte Carlo Analysis.
[e] Distinction between cropland and grassland, domestic and abroad land.

accounted for, iv) the choice of impact category based on whether differentiation between land use types is made and whether GHG emissions from direct and/or indirect land use change are accounted for, and v) the handling of uncertainty based on whether an uncertainty analysis of the results is made or not. Specification of land use types is considered to be made if quantitative data of the land demand for more than one land type is given. If no information is given on whether emissions from direct or indirect land use change are included in quantifications of GHG emissions, it is assumed that these emissions are not accounted for.

3. Results

3.1. Located literature

In total, 14 articles that fulfil the inclusion criteria were identified in this assessment. Out of these 14 articles, two investigated both the effect on GHG emissions and land use demand, two the impact of land use only, and ten the impact of GHG emissions only. Five articles were published between 2009 and 2011, and nine articles between 2012 and Feb 2014 (Tables 1 and 2). Additional information on the study design and scenarios in the reviewed articles is found in Tables A1 and A2.

3.2. Potential to reduce GHG emissions

The impact of dietary change on GHG emissions from diet is summarized in Table 1 and Fig. 2. Completely avoiding all animal-based products (vegan) provides the largest potential for reducing GHG emissions from the diet, followed by scenarios of avoiding all meat (vegetarian), replacing ruminant meat with pork and poultry and eating a healthier diet.

3.3. Potential to reduce land use demand

The impact of dietary change on land use demand from the diet is summarized in Table 2 and Fig. 3. According to the results, a change to vegan or vegetarian diets has the largest potential to reduce the demand for agriculture land, followed by changing to a healthier diet and diets in which meat is partially replaced by plant-based food.

3.4. Methodological aspects of importance

This section presents an assessment of key methodological aspects in dietary scenario analyses. Table 2 provides an overview of methodological choices made in the articles reviewed.

3.4.1. Scenario development

Dietary scenarios can be developed based on registered or hypothetical diets. Registered dietary data provides information of consumption patterns of individuals or groups of populations. Individual based data has the advantage that dietary patterns can be linked to personal characteristics (e.g. gender, age, nationality, socioeconomic group), whereas average consumption data may hide variations between different population groups (Hallström and Börjesson, 2013). Compared to hypothetical dietary scenarios, scenarios based on registered consumption data have the advantage of being realistic, not only in theory, but in practice. A disadvantage of self-registered data is that people tend to change their food habits when the consumption is recorded and/or misreport the consumption (Ferro-Luzzi, 2003). Hypothetical diets have the advantage that any consumption patterns, realistic or not, can be investigated. By using future scenarios, the effect of technology and product development, population growth and other influencing parameters can also be investigated.

In the articles reviewed the reference scenarios are based on average per capita consumption data of the studied population, i.e. current consumption patterns. The exception is the study by Pathak et al. (2010), in which the reference diet is based on a hypothetical well-balanced diet consisting of common Indian foods. The reference diets are thereafter modified in order to study the environmental impact of different hypothetical changes in the diet. In some of the articles reviewed (Aston et al., 2012; Berners-Lee et al., 2012; Vieux et al., 2012), not only the reference scenarios but also the studied dietary scenarios are based on registered consumption data, e.g. self-selected diets. The population studied is in general defined by its nationality. In all articles reviewed except for Pathak et al. (2010), the effect of dietary change is studied in European populations characterized by having affluent diets. In a few articles, the reference diets reflect consumption patterns of particular groups of the population, for example, women (Macdiarmid et al., 2012; Temme et al., 2013; van Dooren et al., 2014).

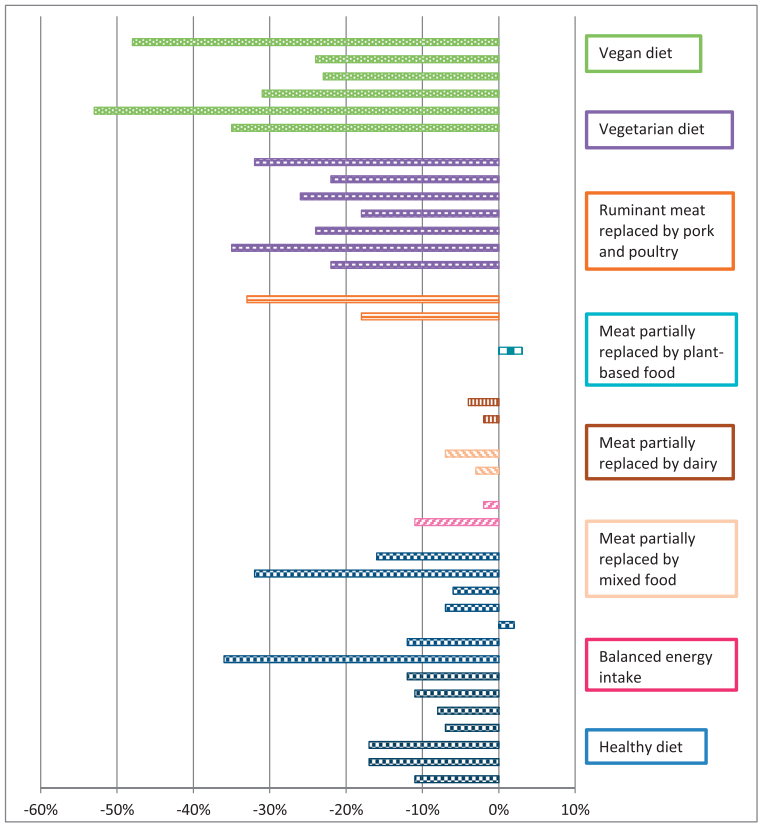

Fig. 2. Impact of dietary change on GHG emissions from the diet, in % of relative change in GHG emissions compared to the reference scenarios. Presented data are based on the results from 12 articles, for references and detailed information about the specific diets see Table A1.

3.4.2. Uncertainty analysis

Since scenarios are used to study a possible future course, they have intrinsic uncertainties. Besides these inherent uncertainties, uncertainty factors in scenario analysis can broadly be categorized into data uncertainties and uncertainties due to methodological choices. Data uncertainties can be due to the use of inaccurate or unrepresentative data or that data is unavailable or missing (Björklund, 2002; Huijbregts, 2002). When it comes to food, consumption and production data as well as, environmental and nutritional data are subject to uncertainty. Emissions arising from biological processes such as enteric fermentation in ruminants and nitrous oxide emissions from soils are examples of uncertain data, as these emissions often vary and are difficult to measure and model (Eggleston et al., 2006). Uncertainty and variability in the nutrient content of food items add to the overall uncertainty in nutrient calculation. There is also a recurring problem in nutrient calculation, that of how to handle uncertainty in food consumption data.

The methodological approach of scenario analysis can have a decisive effect on its quality and final outcome. Methodological choices of importance in dietary scenario analysis are, for instance, the choice of functional unit (3.4.3), system boundaries (3.4.4), assumptions on direct and indirect land use change (dLUC and iLUC) (3.4.5) and the categorization of land use (3.4.6). Variability in results can also be due to geographical, temporal or technological variability in input data (Björklund, 2002; Huijbregts, 2002).

Out of the 14 articles reviewed, Vieux et al. (2012) is the only one to perform an uncertainty analysis of the results.

3.4.3. Nutritional considerations in functional unit

In LCA, the functional unit (FU) is the reference base which describes the function of the studied object, thus enabling comparison between different systems (ISO, 2006b). When it comes to food, the environmental impact is commonly expressed in relation to the quantity or volume consumed or produced (Schau and Fet, 2008), for example per kilogram, liter, serving portion or meal. However, the FU can also be based on the food's economic value (e.g. profit or price) or demand of resources (e.g. area). As meals in general are not representative for the average food consumption, assessments of complete diets are recommended to account for

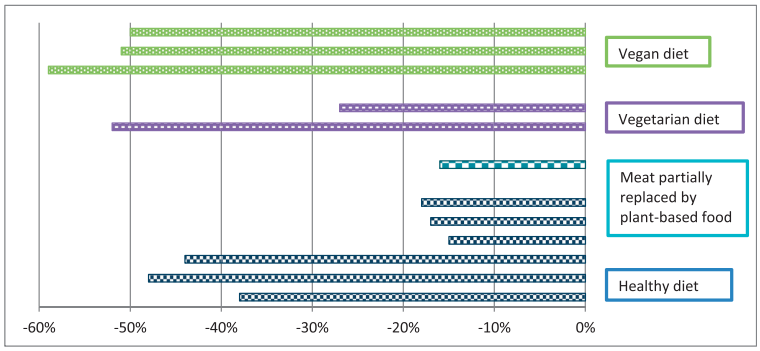

Fig. 3. Impact of dietary change on current demand of land from the diet, in % of relative change of land demand compared to the reference scenarios. Presented data are based on results from four articles, for references and detailed information about the specific individual diets see Table A2.

nutritional aspects in environmental assessments of dietary change (van Kernebeek et al., 2014). In order to account for the quality of food, it has become increasingly common to use FUs which relate to the nutritional content, for example to the energy or protein content, or by using nutritional indices and recommended intake levels of nutrients (Schau and Fet, 2008). In comparative studies of food and diets the choice of FU can have a large effect on the outcome. When mass- or volume-based FUs are used the density (i.e. the water content) of the food plays an important role. It is therefore essential to specify whether the FU refers to fresh, dried or cooked weight of the food product (Hallström and Börjesson, 2013). In comparisons between plant-based and animal-based foods, the environmental impact is often expressed per kilogram of food, an approach that has been criticized for favoring plant-based foods because they generally have higher water content than animal-based products. Functional units that relate to the energy or nutritional content are proposed for more fair comparisons (Vieux et al., 2012). However, in order to not promote excessive consumption of nutrients, it is suggested that nutritional based FUs should be related to recommended intake levels of energy and/or nutrients (van Kernebeek et al., 2014). To present the results using several parallel and complementary FUs is a preferable way to illustrate the results from different perspectives.

In the reviewed articles, the most common approach to account for the nutritional value of the diet is to use iso-caloric substitution, i.e. that all dietary scenarios contain the same energy content. In addition, some articles design the scenarios so that the studied diets are comparable for other nutrients. Several of the articles also use additional criteria, for example, that the dietary scenarios should be in line with healthy recommendations. In the paper by Temme et al. (2013) the FU only relates to the foods weight. This makes it difficult to evaluate the comparability of nutrient content in the different dietary scenarios. This study, however, quantifies the intake of iron and saturated fatty acids from all scenarios studied.

3.4.4. System boundaries

In LCA the system boundaries define which processes are included and excluded. Boundaries can also be set against the life cycles of other products, to define the natural system and the geographical and temporal coverage of the study (ISO, 2006b). Ideally, LCAs should include all phases of the products life cycle, from the cradle to the grave. When it comes to food this means that all activities between the primary production of raw materials and the waste disposal are accounted for. In practice, however, it is common to exclude activities deemed to have a negligible impact on the results. The agricultural production in general constitutes the largest share of the total environmental impact of food products, many LCAs on food thereby only include activities up to the farm gate (Schau and Fet, 2008; Sonesson et al., 2010). However, for some foods (e.g. foods that emit small amounts of GHG in the production), it may have a substantial effect whether the environmental impact is calculated up to the farm gate, retail or final consumption. When comparing different foods or diets it is therefore important that the system boundaries in the studied systems are comparable.

Losses and waste occurring between production and consumption may also be of importance in the choice of system boundaries. Due to losses along the production and distribution chain, there might be a difference of a factor two or more between the amount (based on weight) of food available for consumption and the amount actually eaten (Hallström and Börjesson, 2013). It is therefore important to adjust consumption data if they are to be used to calculate the environmental impact from the diet, and the opposite if production data are used to calculate, for example, the nutrient intake from the diet. To make data sources comparable, ideally, all processes which contribute to weight losses between production and consumption, e.g. food loss and waste at all stages, deductions for inedible parts of the food (e.g. bones, peels etc.) and weight losses in cooking, should be accounted for. If self-reported consumption data are used it might also be relevant to consider the effect of underreporting (Hallström, 2013).

Only two of the reviewed articles (Pathak et al., 2010; Tukker et al., 2011) use system boundaries including the production system from primary production to consumer phase. Tukker et al. (2011) also includes emissions coming from the waste disposal. The most common procedure is to set the system boundaries to include emissions produced up to the distribution of the food, e.g. to the stage of retail. In three of the articles (Fazeni and Steinmüller, 2011; Risku-Norja et al., 2009; van Dooren et al., 2014) quantifications of GHG emissions from the diet are limited to emissions taking place in the agriculture phase, e.g. up to the farm gate.

Losses and waste along the food chain are accounted for in various ways in the reviewed articles, for example, by using LCA data that includes emissions from all stages up to the retail or consumer level in which emissions from food wasted in upstream

processes are added to the remaining food that becomes available for the consumers. The difference between per capita agricultural supply data and consumption data of actual intake levels is often used as an estimate of the amount of food that is lost and wasted during the lifecycle (Berners-Lee et al., 2012; Hoolohan et al., 2013).

3.4.5. Emissions from land use change

In the IPCC reports, GHG emissions from transportation and processing of food and inputs used in agriculture (e.g. fertilizers, pesticides) are categorized to emissions from the transport and industrial sector and emissions from deforestation (and other changes in land use) are categorized as emissions from land use change. According to the IPCC, the agriculture sector is responsible for 10–12% of the global anthropogenic GHG emissions (Smith et al., 2007). However, if emissions from the entire life cycle of food are accounted for (including emissions from land use change) the agri-food sector is responsible for about one third the global anthropogenic GHG emissions (Garnett, 2011). Emissions from land use change are the main reason to why the two results differ. Until recently, LCAs have only included direct GHG emissions from the life cycle of food. However, over the past decade it has been found that land use change is a major source of GHG emissions from agriculture. Expanding agricultural land is estimated to be the responsible driver for 80% of global deforestation (Kissinger et al., 2012). In addition to GHG emissions from direct effects of land use change, a discussion about how to account for emissions coming from indirect effects of changes in land use has emerged (Havlík et al., 2011). Emissions from iLUC have so far mainly been debated in association with production of biofuels, but are relevant in the production of all agricultural products, including food.

In the articles reviewed two articles (Hoolohan et al., 2013; Meier and Christen, 2012) account for GHG emissions from dLUC and none for emissions from iLUC.

3.4.6. Specification of land use

In LCA the land use demand of a food product is quantified as the area (e.g. m^2 or hectares) of land required to produce one functional unit of the specific food. However, the effect of using land for food production is largely dependent on the type of land used, the previous use of the land and the geographical location of the land. When assessing the effects of land use and availability of land it may therefore be of advantage to distinguish between different types of land.

Of the four articles reviewed which include land use demand, only Meier and Christen (2012) report the demand for cropland and pasture land separately. In addition, this article makes a distinction between domestic and imported land which is not done in the other articles.

4. Discussion

This review is, to our knowledge, one of the first to systematically assess the current state of knowledge of the environmental impact, expressed as changes in GHG emissions and land use demand, of dietary change. The review includes peer-reviewed journal articles published over the past ten years.

4.1. Scientific basis

This review located 14 articles that met the defined inclusion criteria. In accordance with what has been shown in Heller et al. (2013), this study illustrates that LCAs of food is an expanding research field. Nine of the articles were published during just the two last years. Although there are still gaps in knowledge, the increased number of publications in this area has significantly contributed to a better understanding of sustainable production and consumption of food.

4.2. Potential to reduce GHG emissions

The results show that the potential to reduce GHG emissions from food consumption through dietary change can be substantial in regions with affluent diet. The reduction potential seems mainly to depend on the amount and type of meat and animal products included in the diet. Diets in which all animal products (vegan), meat (vegetarian) or ruminant meat are removed have the lowest GHG emissions. However, a healthier diet including meat can, according to the results, reduce the GHG emissions of the diet by up to 35%. The impact, however, largely depends on what is considered to be a healthy diet, and in five of the 14 healthy dietary scenarios the reduction potential is less than 10%. The amount of red meat, and especially ruminant meat allowed seem to be a decisive parameter for the climate impact of healthy diets. The difference in climate impact between different types of meat is also demonstrated by the results from the scenarios studying the effect of reduced or changed meat consumption. Whereas replacement of all ruminant meat by poultry and pork can reduce the GHG emission by up to 35%, moderate reduction (up to 20%) in total meat intake (including white meat) seems to have a negligible effect. In addition, the climate impact of the diet is, to a large extent, dependent on which foods that replace the meat, therefore, consumption of meat substitutes with high climate impact, such as cheese and air transported fruit and vegetables, should be restricted (Carlsson-Kanyama and Gonzalez, 2009). Only eating necessary amounts of food has been identified as another priority measure to reduce GHG emissions from the diet (Garnett, 2011) that also would be beneficial for health. Balancing energy intake and expenditure can, according to the results in this review, reduce the climate impact of the diet by 0–10%, depending on the assumed energy requirements.

The GHG emissions from the reference scenarios, i.e. the current average diet in the studied populations, ranged from 0.9 to 1.7 and 1.4 to 3.2 tons (0.4 tons for Indian diet) of CO_2e per capita per year in the studies accounting for emissions up to farm gate and retail, respectively. The annual GHG emissions for the average EU citizen are around nine tCO_2e (EEA, 2012), which means that food consumption is responsible for about 15–35% of the total climate impact. Based on these figures, the potential to reduce the total per capita GHG emissions through dietary change is about 4–20% for a transition to a vegan diet, and up to 12% by a transition to either a vegetarian diet, a diet in which ruminant meat has been substituted by monogastric meat or a healthier diet with restricted intake of red and ruminant meat.

4.3. Potential to reduce land use demand

Also the potential to reduce the land use demand from the diet through dietary change may be considerable. It should, however, be kept in mind that the impact on land use demand in this paper is based on only four articles. The potential to reduce the land demand of the diet appears to be largely dependent on the amount of ruminant meat consumed. Substituting all animal products with plant-based food can, according to the results, reduce the land demand from the diet by up to 60%. According to Audsley et al. (2010) a replacement of 75% of the ruminant meat with pork and poultry can reduce the land demand by 40%. Replacing half of the consumption of pork and poultry with plant-based food would, on the contrary, only reduce the land demand by 5%. A healthy diet including meat may therefore also have a large potential to free land, if the consumption of red meat is limited. Diets including

ruminant meat have previously been suggested to increase the number of people that can be fed from the same land area compared to vegan diets, up to the point that land limited to pasture and perennial forages has been fully utilized (Peters et al., 2007). However, maximum output of food is necessarily not always the primary objective, given that released land also can be used for bioenergy production, for example (Fazeni and Steinmüller, 2011). Either way, as will be discussed further, differentiation between types of land is essential to fully understand the effect from diet on land use demand.

The land demand of the reference dietary scenarios ranged from 1400 to 2100 m^2 per capita. This can be compared to the current global per capita availability of agriculture land which is about 7000 m^2 (divided between approximately 30% arable land and 70% pasture) if global croplands are assumed to be distributed equally across the population.

In studies assessing the long-term effect of dietary change, global average per capita land demand in 2030 and 2050 is projected to about 5000 m^2 (Powell and Lenton, 2012; Stehfest et al., 2009; Wirsenius et al., 2010). A transition to a healthier diet with restricted intake of red and ruminant meat is estimated to reduce the future per capita land demand to 2200–3500 m^2 (of which 1600–1700 m^2 consists of arable land) and a vegetarian or vegan diet to 2100 and 1600 m^2 (of which 1600 m^2 consists of arable land), respectively (Stehfest et al., 2009). The results from these studies further indicate that both improvements in the production and changes in consumption will be needed to prevent future increase in demand of agriculture land (Powell and Lenton, 2012).

4.4. Identified research gaps

The assessment of key methodological aspects in dietary scenario analysis showed that these can be performed in various ways and that the choice of method can affect the scientific quality and outcome of the study. This section describes identified gaps of knowledge and suggests ways for further improving the understanding of sustainable food production and consumption.

4.4.1. Differentiation on individual, regional and social level

The general approach to study the impact of dietary choices by using scenario analysis is to use a reference scenario based on the average per capita consumption in the population studied. Since consumption patterns and nutritional requirements differ depending on, for example, gender, age and physical activity level, it would be interesting to see more research on specific groups in the populations. It is also noteworthy that all articles reviewed, except one (Pathak et al., 2010), study the impact of dietary change in European countries/regions characterized by having affluent diets. To understand the impact of dietary change in a broader and global perspective similar studies are required in countries/regions with different habits, culture and conditions.

4.4.2. Differentiation of plant-based scenarios

Previous findings suggest that plant-based food consumption based on self-selected diets tend to have a higher climate impact compared to plant-based consumption in hypothetical scenarios (Vieux et al., 2012). In hypothetical plant-based dietary scenarios meat is often replaced by unprocessed foods such as pulses, cereals, breads, salads, vegetables, fruit, nuts and seeds. Vegetarian diets are in general characterized by a higher proportion of these food groups (Craig, 2010; Key et al., 2006), however, processed plant-based meat substitutes (e.g. quorn, tofu, tzai, and tempeh) represent an increasingly important component of modern plant-based diets. The environmental impact of such processed vegetarian meat substitutes has so far only been investigated in a limited number of studies (Blonk et al., 2008; Davis et al., 2010; Finnigan, 2010a; Finnigan et al., 2010b; Leuenberger et al., 2010; Nijdam et al., 2012; Nonhebel and Raats, 2007; Xueqin and Ierland, 2004). The results indicate that these products may have relatively high energy demands due to the higher degree of processing but a lower climate and overall environmental impact, in comparison to most types of meat. Few of the reviewed articles specify that these types of processed meat substitutes are included in the dietary scenarios. The potential and limitations for reducing the environmental impact of the diet through increased consumption of this group of products requires further analysis.

4.4.3. Differentiation of agricultural land

Current global food supply is mainly dependent on cultivated land (Johansson, 2005) why the pressure on agricultural land is especially intense on cropland. Previous studies indicate that dietary change, in particular, has the potential to free pasture land (Hallström et al., 2011). Of the land released through reductions and changes in meat consumption, for example, only 5–10% is estimated to consist of cropland (Hallström, 2013). Others suggest that replacing beef with pork and poultry even may increase the total demand of cropland and the land use competition between humans and animals (Audsley et al., 2010; de Vries and de Boer, 2010). A net gain in cropland is also not obvious if consumption of dairy products is replaced by plant-based food or when monogastric meat is replaced by processed vegetarian meat substitutes (Audsley et al., 2010; Stehfest et al., 2009).

If the distinction is not made between different types of land, there is thus a risk of overestimating the land areas for agriculture that can be released by reducing ruminant meat consumption as only a limited share of pasture land is suitable for cultivation. To avoid a situation where demand for agriculture land is exported to other countries where it might increase the risk for deforestation and other negative impacts connected to increased land use pressure, it may also be of interest to in a greater extent distinguish between domestic and foreign land use in dietary scenario analysis.

4.4.4. Accounting for uncertainty

Despite the knowledge of the uncertainty related to dietary scenarios the environmental impact of dietary scenarios is in general reported in absolute numbers without standard deviations. This is questionable as it makes it difficult to evaluate the reliability of the results. According to the ISO standard, the interpretation phase in LCAs should include an evaluation of the completeness, sensitivity and compliance of the analysis (ISO, 2006b). This is required in order to help the reader to determine what conclusions can be drawn from the results and would be useful also in dietary scenario analysis.

The majority of articles which quantify GHG emissions from the diet exclude emissions coming from dLUC. The exceptions are Meier and Christen (2012) and Hoolohan et al. (2013) who include GHG emissions from deforestation resulting from livestock supply chains. None of the articles include the effect of GHG emissions from iLUC. This is not surprising as the knowledge of emissions from iLUC is quite new, the availability of LCA data which includes these effects is still poor and methods used to account for iLUC are inconsistent (van Middelaar et al., 2013). As GHG emissions from direct and indirect LUC has been suggested to have substantial impact on the climate impact from agricultural products (Cederberg et al., 2011; Ponsioen and Blonk, 2012; Schmidinger and Stehfest, 2012), these aspects will be important to consider in future studies. This in turn leads to an increased need of taking into account specific local and regional production conditions.

4.5. Strengths and limitations

To minimize bias, this review includes only peer-reviewed journal articles selected by the use of predefined inclusion criteria. The aim has been to assess the articles with a high level of objectivity and transparency. A limitation with the study is that the review only located a small number of articles which met the inclusion criteria. The limited number of articles can partly be explained by the novelty of the research field but is also due to the narrow inclusion criteria which excluded several relevant articles (e.g. Audsley et al., 2010; Eshel and Martin, 2006; Kastner, 2012; Macdiarmid et al., 2011; Marlow et al., 2009; Westhoek et al., 2011; Collins and Fairchild, 2007). Most of these studies were excluded for not being published in peer-reviewed journals or for using methodological approaches that did not allow for direct comparison between studies, for example, if only parts of the diet is analyzed or alternative quantitative measures such as the ecological footprint are used. Relevant publications and data would perhaps also be found in non-English publications.

The climate impact of diet is quantified based on the global warming potential (GWP) of GHG emissions. In the fifth IPCC assessment report published in 2013 (Myhre et al., 2013) the GWP of methane over a time horizon of 100 years was increased from previously 25 to 34 kg CO_2e per kilograms of emissions. This review includes articles published before the new IPCC report was published and therefore use the lower GWP for methane in their calculations. This means that the climate impact of diets containing ruminant meat is likely to be higher than the results shown by this review.

In this paper the environmental impact of dietary scenarios is assessed only based on the emissions of GHG and demand of agriculture land. These aspects can often, but not always, serve as indicators of other environmental impact categories such as eutrophication, acidification and loss of biodiversity (Rockström et al., 2009; Röös et al., 2013; van Dooren et al., 2014). However, for a full assessment of the environmental impact of the diet other environmental impact categories also have to be included. Within the wide concept of sustainable food production and consumption also several other aspects, of ecological, social and economic dimensions are included (FAO, 2013). These aspects, however, go beyond the scope of this paper. In future studies interdisciplinary and holistic assessments of the diet which include more sustainability aspects are thus required. The current studies exploring the environmental impacts of diets are based on attributional LCAs. This means that environmental impacts from activities other than in the food chain are not accounted for although they can be influenced by dietary change. Examples are non-food functions from meat production, such as manure and leather (van Kernebeek et al., 2014) and in a vegan scenario materials for e.g. shoes and bags have to be manufactured with resulting emissions of GHGs. Another example is emissions of GHGs from the health sector that could be diminished with a more healthy diet. Clearly, dietary change impacts assessments require a broad system perspective including consequential LCA's.

5. Conclusions

This systematic review evaluates the potential of dietary change as a measure for more sustainable food systems. The results suggest that dietary change, in areas with affluent diet, can play an important role in reaching environmental goals, with up to 50% potential to reduce GHG emissions and land demand of the current diet. The reduction potential mainly depends on the amount and type of meat included in the diet but also on the environmental performance of the food substituting meat. The choice of functional unit, system boundaries and methods for scenario development and accounting for uncertainties are methodological aspects identified to have major influence on the quality and results of dietary system scenarios. In future research interdisciplinary and holistic assessments of the diet including more sustainability aspects are required. Improved knowledge is also needed on the uncertainty in dietary scenario studies, the environmental impact of substitutes and complements to meat, and the effect of dietary change in different groups of populations and geographical regions.

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Appendix

Table A1
Effect on GHG emissions of dietary change.

Reference, description	Scenario	GHG emissions (tCO_2e/person, year)	Change compared to reference scenario
van Dooren et al., 2014	Ref Average Dutch diet of women in 1998	1.5	
Cultivation-farm gate	Healthy diet according to Dutch Dietary Guidelines	1.3	−11%
	Healthy diet with 50% of meat replaced by plant-based food[a]	1.2	−17%
	Healthy Mediterranean diet[b]	1.2	−17%
	Vegetarian diet	1.2	−22%
	Vegan diet	1.0	−35%
Hoolohan et al., 2013	Ref: Average UK consumption in 2010	3.2	
Cultivation-retail	Healthy vegetarian diet	2.1	−35%
	Ruminant meat replaced by pork and poultry	2.6	−18%
Saxe et al., 2013	Ref: Average Danish diet in 2005	1.9	
Cultivation-retail	Healthy diet according to NNR[c]	1.8	−8%
	Healthy, diet according to OPUS[d]	1.8	−7%
Meier and Christen, 2012	Ref: Average German diet in 2006	2.1	
Cultivation-retail	Healthy diet according to D-A-C-H[e]	1.8	−11%
	Healthy diet according to UGB[f]	1.8	−12%
	Lacto-ovo vegetarian diet	1.6	−24%
	Vegan diet	1.0	−53%

Table A1 (*continued*)

Reference, description	Scenario	GHG emissions (tCO_2e/person, year)	Change compared to reference scenario
Berners-Lee et al., 2012	Ref: Average UK diet in 2009	2.7	
Cultivation-retail	Meat replaced by dairy products	2.1	−22%
	Self-reported vegetarian diet[g]	2.2	−18%
	Healthy vegetarian diet[h]	2.0	−26%
	Vegan diet, no health considerations	1.9	−31%
	Self-reported vegan diet[g]	2.1	−23%
	Healthy vegan diet	2.0	−24%
Vieux et al., 2012	Ref: Average French diet in 2006–2007	1.5	
Cultivation-retail	Balancing energy intake and expenditure, assuming low physical activity	1.3	−11%
	Balancing energy intake and expenditure, assuming moderate physical activity	1.5	−2%
	−20% of meat intake (min 50 g meat) replaced by self-selected[i] fruit and vegetables	1.5	0%
	−20% of meat intake (min 50 g meat) replaced by self-selected[i] dairy products	1.5	−2%
	−20% of meat intake (min 50 g meat) replaced by self-selected[i] mixed foods	1.5	−3%
	Reduced meat intake to 50 g/d and removal of deli meat replaced by self-selected[i] fruit and vegetables	1.5	+3%
	Reduced meat intake to 50 g/d and removal of deli meat replaced by self-selected[i] dairy products	1.4	−4%
	Reduced meat intake to 50 g/d and removal of deli meat replaced by self-selected[i] mixed foods	1.4	−7%
Macdiarmid et al., 2012	Ref: Average UK diet of women in 1990	1.4	
Cultivation-retail	Healthy and sustainable diet[j]	0.9	−36%
Aston et al., 2012	Ref: Average UK diet in 2000/2001	1.4	
Cultivation-retail	Doubled proportion of vegetarians, low consumption of red and processed meat in remaining population[k]	1.3	−12%
Tukker et al., 2011	Ref: Average diet in five EU27 regions in 2003	2.6	
Cultivation-waste disposal	Healthy diet based on European dietary guidelines[l]	2.6	+2%
	Healthy diet with less than 300 g red meat per week and avoidance of processed meat	2.4	−7%
	Mediterranean diet with reduced intake of red meat	2.4	−6%
Fazeni and Steinmüller., 2011	Ref: Average Austrian diet in 2001–2006	0.9	
Cultivation-farm gate	Healthy diet[m]	0.6	−32%
Pathak et al., 2010	Ref: Hypothetical Indian non-vegetarian diet[p]	0.4	
Cultivation-consumer	Balanced Indian vegetarian diet[n]	0.2	−32%
Risku-Norja et al., 2009	Ref: Average Finnish diet in 2007	1.7	
Cultivation-farm gate	Healthy diet[o]	1.4	−16%
	No dairy products, ruminant meat replaced by pork and poultry	1.1	−33%
	Vegan diet	0.9	−48%

[a] Semi-vegetarian diet, an average between a healthy omnivorous and a vegetarian diet.
[b] Diet lower in meat, high in fish, fruits and vegetables, and plant oils instead of animal fats.
[c] Healthy diet according to Nordic Nutrition Recommendations.
[d] Healthy diet inspired of Nordic diet from old days, with increased intake of roots, berries, nuts, fish and whole grain products and lower content of animal-based food.
[e] Official dietary recommendations of the German Nutrition Society.
[f] Dietary recommendations by the Federation for Independent Health Consultation.
[g] Self-selected diets based on food choices of American vegetarians and vegans.
[h] Meat is replaced by plant-based food categories considered to be healthy (e.g. pastas, rice, pulses, cereals, breads, salads, vegetables, fruit, nuts, seeds), dairy consumption remains unchanged.
[i] Self-selected diets based on food choices of a sample of adults living in France.
[j] Diet fulfilled nutrient requirement of fertile women, contained 190 g of cooked red meat per week and 555 g fruit and vegetables per day, and was created to minimize food waste and GHG emissions.
[k] Consumption of red and processed meat adapted to the dietary pattern of the lowest fifth of population. Average intake of red and processed meat reduced from 91 to 52 g/d in men and 54 to 30 g/d in women.
[l] Dietary recommendations based on Health Council of the Netherlands and the WHO/FAO.
[m] Based on DGE recommendations including reduced meat intake (−60%, all meat types decrease to the same extent), increased intake of fruit, vegetables and cereals and reduced intake of fish and sugar.
[n] Balanced non vegetarian diet based on foods which commonly form a part of the Indian diet.
[o] Balanced lacto-vegetarian diet based on foods which commonly form a part of the Indian diet.
[p] Based on national health impact dietary recommendations, including increased share of plant based food, reduced share of animal based food and 60% share of present milk consumption.

Table A2
Effect on current land use of dietary change.

Reference, description	Scenario	Total land use demand (m^2/person, year)	Change compared to reference scenario
van Dooren et al., 2014	Ref: Average Dutch diet of women in 1998	1900	
	Healthy diet according to Dutch Dietary Guidelines	1200	−38%
	Healthy diet with 50% of meat replaced by plant-based food[a]	1100	−44%
	Healthy Mediterranean diet[b]	1000	−48%
	Vegetarian diet	900	−52%
	Vegan diet	800	−59%
Temme et al., 2013	Ref: Average Dutch diet of young women in 2003	1400	
	30% of dairy and meat replaced by plant-based food	1100	−16%

(*continued on next page*)

Table A2 (*continued*)

Reference, description	Scenario	Total land use demand (m^2/person, year)	Change compared to reference scenario
	Vegan diet	700	−51%
Meier and Christen, 2012	Ref: Average German diet in 2006	2100	
	Healthy diet according to D-A-C-H[c]	1800	−15%
	Healthy diet according to UGB[d]	1700	−17%
	Lacto-ovo vegetarian	1500	−27%
	Vegan	1100	−50%
Arnoult et al., 2010 [e]	Ref: Average UK diet in 2003–2004	2000	
	Healthy diet[f]	1600	−18%

[a] Semi-vegetarian diet, an average between a healthy omnivorous and a vegetarian diet.
[b] Diet lower in meat, high in fish, fruits and vegetables, and plant oils instead of animal fats.
[c] Official dietary recommendations of the German Nutrition Society.
[d] Dietary recommendations by the Federation for Independent Health Consultation with less meat and more legumes and vegetables.
[e] Results were modified from livestock units to agriculture land based on a total agriculture area of 10,359,000 ha in 2006, and to per capita land use demand based on a population of 53 million in England and Wales.
[f] Based on national and international dietary recommendations including reduced consumption of dairy, red meat and sugar and increased intake of fish, fruit, vegetables and cereals.

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