WORKING PAPER 2012–1

Environmental and climate analysis for the Norwegian agriculture and food sector and assessment of actions

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About NILF

- Research and analyses on agricultural policy, food industry, commodity trade, economics, farm management and rural development.

- Documentation of economic results in the agricultural sector at national and farm level. This includes serving as secretariat for the Budget Committee for Agriculture and preparing the annual statistics for Account Results in Agriculture and Forestry.

- Develops tools for farm management and accountancy.

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Foreword

Working paper 2012-1 presents the outcome of the project "Miljø- og klimaanalyse for norsk landbruk og matsektor" (Analysis of Environmental and Climate-related aspects of Norwegian agri-food sector). The project has been led by NILF in cooperation with MiSA and John Hille. The objective has been to develop methodological guidelines for comprehensive assessments of measures aimed at relieving environmental and climatic stresses from Norwegian food production and consumption. The project has been 70 % funded by the National programme "Nasjonalt utviklingsprogram for klimatiltak i jordbruket", while NILF has covered the remaining 30 %. We acknowledge the Norwegian Agricultural Authority (Statens Landbruksforvaltning) for this support which made the project possible.

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Ivar Pettersen
Director General

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Sammendrag, diskusjon og konklusjon

Sammendrag: Hovedfunn

I denne rapporten studerer vi miljøeffekter, derav spesielt klimaeffekter, fra landbruk og mat sektoren. Fokus på landbruk innebærer å se på landbrukets produksjon av mat, mens fokus på mat innebærer å også vurdere foredling, distribusjon og forbruk av mat. I del 1 rapporterer vi karbon-fotavtrykk og andre mål for miljøpåvirkninger som forårsakes av matproduksjon – basert på resultater fra andre studier rundt om i verden. I del 2 diskuterer vi den nåværende kunnskap om klimaendringene fra mat og landbruksproduksjon i Norge med særlig vekt på resultater fra enkelte nyere studier. I del 3 blir miljøeffektene fra jordbruket beregnet basert på input-output modeller, og vi diskuter mulighetene for å bruke en slik modell. I del 4 har vi sett på ulike indikatorer for miljøbelastningen fra landbruket og også deres potensial for å brukes i utforming av miljøpolitikk. Hverken i del 3 eller del 4 vurderer vi valg som gjøres av forbrukerne. I del 2 presenterer vi en analyse utført av NILF/MiSA om klimagassutslipp fra mat og landbruk i Norge hvor både forbruks- og produksjons perspektivet er vurdert. I del 4 ser vi i hovedsak på indikatorer som er tilgjengelige fra jordbruket på gårdsnivå. På grunn av begrensede ressurser kan vi ikke, i del 3, teste betydningen av usikkerhet i modellen (selv om dette problemet delvis kan behandles av en tilgjengelig verktøykasse som bruker sensitivitets testing). I litteraturundersøkelsen, i del 1, fant vi at ulike estimater av klimagassutslipp i samme geografiske område kan variere vesentlig. Disse forskjellene kan delvis skylden usikkerheter i målinger av input data, men de kan også være forårsaket av bruk av ulike estimeringsmetoder (modell usikkerhet).

To tidligere norske studier (Hille m.fl. 2008; Hertwich og Peters 2009) indikerer at matforbruket i Norge produserer klimagassutslipp på ca. 2,5 t/person/år. Dette er innenfor rekkevidden av resultater fra undersøkelser i andre vestlige land, og forutsetter at maten bidrar til minst en sjettedel, eller kanskje mer, av de totale utslippene fra konsumentene. Studier som er nevnt i del 2 og del 3 presenterer resultater som er i overensstemmelse med dette. Refsgaard et al. (2011) beregner utslipp fra noen viktige matvarer som står for omtrent 50 % av matforbruket (på en vekt og energiinnhold basis), og viser at disse produktene er ansvarlig for utslipp av 5,7 Mt/år, eller 1,2 t/person. I del 3 presenteres et utvalg av mat der resultatene viser at nasjonalt produserte matvarer bidrar til utslipp av om lag 7,8 Mt/år, eller 1,6 t/person. Utslipp kan også stamme fra importerte matvarer. Vi kan dermed slå fast at matforbruket er en betydelig bidragsyter til klimagassutslippene i Norge.

Matproduksjon er også en dominerende bidragsyter til arealbruk og til forbruk av energi- og materielle ressurser. Videre er matproduksjon også en kilde til forsuring og andre miljøpåvirkninger som for eksempel overgjødsling. Flere nyere norske studier, presentert i del 2 og del 3, vurderer bare klimagassutslipp og arealbruk.

Ulike typer mat gir svært forskjellige klimagassutslipp, enten måleenheten er utslipp per kilo eller i forhold til innholdet av kostenergi. Målt per kg viser alle internasjonale studier at kjøtt fra drøvtyggere er blant de største bidragsyterne til klimagassutslipp fra matvarer. Kjøtt fra kylling og gris, egg og melk har lavere utslipp enn kjøtt fra drøvtyggere. Generelt gir vegetabilsk mat lavere utslipp enn animalsk, men grønnsaker fra oppvarmede drivhus og/eller sterkt bearbeidet mat gir like store eller større utslipp per kilo enn melk. Lokalprodusert korn, poteter og friske grønnsaker gir generelt lavere utslipp. Ser vi på utslipp per kostenergi innhold, vil vegetabilsk mat vanligvis gi lavere utslipp enn animalsk mat, men forskjellen minker, for eksempel mellom melk og kjøtt fra drøvtyggere eller mellom animalsk mat og grønnsaker og frukt når den måles per kostenergi Kcal i stedet for per kg på grunn av det høyere innhold av energi i kjøtt sammenlignet med melk og grønnsaker. Bruk av kostenergi som mål, innebærer at de mest energitette vegetabilske produkter (korn, sukker og vegetabilske oljer) har lavere klimagassutslipp enn for eksempel frukt og grønnsaker.

De siste norske studier presentert i del 2 og 3 viser at kjøtt fra drøvtyggere gir de største utslippene per kilo. Begge studiene som er presentert viser at melk gir langt mindre utslipp, mens poteter og kornprodukter bidrar i enda mindre grad. Studien i del 3 viser også at kylling og svin, etterfulgt av egg, ligger mellom storfekjøtt og melk i rangeringen fra største til minste klimagassutslipp per kg.

Klimagassutslippene fra mat forekommer i jordbruksproduksjon eller i produksjon av innsatsfaktorer til landbruket, for eksempel gjødsel. De kan også oppstå gjennom behandling, distribusjon og handel av varer. Internasjonale studier viser at utslipp fra utslipp fra jordbruksproduksjon og produksjon av innsatsvarer til landbruket er den klart største medvirkende kilden ved produksjon av animalske varer. Minst 80–90 % av utslippene fra disse varene skjer før produktene forlater gården. For planteprodukter, derimot, vil bearbeiding, distribusjon og handel stå for et mye større relativt bidrag til det totale utslipp, i noen tilfeller over 50 %. Modellen som er utviklet i del 3 ignorerer utslipp som stammer fra bearbeiding, distribusjon og handel med mat. Refsgaard et al. (2011) har analysert disse faktorene i sammenheng med poteter, brød, melk og oksekjøtt, og viser at utslipp fra både industri og handel kan være svært liten i Norge for disse matvarene, noe som sannsynligvis skyldes en klimavennlig norsk energiblanding.

Internasjonale studier tyder på at økologisk produksjon ofte gir lavere utslipp enn konvensjonell produksjon når det gjelder korn. Men for poteter, grønnsaker, frukt og animalske produkter, viser litteraturen forskjellige resultater. Enkelte studier viser at konvensjonelt produserte varer forårsaker lavere utslipp, mens andre indikerer at økologisk produksjon av de samme varene presterer bedre. Studien av Refsgaard et al. (2011) viser at økologisk produksjon gir bedre resultater i Norge for brød, melk og biff for spesielle typer produksjonssystemer. Flere faktorer som fôringsstrategi, dyrkningssystem og hvilke arealtyper som brukes kan påvirke disse resultatene betraktelig. Både studien av Refsgaard et al. (2011) og andre europeiske studier viser at økologisk produksjon krever mer areal enn den konvensjonelle. Studien fra Norge viser videre at produksjonen av storfekjøtt krever langt mer areal per enhet av kostenergi enn produksjonen av melk, som igjen krever mer enn korn eller poteter.

Diskusjon: Mulige tiltak for å redusere klimagassutslippene?

Det er politisk forventet av alle aktører sektorer av økonomien at de bør finne fram til tiltak for å redusere klimagassutslipp. Dette gjelder også de som driver med matproduksjon. Hvis vi tar sikte på å realisere så mye av potensialet for å redusere karbonutslipp fra matforbruk som mulig, er det klart nyttig å få en forståelse av i hvilken grad de ulike matvaretypene bidrar til utslippene. Likeledes er det nyttig med en forståelse av bidraget fra ulike stadier i matproduksjonens livssyklus og interaksjonene innenfor produksjonskjedene. Vi har i denne rapporten forsøkt å samle det nyeste og siste av litteratur med internasjonale og nasjonale analyser av hva mat og landbrukssektoren bidrar med i form av klimagassutslipp. Basert på dette vil vi forsøke å peke ut en rekke mulige strategier for å redusere klimagassutslippene.

Produksjonsorienterte strategier

I en produksjonsorientert strategi er tiltakene for å redusere klimagassutslippene forskjellige i de ulike deler av sektoren. I sine stortingsmeldinger, Klif (2010) og LMD (2009), diskuteres klimagass reduserende tiltak på gårdsnivå, for eksempel kan produksjon av biogass og biokull være et tiltak, og en bedre behandling av husdyrgjødsel. Men som vi diskuterer i del 2 (kapittel 2), når en skal forsøke å finne strategier for å redusere klimagassutslippene fra matproduksjon, er det også viktig å vurdere nivåene både oppstrøms og nedstrøms fra gårdsnivået. Dette understreker også LMD i sin stortingsmelding. Det er også viktig å velge de best rankede strategier først. Vi kan oppsummere de mulige tiltakene på følgende måte:

1. Forholdet mellom produkt, volum og klimagasser innenfor landbruket:
   a. Redusere antall laktasjoner per ku
   b. Endret fôringsregime
   c. Kjøtt produsert i kombinasjon med melk
   d. Import versus innenlandsk produksjon
   e. Intensiteten av kapital og andre innsatsfaktorer
2. Endring i produksjonssystemet (vurdere arealbruksendringer)
3. En viktig sak er bruk av areal og den sammenhengen at innholdet av bundet karbon i jord varierer mellom ulike jordtyper
4. Teknologiske endringer som skjer utenfor landbrukssektoren
   a. Landbruk/fiskeoppdrett
   b. Produksjon av innsatsfaktorer
   c. Foredling, distribusjon og handel.

Forbrukerorienterte strategier

Alternativt kan man fokusere på problemet fra et forbrukerperspektiv, og foreta en sammenligning av hvordan tilberedning av måltider kan skje ved bruk av ulike typer matvarer og sammensetning. Dermed vil analysen vise hvordan endringer i kosthold og matvalg påvirker klimagassutslippene. For eksempel har vi følgende muligheter:

1. Endre sammensetningen av matvarer (rødt kontra hvitt kjøtt, plante versus animalsk, redusere forbruket av kjøtt);
2. Redusere kast av mat (på butikker, i hjemmet, økt utnyttelse av råvarene).
3. Undersøke betydningen av ulike typer transport, og hvorvidt produksjonen skjer lokalt eller globalt.
4. Velge matvarer ut fra hvordan de er produsert, transportert, foredlet eller distribuert, for eksempel organisk kontra konvensjonell produksjon, at handel foregår rettferdig, produsert lokalt eller om den er importert, og så videre.

Arealbruk i produksjonen

Når en vurderer nye tiltak for å redusere klimagassutslippene, er det også nødvendig å se på mulige konsekvenser av disse handlingene. En rettferdig sammenligning av alternative landbruksprodukter må omfatte en vurdering av hvor mye land som trenges for å produsere en gitt mengde mat. Areal kan erstatte fossil energi og økologisk landbruk (i det minste under europeiske forhold) men har en tendens til å kompensere for at bruken av kunstgjødsel og sprøytemidler blir eliminert ved istedenfor å kreve mer land per enhet produkt sammenlignet med konvensjonelt landbruk. Et foreløpig anslag på konsekvensene av å konvertere til 100 % økologisk produksjon av brød, melk og storfekjøtt i Norge, vil ifølge estimater fra Refsgaard et al. (2011), redusere utslippene med omtrent 0,9 Mt CO_2-ekvivalenter. Dette er før endringen i arealbruk er vurdert. En slik overgang vil kreve 2,1 millioner daa mer jordbruksareal, inkludert både gressmark og dyrket jord. Hvis produksjonen bare skal skje innenlands vil sammensetningen være at det trengs 2,1 millioner daa mer dyrket jord og i tillegg 1,6 millioner daa innenlandsk grasmark som erstatning for 1,6 millioner daa reduksjon i «importert» land. Omtrent 12,2 millioner daa av jordbruksareal er tilgjengelig i Norge, men på grunn av behovet for endring i sammensetningen av dyrket mark, må også graden av karbonbinding i jord vurderes. En stor mengde av potensialet av areal består av skogkledde områder og er primært lokalisert i de midtre og nordlige deler av Norge. Derfor er videre analyser og beregninger nødvendige for å avgjøre hvorvidt utslippene vil reduseres eller økes som følge av en total endring til økologisk landbruk.

Forbruksmønstre

Analysene av Refsgaard et al. (2011) viser at både forbrukerne selv og landbrukssektoren kan bidra til å redusere karbonutslipp fra mat forbruk og produksjon. Mange nyere analyser og forslag til klimastrategier har fokusert på tiltak innenfor landbrukssektoren, herunder bedre håndtering av husdyrgjødsel, optimalisering av bruken av gjødsel, bedre drenering, produksjon av biogass, og blanding av biokull til jord. Men alle nivåer i produksjonskjeden – inkludert foredling og distribusjon - må utforskes videre med tanke på å identifisere potensialet for klimagass reduksjoner. Et alternativ (eller supplerende) tilnærming er å starte fra forbrukerens ende av matkjeden og spørre hvordan endringer i kosthold, representert ved alternative «kurver» av mat, kan bidra til å redusere utslipp og endringer i arealbruk.

Endringer i forbruksmønster kan gjøre at klimagassutslippene fra matvarekjeden reduseres uten at det skjer noen økning i behovet for jordbruksland. Spesielt vil to endringer kunne gjøre en stor forskjell: å redusere mengden av matavfall og å øke andelen av plante baserte – til erstatning for animalsk baserte - matvarer i kostholdet vårt. I dag ender omtrent 30 % av maten opp som avfall i Norge. Herav er minst 50 % spiselig og det er anslått at utslippene kan reduseres med omtrent 0,9 Mt CO2-ekvivalenter hvis dette avfallet ble eliminert. Disse tallene er basert på analysen av fire produkter og det forutsettes konvensjonell produksjon (Refsgaard et al., 2011). Som illustrert i del 2, har planteføde ikke bare en tendens til å generere mye mindre klimagassutslipp per enhet av kosttilskudd energi enn animalsk mat. Samtidig kreves også mye mindre jordbruksareale.

Betydningen av lokal eller global transport

Den lokale transporten i Norge bidrar i en mye høyere grad til utslipp av drivhusgasser i forhold til transporten av varer som ankommer landet. Dette gjelder sannsynligvis for mange land. Klimagassutslippene fra bøndene er hovedsakelig ikke knyttet til bruken av elektrisk energi eller fossilt brensel, mens det er i større grad nedstrøms fra gården (etter gårdsporten). Trailer transport krever mye mer energi og bidrar til en mye høyere grad til klimagassutslippene (målt per tonn-km) sammenlignet med transport med skip eller tog. Dette skaper en utfordring i analyser knyttet til lokal versus global mat. Norske empiriske analyser av drivstofforbruk og utslipp fra matdistribusjon mangler. Dette skyldes delvis det faktum at veitransporten er den dominerende modusen for de fleste matvarer, selv om skipet og jernbanetransport er mye mer energieffektive.

Konklusjon: Fremtidige behov for studier

Generelt er det et stort gap i litteraturen om påvirkning på miljøet fra den norske matsektoren. Del 1 framhever at det var få LCA studier på mat i Norge før studiene som er beskrevet i del 2 og 3. De tidligere studiene inkluderer en studie på norsk sjømat av SINTEF (Winther et al. 2009) og en studie på melk av Høgås Eide (2002), samt individuelle studier av veksthusproduksjon. Det er imidlertid mange LCA studier på gang for tiden.

Når det kommer til flere helhetlige studier, eller de som er fokusert på totalt matforbruk eller på større deler av matvaresektoren, ble det bare funnet et fåtall studier. De undersøker et begrenset sett av spørsmål:

- Hertwich og Peters (2009) gir et anslag for klimagassutslipp fra totalt matvareforbruk (utslippene kan ikke brytes ned til individuelle matvaregrupper eller foredlingsprosesser). Andre miljøpåvirkninger blir ikke overveid.
- Hille et al. (2008) gir estimater på energi og arealbruk knyttet til totalt matvareforbruk, men kun anslag for hvordan arealbruken er for hver enkelt av de ulike matvaregruppene. Hille og Germiso (2011) vurderer bare jordbruksarealet.
- Før disse studiene, må man gå helt tilbake til Breirem et al. (1980) for å finne en studie som forbinder energi- og arealbruk i forhold til mat produsert i Norge. På den annen side, Breirem et al. (1980) sa ingenting om effekten av importert mat eller sjømat, på energibruk nedstrøms fra gårdporten eller noe om utslipp.

Studiene som presenteres i del 2 og del 3 av denne rapporten dekker et begrenset antall matvarer. Refsgaard et al. (2011) dekker fire typer matvarer og estimerer tall for klimagassutslipp og arealbruk. Studien beskrevet i del 3 inkluderte flere matvarer, men bare opp til gårdsnivå, og kun tall for klimagassutslipp er gitt. Grønnsaker og importert mat ble ikke inkludert.

Refsgaard et al. (2011) kombinerer Referansebruksdata og økonomiske drifts-modeller for å beregne forholdet mellom mengder/type innsatsvarer og produksjon på en typisk gård. Det finnes et stort antall slike Referansebruk som representerer norsk landbruksproduksjon for ulike produksjonssystemer, regioner og størrelser, og dette kan kombineres med metoder for å beregne klimagassutslipp. Ved å videreutvikle dette vil en kunne gi anslag for utslipp fra flere ulike produksjonssystemer og/eller typer mat. I tillegg til slik statistikk kan dette gi mer kunnskap om effektene oppstrøms og nedstrøms i landbrukskjeden. Som påpekt ovenfor, viser resultatene fra denne studien at det er et potensial for å redusere klimagassutslipp fra mat og landbruk.

Spesifikke behov som er påpekt i del 2–4

Man kan ikke overse det faktum at de globale miljøeffekter av tiltak som kan foretas i Norsk landbruks- og matsektor ikke kan gjøres rede for før vi studerer effektene av endringer i sammensetningen av måltider, samt ser på summen av endringer i norsk landbruk og matdiett og import av innsatsfaktorer. I del 2 uttrykkes ønske om flere studier basert på et forbrukerperspektiv, inkludert studier som fokuserer på importerte matvarer og hva som skjer særlig i forbindelse med transport etter at maten har blitt levert fra gårdene eller importert. Vi ber også om for mer fokus på forholdet mellom bruk av areal, fôringsstrategi for drøvtyggere, og tilhørende karbonbinding.

Analysen i del 3 er en prøveanalyse som er gjort for å teste den foreslåtte metoden og dels for å teste nytten av eksisterende datakilder i en slik modell. For å gjøre modellen pålitelig må følgende spørsmål vurderes:

- Bedre oppløsning av den aktuelle landbrukssektoren for å få bedre anslag på virkningen av kryssleveranser (særlig vedrørende fôr);
- Bedre integrering av prosesser både oppstrøms og nedstrøms;
- Bedre geografisk oppløsning for produksjon av importerte innsatsvarer for å få mer realistiske utslipp anslag, fremfor å anta at alt er produsert med norsk (eller tysk) teknologi;
- Bedre integrering med pågående forskning på utvikling av nasjonale/internasjonale modeller, for eksempel HOLOS, CAPRI, eller Jordmod, for å få mer presise estimater av de direkte utslippene fra landbruket; og
- Sensitivitetstesting.

Som det fremgår i del 3, utslipp av klimagasser er det bare én type miljøkonsekvenser som kan håndteres med modellverktøyet som blir brukt av MiSA. Dette har sammenheng med del 4, hvor vi drøfter behovet for videre utvikling av indikatorer for effekter fra jordbruksproduksjon på miljøet.

Utbyttet av denne rapporten

Miljøanalyse av mat og landbrukssektoren er vanskelig. Videre vil metoder og de indikatorene som benyttes være av stor betydning for resultatet. I denne rapporten har vi etablert et grunnlagsmateriale for videre studier og for prosjekter som blir stadig mer etterspurt av det offentlige, av landbruks- og miljøorganisasjoner og av matvareprodusentene selv.

Introduction to report

There is increasing global, national, and regional awareness of environmental concerns, with a particular focus on issues related to the emissions of greenhouse gases (GHG). How food is produced, compiled, and processed as well as the composition of food consumption have great influences on global greenhouse gas emissions and also play a role in for instance water pollution. Food production consumes limited resources such as fresh water, phosphorus, and agricultural land. At the same time, food demand is increasing because of the need to feed the growing world population, which creates challenges in terms of making tradeoffs between greenhouse gas emissions, consumer interests, and social distribution issues. The International Panel for Sustainable Resource Management (UNEP, 2010)[1] concluded that a dramatic reform, or the innovation and redesign of the energy and agriculture sectors, could generate significant environmental, social, and economic returns.

In a white paper on agriculture and climate change challenges[2], the Norwegian Government showed that it has an ambitious climate strategy and is concerned about environmental problems. Consumer groups, food store chains, and environmental organizations have also recently focused their attention on the environmental impacts of food production. It is important to analyse the degree to which activities are carried out in accordance with the principle of sustainability and to better understand the extent to which the Norwegian food sector affects the environment. Such documentation will be useful for initially being able to identify effective environmental measures and, next, being able to analyse the management tools governments can use to implement them.

The task of carrying out an integrated environmental and climate analysis for the agriculture and food sector is overwhelming. The main purpose of this project will be to draw up guidelines for a doable comprehensive assessment of measures aimed at environmental and climatic stresses from the Norwegian food production and consumption. The analysis will be broader in terms of seeing the whole sector as a whole, and not only a part or a product. All in all, we shed light on how the major environmental impacts occur in the value chain and thus indicate which measures provide the greatest effect.

The environmental and climate analysis of the food sector is a relatively new area in both the international and national arenas. Within the Norwegian agriculture and food sector, there have been few completed environmental assessments or cost-effect estimates of climate change. In Part 1 of this report, we collect, identify, and interpret results from the economic literature regarding this subject. The review we present provides useful

[1] UNEP (2010) Assessing the Environmental Impacts of Consumption and Production: Priority Products and Materials, A Report of the Working Group on the Environmental Impacts of Products and Materials to the International Panel for Sustainable Resource Management. Hertwich, E., van der Voet, E., Suh, S., Tukker, A., Huijbregts M., Kazmierczyk, P., Lenzen, M., McNeely, J., Moriguchi, Y. http://www.unep.fr/shared/publications/pdf/WEBx0159xPA-PriorityProductsAndMaterials_Report.pdf

[2] St. meld. nr. 39, 2008-2009, Climate Challenges - Agriculture as a part of the solution. Landbruks- og matdepartementet. «Klimautfordringene - landbruket en del av løsningen.» http://www.regjeringen.no/nb/dep/lmd/dok/regpubl/stmeld/2008-2009/stmeld-nr-39-2008-2009-.html?id=563671

information on the conditions that should be the basis for a cost-effect analysis of Norway's food industry in addition to future measures to reduce emissions.

Apart from focusing on climate gas emissions that originate from agriculture, we want to include emissions that do not enter into official climate accounts (in accordance with the Kyoto Protocol). While much of the published and ongoing work has been based on a "producer perspective," meaning that the measurements link GHG emissions to enterprises in the food production chain, many actors and activities outside the agricultural sector affect the contribution that food makes to global warming (e.g., industries that deliver inputs to agriculture or process its products, agents that carry out the distribution to retail shops, and the consumers who carry out their shopping by car). If one views the food chain from the consumers' end, further issues become apparent, including the importance of international trade. Norway is a net importer of food and input factors to agriculture, so changes in Norwegian production and/or consumption may lead to larger or smaller emissions elsewhere, depending on how the changes affect net imports. Because the impact of GHG emissions is independent of where they occur, it is important to consider this effect in assessing strategies to reduce emissions. This in discussed in more detail in Part 2.

Part 3 describes the methods and results of our test run on how the data available for the process level in Norway can be used in a hybrid analysis along with data from Statistics Norway and farm level accounting data (FDA) from NILF. The result is an overall analysis of the agriculture sector. We discuss experiences that may be of help in the future development of methodology for calculating environmentally related agricultural management accounts.

In Part 4, we evaluate various environmental indicators that may be used in future FDA data assessments. Energy consumption is a key issue and is strongly profiled in the environmental debate, particularly with regard to CO_2 emissions, but we also evaluate other environmental indicators that are appropriate to collect.

PART 1: The carbon footprint and environmental impacts of food consumption: A review of results from previous studies

The purpose of this paper is to provide a background to the original research on environmental impacts of Norwegian agriculture that is described in other papers from the present project.

Research on the environmental impacts of food consumption has hitherto been rather limited in Norway. This applies not only to studies of its aggregate impact (which are not very numerous anywhere) but also to life cycle analyses (LCAs), carbon footprinting studies and other environmental analyses of specific food products. However, considerably more work has been done in some of our neighbouring countries. At the international level, the body of literature in this field is not only substantial but rapidly growing.

This paper compares and summarises some of the results from previous studies, with an emphasis on studies from other countries in Northern and Western Europe, though not to the exclusion of others where they are deemed relevant. Topically, the main emphasis is on the contribution of foodstuffs and of food consumption in general to global warming, but contributions to other environmental impacts and to resource consumption are also discussed.

Four main questions are addressed. The first is how much food consumption contributes to the total environmental loads of consumption in affluent countries such as Norway.

The second is *where* in the production chain – from production of inputs to primary production, via primary production itself (i. e. agriculture, fisheries and aquaculture) to processing, distribution and trade – the greatest environmental impacts from food arise. The research described in other papers from the present project mainly concerns impacts from agriculture and activities "upstream" of agriculture, i.e. production of inputs and capital goods for that sector. The wider context – i.e. an understanding of how much these activities contribute to the environmental impact of food, compared to what happens on the way from the farm gate to the consumer – may also be important, if policies are to promote sustainability in the food sector as a whole.

The third question is *which* foods contribute most to the carbon footprint and other environmental impacts of food consumption as a whole. Again, this is a question of obvious relevance to sustainable policymaking – and sustainable consumer choice. Can we significantly reduce the environmental impact of our food consumption through changes in diet, and if so, which changes might have the most effect?

The fourth question is whether *alternative systems* of food production – including processing and distribution – might significantly reduce the environmental impact of food. The main – though not exclusive – emphasis here is on the performance of organic vs. conventional systems of (primary) production, for the very simple reason that far more comparative studies of these exist than, for instance, of alternative systems of food distribution and trade.

The fifth and final chapter addresses the issue of whether new insights, specifically regarding process emissions of greenhouse gases from agriculture, may significantly alter the conclusions that can be drawn from the body of existing literature - either regarding the total carbon footprint of food consumption, or the relative contributions of different foods.

1 The environmental load of food consumption

1.1 Food and greenhouse gas emissions

Several studies from Norway as well as other countries indicate that consumption of food and drink is responsible for a significant share of our carbon footprints.

Although they do not (yet) measure carbon footprints, official statistics show that this must be true in Norway. According to Statistics Norway, GHG emissions from *Norwegian agriculture* in 2009 were 4.7 Mt CO_2eq (4.7 million tons of CO_2 equivalents), or 9 % of total emissions from Norwegian territory. This was slightly less in absolute as well as relative terms than in 1990 when agriculture was responsible for just over 10 % of emissions. Norwegian exports of agricultural products are minimal, which means that almost all emissions from agriculture can be ascribed to Norwegian consumption. On the other hand, Norway *imports* half of the food we eat (by energy content), and fisheries also make a contribution. Production of capital goods and inputs to the primary industries (”upstream” processes) also generate emissions, as do processing distribution and trade in food (”downstream” processes).

The usefulness of national statistics is rather limited with regard to most of these contributions to the carbon footprint of food consumption. They do provide figures for emissions from food processing, which with the exception of fish processing mostly supplies the domestic market. Excluding fish processing, this industry emitted 0.5 Mt CO_2eq in 2009. But figures for emissions from transport and trade do not specify the fraction related to food. Looking upstream, we do find a statistic on emissions from Norwegian fertiliser production, while other data can be used to split the production between Norwegian consumption and exports. This is about as far as emissions statistics will take us. Concerning emissions from production of other inputs to agriculture, or to processing, distribution and trade, Norwegian statistics are useful mainly insofar as a source of data on the volume or value of deliveries. The same applies to food imports. In these cases as well as for most of the downstream processes, we must rely on other sources and procedures to estimate emissions per unit of volume or value.

One attempt to estimate the aggregate carbon footprint of food consumption in Norway was made by Hille et al. (2008). Their method may be described as eclectic. A variety of sources and estimation procedures were used for different parts of the production chains. They arrived at an aggregate carbon footprint of either 12.5 or 16.7 Mt CO_2eq in 2006, depending on whether electricity consumed in Norway was assumed to be a Norwegian mix (almost all renewable) or – taking a more consequential approach – to represent a European mix. The share of food in the total Norwegian carbon footprint was estimated at 29 or 21 per cent respectively – in other words, assumptions about electricity made less difference to the footprint of food chain than to that of other kinds of consumption. Much of the former derives from non-energy emissions.

Hertwich and Peters (2009) calculated the carbon footprint of Norway as well as 72 other countries in 2001, using a considerably more stringent method – a hybrid, multiregional input-output analysis. They arrived at a per capita carbon footprint of 14.9 t CO_2eq for Norway, of which 2.2 t or 15 % was attributed to food. This corresponds to 10 Mt for the whole population in 2001. They assumed a national mix of electricity generation. It should be added that Hertwich og Peters (2009) as well as Hille et al. (2008) were left with some emissions that could not be attributed to consumption categories. By coincidence, the unattributed emissions made up some 12 % of emissions in both studies, but there the similarity ends. Hertwich and Peters were unable to allocate emissions from construction and trade to consumption categories. The food chain is likely to be responsible for a significant share of emissions from trade in particular. By contrast, the emissions left unattributed by Hille et al. mainly derived from services of marginal relevance to the food chain. Could all emissions in both cases have been distributed among consumption categories, then the two studies might have yielded quite similar results for food, assuming a Norwegian electricity mix. However, the study by Hille et al. would still have indicated a higher share of food consumption in the total carbon footprint.

Hertwich and Peters (2009) also found that food was responsible for a smaller share of the carbon footprint of several other Western European countries than of Norway (e.g. Denmark 12 %, Germany 13 %, UK 14 %). For the USA, their figure was as low as 8 %. The main reason for this is simply that shelter was responsible for more of the carbon footprint in countries where home heating and appliances depend largely on fossil fuels and fossil-generated electricity, than in Norway. In countries where nuclear power and/or renewables are more important, food made a somewhat larger contribution to the carbon footprint (e.g. Sweden 16 %, France 19 %). These are still minimum figures, since some emissions were not allocated to consumption categories.

Some other studies covering relatively affluent countries or regions have arrived at significantly higher estimates of food's share in the carbon footprint than did Hertwich and Peters (2009). For instance, Tukker et al. (2006) estimated that food – in this case excluding alcoholic beverages – was responsible for 29.3 % of the carbon footprint of the EU-25 in 2000. They used a single-region input-output analysis, so that the same carbon intensity was ascribed to imports as to domestic production. Garnett (2008) found, by using a mix of estimation procedures, that food was responsible for 19 % of the UK carbon footprint. Jones (2005) arrived at an estimate of 16 % of the carbon footprint of households in the USA. He used a hybrid approach, calculating emissions from direct use of energy form statistics and process LCA data, but other emissions by input-output analysis. Weber and Matthews (2008) performed an input-output based analysis of the carbon footprint of US food consumption, using the same input-output tables as Jones (2005), and arrived at a very similar absolute figure. Griesshammer et al. (2010) found, in a study relying largely on meta-analysis of previous work, that food was responsible for 15.2 % of the German carbon footprint. (However, their estimates of specific emissions for some foods appear rather low in the light of other studies, as we shall see later.) Saxe er al. (2010) estimated the carbon footprint of an average Dane's diet in 2006 at 1.92 t CO_2eq per year, based on consumption statistics and previous process analyses. They did not estimate the carbon footprint from other consumption categories.

A study of the Dutch carbon footprint by Vringer et al. (2009) is particularly interesting from a methodological perspective. They estimated it using four different methods: a single-region and a multi-regional input-output analysis, and a single-region as well as a multi-regional hybrid analysis, in which data from life cycle process analyses were used to adjust the input-output data. Regarding the carbon footprint of food, straight input-output analysis and hybrid analysis did not yield very different results. On the other hand, the difference between single-region analysis, in which it was assumed that foreign production sectors exporting to the Netherlands had Dutch emissions intensities, and a multi-regional analysis was considerable. Whereas the hybrid single-region analysis showed a carbon footprint from food consumption of 2.8 t CO_2eq per capita in 2006, the corresponding multi-regional analysis raised the figure to 3.9 t per capita. The share of food in the total Dutch carbon footprint also rose, from 28 % to 33 %.

The table below summarises the results of the studies mentioned above.

Table 1.1 Estimates of the carbon footprint of food consumption and its share of the total carbon footprint of consumption

Authors	Country	Reference year	Emissions per capita, t CO_2eq	Share of total carbon footprint, %
Hertwich and Peters 2009	Norway	2001	2.2+	15+
Hille et al. 2008	Norway	2006	2.7+ or 3.6 +	29+ or 21+
Tukker et al. 2006	EU-25	2000	N.A	29+ (households only)
Garnett 2008	UK	c. 2007	2.4	17
Griesshammer et al. 2010	Germany	c. 2007*	1.7	15
Saxe et al 2010	Denmark	2006	1.9	N.A.
Jones 2005	USA	?	3.2	16 (households only)
Weber and Matthews 2008	USA	1997	3.1	N.A.
Vringer et al. 2009 (Hybrid analysis)	Netherlands	2000	2.8 or 3.9	28 or 33

All the results confirm that the carbon footprint of food consumption is considerable.

Excepting the studies by Griesshammer et al. and Saxe et al. – the only studies in which emissions from the food chain were estimated from process LCA data alone – all the authors arrived at emissions of more than 2 t CO_2 per capita in the country or region concerned. (There can be no reasonable doubt that the relative figure arrived at by Tukker et al. (2006) translates into a significantly higher absolute amount than 2 tons.) The third lowest figure – that for Norway according to Hertwich and Peters (2009) - would increase to 2.5 t if the unattributed emissions in their study were distributed pro rata among consumption categories. However, their study showed somewhat lower footprints from food consumption in some other affluent countries than Norway –not merely in a relative but also an absolute sense. No conclusions about actual differences between countries can be drawn from table 1.1, since the studies differ significantly in methodology.

Of the sources above, only Garnett (2008) considered emissions by consumers (related to transport from supermarket to home, storage and preparation of food) and this component of the UK food footprint has been omitted from the figures in table 1.1. By conventional groupings of consumption categories, such emissions should be attributed to household consumption of transport equipment, household appliances and energy goods, not to food consumption. However, the studies also differ as to whether they include (a) government consumption and (b) food and drinks consumed in restaurants and the like. Hertwich and Peters (2009) included neither in their definition of food consumption. Hille et al. (2008), Garnett (2008) and Griesshammer et al. (2010) included both. Tukker et al. (2006), Jones (2005) and Weber and Matthews (2008) left government consumption completely out of their studies, but the two latter did include food consumption outside the home. Tukker et al. (2006) did not. The very high share of food in the carbon footprint according to this source would have become still higher if some portion of the footprint from services of hotels and restaurants (9.1 % of the total carbon footprint) and consumption of alcohol and tobacco (1.7 %) had been included in the food category.

None of the studies mentioned above include estimates of carbon sequestered or released as a result of new land being cleared or of regular agricultural practices. In recent years, some studies have appeared which do include estimates of carbon exchange between soils, vegetation and the atmosphere. IDA (2009) estimated the carbon footprint of food consumption in Denmark at 2.0 t per year – a result similar to that of Saxe et al. (2010) - if effects of land-use change were not included, but at 2.8 t if they were. Audsley et al. (2009), referring to Garnett's (2008) estimate of the UK food carbon footprint, estimate that it would grow by 30 % if effects of land-use change were included. We shall return to these issues later.

We may ask whether there are real reasons to assume that the per capita carbon footprint of food consumption in Norway may differ from that in the other countries represented in table 1.1 – and especially whether there are *more* such reasons than the exceptional mix of electricity sources in Norway. There could for instance be differences in:

- diet
- the intensity of input use and/or investments in agriculture
- transport distances between points along production chains and in final distribution of foods, and/ or
- the structure of and the technologies employed by food processing industries.

Each of these is a complex issue. There is a paucity of comparative international studies on the last three of them. Concerning diet, however, FAO statistics on food consumption give some indication of differences between countries. There may be discrepancies between the FAO figures and the national statistics on which some of the studies cited above have relied. However, the FAO statistics are, at least in principle, based on a uniform methodology across countries, whereas definitions and classifications in national statistics on nutrition vary substantially. The table below shows the FAO figures for food consumption expressed in terms of primary products, in countries from which we have already cited studies and a few others from which we shall shortly do so.

Table 1.2 Food consumption in 2007, mainly recalculated to primary product equivalent*. Kg per capita (Source: FAOSTAT database)

	Norway	Sweden	Denmark	Finland	UK	France	Germany	Netherlands	USA	EU avg.
Cereals	125	102	117	108	113	119	114	83	112	125
Potatoes and roots	74	61	82	69	107	65	70	92	58	77
Other vegetables	78	88	98	79	92	98	94	103	128	117
Legumes	5	2	1	1	3	2	1	1	4	3
Fruit	142	117	112	94	127	117	88	136	111	104
Nuts	4	5	4	1	4	3	6	10	4	5
Sugar, sweeteners	43	43	62	34	36	37	51	47	68	39
Vegetable oils	15	16	6	11	18	20	17	18	29	19
Dairy products (as milk)	262	356	296	361	241	260	247	320	254	241
Eggs	11	11	20	9	10	15	12	18	14	12
Meat	65	79	98	73	86	89	88	71	123	86
of which ruminant meat	26	25	28	19	28	30	14	19	42	20
Seafood	51	29	25	32	20	35	15	19	24	22
Alcoholic drinks etc.	78	66	117	106	113	91	137	83	98	109
Coffee, tea etc.	14	12	13	14	10	10	10	10	7	8

** Alcoholic drinks and stimulants are presented on the basis of weight as offered to consumers, while the amounts of cereals and fruits used to produce these products have been subtracted from (other) consumption of cereals and fruits. Derivatives of vegetable oils and sugar have been recalculated only to equivalent amounts of vegetable oils or sugar; none of these products have been recalculated to a truly primary basis, such as amounts of harvested soybeans, rapeseed, sugarcane or sugar beet used to produce the oil or sugar.*

It appears that Norwegians consume the least meat and the most seafood of all the countries in the table. FAO definitions make apparent consumption of meat in Norway slightly lower than actual consumption, since price differentials lead to some informal (statistically unregistered) cross-border trade in this product. In other words, Norwegian individuals travel to Sweden, or to a lesser extent either to Denmark or to Finland, to buy meat (and other products that happen to be cheaper in neighbouring countries). According to estimates by the Norwegian Directorate of Health, this informal cross-border trade adds about 4-5 kg to Norwegian per capita meat consumption. If so, most of this should be subtracted from Swedish meat consumption, but the per capita effect for Sweden is less, since the Swedish population is almost twice as large.

This point notwithstanding, consumption of meat is probably less in Norway than in most other affluent countries. It is clearly much less than in Denmark and very much less than in the USA. However, the share of *ruminant* meat in total meat consumption is higher in Norway than in any of the other countries in the table. We shall return to the significance of this point.

Norwegian consumption of dairy products per capita was somewhat less than in Sweden or Finland, but not very different from that in most of the other countries. Almost the reverse is true of vegetables – Norwegian consumption is similar to that in Sweden and

Finland, but less than in all other countries. Consumption of fruit in Norway appears surprisingly high, and higher than national statistics published by the Directorate of Health would suggest. Some of the latter difference is probably due to different treatment of consumption of imported fruit juices, (mainly orange juice). In the national statistics these are reckoned on the basis of actual imported product weight – often in the form of concentrates – while the FAO figures refer in principle to the equivalent amount of primary products, e.g. oranges. Norwegian consumption of cereal products is on a par with the EU average, but slightly higher than in the individual countries represented in the table. On the other hand, Norwegian consumption of alcoholic drinks is less than in any other country but Sweden, which suggests that the amounts of cereals and/or fruit used to supply Norwegians with such drinks must be less than for most other countries. The significance of these differences in diet for the carbon footprint of food consumption depends of course on the relative emissions associated with the various product groups. We shall return to this issue in chapter 3.

There are some reasons to suspect that agriculture in Norway is more capital intensive than in most other affluent countries. The climate demands that animals be housed during winter. Also, Norwegian farms are smaller on average than those in most other countries represented in table 1.3. If each farmer owns at least one tractor and a range of other machinery, this will make for more equipment per hectare and per unit of production. As concerns the input to production that generates the largest GHG emissions – nitrogen fertiliser – consumption per unit of farmland and of production has at least until recently been very high in Norway. (IFA 2010, cf. FAO 2011). (Fertiliser consumption in Norway has fallen somewhat since 2008, in reaction to higher fertiliser prices.) Table 1.3, which was taken from Nymoen and Hille (2010) compares the number of tractors and inputs of artificial nitrogen fertiliser in Norway, Sweden and Denmark with the amount of agricultural land and yields of cereals and potatoes in the same countries.

Table 1.3 Agricultural area, number of tractors,, consumption of artificial nitrogen fertiliser and yields of cereals and potatoes in Denmark, Sweden and Norway

	Denmark	Sweden	Norway
Agricultural area, 1000 ha (2008)	2,668	31,763	1,024
Consumption of N in fertilisers, t (2007/2008)	220,000	172,830	116,000
N fertiliser, kg per ha	82	54	113
Number of agricultural tractors (2009/2007/2005)	140,366	119,582	114,110
Tractors per 1000 ha	53	38	111
Yield of cereals, kg per ha, 2006-2008	5,830	4,700	4,010
Yield of potatoes, kg per ha, 2006-2008	34,330	26,930	25,850

In relation to agricultural area, there are twice as many tractors in Norway as in Denmark and three times as many as in Sweden. In relation to production, these differences are even larger. Consumption of nitrogen fertiliser per unit area in 2007-08 was over twice as high as in Sweden and 40 per cent higher than in Denmark. Such factors may contribute to higher emissions per unit of production in Norway than in these neighbouring countries, and therefore make the carbon footprint of food consumption higher in Norway – especially for those foods in which Norway is largely self-sufficient.

No comparative analysis of transport logistics within the food chain or of final distribution of food products in Norway vs. other countries appears to have been carried out yet. However, there are some reasons to suspect that transport may contribute more to the carbon footprint of food in Norway than in most other affluent countries. The most obvious reason is that the country is large in area (and elongated in shape) but sparsely populated, at least by European standards. Population density in Denmark is 8 times higher than in Norway, in Germany and the UK some 15 times higher, and in the Netherlands 25 times higher. In the case of products which are produced domestically only in a small part of the country, or imported via only a few harbours or border crossing points, this is likely to mean that the transport work involved in distribution of each unit will be higher in Norway than in many other countries. The share of food that is imported – particularly of bulky foods – and distances to the countries from which imports are sourced may also be important. Norway does not import very large quantities of animal products, as domestic producers of these (and of some crops that can be grown in Norway) are protected by stiff import tariffs. But Norway does import a very large share of its consumption of plant foods. Due to the country's location at the periphery of Europe and the far north of the planet, they often have to travel long distances. Vegetables from Spain, for instance, have to make a considerably longer journey to Norway than to France or Germany. Whether internal Norwegian logistics are more or less efficient in other ways – say capacity utilisation of vehicles – than in other countries is not known.

No comparative environmental analyses of Norwegian and foreign food processing, or of and retail trade in food, appear to have been carried out. However, it is a fair guess that they would have shown smaller GHG emissions from these stages of the food chain in Norway than in most other countries, due to the low carbon intensity of stationary energy use in this country.

1.2 The importance of the food sector to resource use and other environmental impacts

Production, distribution and trade in food obviously demand land, energy and material resources, and lead to a range of environmental impacts besides GHG emissions. An interesting question - but not an altogether easy one to answer – is whether the importance of the food sector to other environmental issues is greater or less than to climate change.

If we turn first to the question of resource use, we find that the answers often depend not merely on estimation procedures but also on definitions. In the case of (primary) *energy* use, there is admittedly a fairly well established conventional definition of what this is, which can be applied across economic sectors and processes. However, *land* use is a much less clear-cut category. Land may be 'used' by placing buildings or infrastructure on it, by growing crops or allowing animals to graze on it, by extracting timber from forests growing on it or by setting it aside as a national park, thereafter to be exploited only by tourists. Yet it is hardly meaningful to equate these uses with each other. In some environmental analyses – for instance in calculations of the 'environmental footprint' (EF) – such different uses of land are assigned different weights. In EF calculations, the environmental load of a country's or a person's annual consumption is expressed as the (weighted) number of hectares of biologically productive land and sea area it (or he or she) lays claim to. – An altogether different approach is to focus on *changes* in land use, so that the impact of an activity is expressed by the amount of land that needs (on a net basis) to be converted from less to more intensive uses. Once again, if several categories of land use, and therefore several possible types of conversion, are recognized, weighting procedures will be necessary. Methods for analysing such conversions among several intensities of land use have been proposed (Ecoinvent 2010), but there are so far few examples of their having been employed in broad environmental analyses of food consumption. – *Materials* have also been variously treated in environmental analyses. A main distinction is between those which focus on material flows and those which focus on extraction of materials. Material flow analysis focus on the amounts of materials moved as a consequence of an economic activity, generally counting them all in tons, whatever their nature – be the material soil, common rock, metal ores or biotic material. Where the focus is on extraction, it is usually more specifically on extraction of minerals of direct importance to the process or activity being analysed, and a ton of one mineral is seldom considered equivalent to a ton of another. Instead, they are usually weighted, commonly by the size of global reserves – so that if reserves are 1 Mt, extraction of 1 kg counts for ten times as much as 1 kg of a substance whose global reserves are 10 Mt.

Energy use

Hille et al. (2008) estimated the share of food consumption in total primary energy use needed to support Norwegian consumption in 2006 at 10.9 % or 12.5 %, depending on assumptions about the mix of electricity generation. Throne-Holst et al. (2002), using hybrid life cycle analysis, found that food was responsible for 15 % of primary energy use demanded by *household* consumption in Norway in 1995. Studies in Sweden, the UK and the Netherlands within the same research programme as Throne-Holst et al. (2002) arrived at very similar results concerning the share of food in direct and indirect energy use in their respective countries (Moll et al. 2005). This may partly be due to the fact that the studies used a good deal of common data input, based on Dutch research. In an updated study using the same basic methodology but also more recent Swedish data input, mainly from 2002, Räty and Carlsson-Kanyama (2007) found food responsible for 18 % of consumption-related energy use in Sweden.

It is reasonable to expect food to be responsible for a somewhat smaller share of energy use than of GHG emissions in most countries, simply because a large share of emissions in the food chain are *not* energy-related. However – and by the same token - in countries where energy supply is dominated by fossil fuels, it is also reasonable to expect the share of food in CO_2 emissions to mirror its share in primary energy use rather more closely than may its share in total GHG emissions. Most of the non-energy emissions, as usually calculated, consist of N_2O or CH_4. An analysis of CO_2 emissions related to Danish consumption was carried out by Munksgaard and Larsen (1999) who found that food (including beverages which they included in a separate category) was responsible for some 12 %. In Sweden – where energy use is much less dominated by fossil fuels than it was in Denmark in 1999 - Minx et al. (2008) nevertheless arrived at the same result, i.e. that food was responsible for 12 % of consumption-related CO_2 emissions. However, Räty and Carlsson-Kanyama (2007) found the share in Sweden to be twice as high (24 %). Much of the difference is due to the fact that Minx et al. attributed much higher specific CO_2 emissions to the electricity, district heat and biomass which are the main components of direct energy use in Swedish homes, than did Räty and Carlsson-Kanyama. Also, Minx et al. employed a multi-regional input-output analysis, which yielded much higher emissions from many imported products other than food, than did the single-region model used by Räty and Carlsson-Kanyama. Hille et al. (2008) found that the share of food in the *energy-related* Norwegian carbon footprint was 18 % assuming that electricity consumed in Norway represented a Norwegian mix of generation, and 14 % assuming a European mix. In Norway, as in Sweden, energy use in the food chain becomes more CO_2-intensive than the average of energy use demanded by national consumption, as long as one assumes a specifically national mix of sources for the electricity consumed within the country. Denmark, where most of the energy used within the country has hitherto been of indubitably fossil origin, represents a more common situation. In countries such as Denmark in 1999, there is no obvious reason to assume that food's share of the "energy footprint" deviates much from its share in the specific "CO_2 footprint".

Very roughly, the studies cited above suggest that food may be responsible for some 12-15 % of consumption-related energy use in North-Western European countries. Unsurprisingly, this is somewhat less than most estimates of its share in total GHG emissions (as opposed to CO_2 emissions).

Land use

Hille et al. (2008) also estimated the environmental footprint of Norwegian consumption in 2006. This indicator is designed to show what demands consumption makes on biologically productive land and seas. In EF calculations, consumption may demand:

- built-up land
- cultivated land
- grazing land
- forested land for timber production
- marine areas to produce seafood, and
- land to compensate GHG emissions through afforestation.

We shall leave the two last categories aside in this discussion, but comment briefly on the first four in relation to production of food other than seafood. Hille et al. (2008) calculated the demand on cultivated land for food from Norwegian agricultural statistics, national statistics on nutrition and imports and on exports of agricultural products, FAO statistics on yields to arrive at the amount of land demanded by imports, and conversion factors between imported or exported processed products and equivalent amounts of primary products. Agricultural land used to produce textile fibres and tobacco was subtracted to arrive at the amount demanded by food production. Grazing land was estimated from several secondary sources. The main source used for built-up land within Norway was a study by the Norwegian Government Auditor. However, Hille et al. (2008) were unable to estimate net 'imports' of built-up land, i.e. the difference (if any) between land claimed in other countries by agricultural buildings, factories, infrastructure etc. serving production for export to Norway, and the land claimed in Norway by buildings and infrastructure serving exports to other counties. The net imports were therefore set at 0, and the amount of built-up land dedicated to production of food for Norwegians therefore estimated as the share of built-up land within Norway that is dedicated to food production. This share is very uncertain in relative terms, but definitely so small in absolute terms that it has hardly any importance to the environmental footprint of food. Demands on forest land were calculated from statistics on the Norwegian timber cut, imports and exports of forest products and conversion factors from forest products to equivalent amounts of roundwood. The share of timber production that can be attributed to the food chain is mainly, though not solely, that used to produce packaging for food.

The table below shows the total amount of land in each of four categories that Norwegian consumption demanded in 2006, and their contributions to the environmental footprint after weighting (1 km^2 of built-up or cultivated land counts 3.165 times as much as 1 km^2 of forest land, while 1 km^2 of grazing land is equivalent to 0.362 km^2 of forest land). The table also shows the amounts of land that were attributed to food consumption.

Table 1.4 Contribution of food consumption to Norway's environmental footprint (EF) in 2006. Excluding marine area and land needed ton compensate for GHG emissions. Source: Hille et al. (2008)

Category	Land required, km^2	Of which for food, km^2	EF (weighted area), km^2	Of which for food, km^2
Built-up land	2 622	36+	8 299	113+
Arable land[3]	16 018	14 813	50 697	46 489
Grazing land	49 904	47 409	19 263	18 300
Forest land	40 814	2 381	40 814	2 381
Total	109 358	64 639	119 073	67 283
Percentage due to food		59		57

According to these calculations, food consumption is responsible for well over half of the Norwegian environmental footprint, excluding land needed to compensate for GHG emissions (food's contribution to GHG emissions has already been discussed). Had the marine area needed to produce seafood for Norwegian consumption been included, the percentage due to food in the last column above would have increased to 60. The demand for built-up land due to food consumption in the table above is a minimum estimate, which only includes land for agricultural buildings – not sites for manufacturing industry providing inputs to agriculture or processing food, nor for supermarkets nor for infrastructure required to facilitate transport along the food chain.

Hille and Germiso (2011) have since carried out a more detailed analysis of the amount of arable land demanded by Norwegian consumption in 2006. They arrived at a figure 4 % below that in the table above, viz. 15,411 km^2, of which 8,325 km^2 was in Norway and 7,085 km^2 abroad. Of this, they attributed 13,824 km^2 (89.7 %) to food production; the remainder was used to produce fodder for horses and pets, along with natural fibres and tobacco.

Vringer et al. (2009) estimated the share of food in the land demanded by Dutch consumption in 2000 at 63 %, using a hybrid, multi-regional analysis. This apparently refers to the sum of built-up land and agricultural land, including pasture, on a non-weighted basis. On such a basis, food is likely to turn out as the main contributor to land demand in almost any country.

Materials: flows and extraction

Little work has yet been done in the field of material flow analysis in Norway. The methodology for material flow analysis was developed mainly by researchers at the Wuppertal Institute in Germany during the 1990s. Two of them (Behrensmeier and Bringezu 1995) estimated that food, beverages and tobacco were responsible for 10 % of the material flow (Total Material Requirement, TMR) generated by German consumption in 1990, and about 12 % of the TMR generated by *household* consumption. Among Norway's neighbouring countries, material flow analysis has been has apparently been most practiced in Finland, where Mäenpäa (2005) found food responsible for 16 % of the TMR generated by household consumption, while Lähteenoja et al. (2007) attributed 15 %, or 5.9 t of a total TMR of 40 t per capita per year, to food. This included only food produced

[3] Includes grassland that is periodically tilled (covered by the Norwegian term "fulldyrka areal"). Such land is reported as arable by Norway to the FAO but may be included in "permanent meadows and pasture" in some other countries.

in Finland. Risku-Norja (2011), in a more detailed analysis of the TMR of Finnish agriculture, arrived at results that fit well with Lähteenoja's.

Many life cycle assessments of food products include figures for extraction of valuable abiotic resources. They are usually weighted – for assessment purposes – by the size of global reserves, as an indicator of scarcity. Extraction translates into depletion of reserves. The resources considered in such cases never include all the kinds of matter that enter into material flow analysis. Of the studies cited above, only that by Tukker et al. (2006) includes figures for abiotic resource depletion. The authors found food responsible for 20.6 % of the weighted depletion of abiotic resources through household consumption in the EU-25. As in the case of GHG emissions, this is a minimum figure, since Tukker et al. had separate categories for services of hotels and restaurants and for alcoholic drinks and tobacco. 7.0 % and 1.6 % respectively of abiotic resource depletion was attributed to these consumption categories.

Food and resource consumption: summary

To summarise these findings about resource consumption, it appears that food is responsible for a significant share of consumption-related energy demand – somewhere between 10 and 20 per cent – in Norway and some of its neighbouring countries. Nevertheless, food is responsible for a smaller share of energy use than of GHG emissions. Food is almost certainly the category of consumption that demands the most land, irrespective of whether only agricultural and built-up land are included in the analysis, or forest land as well. Whether analyses of land use changes would show food as responsible for the most conversion to more intensive uses is at present uncertain, though not unlikely on a global scale. Studies of the demand placed on material resources by food have not been carried out in Norway, nor apparently very often in other countries. It may be that food is responsible for a share of material flows and/or of abiotic resource depletion that is not very different from its share of energy demand.

Pollutants other than GHGs

We only need to look at official Norwegian statistics to realise that food production is a major contributor to nutrient pollution of water bodies as well as atmospheric emissions of ammonia. This is illustrated by the table below, which also includes emissions of non-methane volatile organic compounds (NMVOCs), the most important group of so-called ozone precursors.

Table 1.5 Emissions of selected pollutants by agriculture and aquaculture in Norway, 2009. (Miljøstatus i Norge,[4] Statistics Norway[5])

Pollutant	Anthropogenic emissions, total, tons	Emissions from agriculture, tons	Emissions from aqua-culture, tons	Percentage due to agriculture or aquaculture
N, to coastal waters	110 596	36 664	50 191	79
P, to coastal waters	13 014	891	10 470	87
NH_3, airborne	22 971	20 926	2	91
SO_2, airborne	15 707	30	2	1
NO_x, airborne	180 601	3 405	80	2
NMVOCs, airborne	141 241	3 351	110	2

4 http://www.miljostatus.no/miljodata/Miljodata/?spraak=NO&dsID=TEOTIL
5 http://www.ssb.no/agassn/tab-2011-02-11-05.html

Two food-producing sectors between them are responsible for the lion's share of nutrient emissions that reach the sea. Admittedly, aquaculture is mainly export-oriented, with only a small fraction of production being consumed in Norway, so the figures cannot be interpreted as contributions to the "nutrient footprint" of Norwegian *consumption*. But agriculture, which produces almost entirely for the home market, is – by a good margin – the second most important contributor to nutrient pollution, responsible for 61 % (N) and 35 % (P) of the pollution that was *not* due to aquaculture. – Of gaseous emissions that contribute to acidification, those of NH_3 come predominantly from agriculture. However, this sector is responsible for very small shares of SO_2 and NO_x emissions. Fisheries were responsible for 11 % of NO_x emissions, but like aquaculture, this is mainly an export-oriented sector in Norway. Food processing did not contribute more than 3 % to Norwegian emissions of any of the pollutants in the table.

Apart from GHGs, acidifying gases and ozone precursors, the Environmental Accounts of Statistics Norway[6] cover atmospheric emissions of 11 other substances, viz. carbon monoxide, two categories of particulates and 8 environmental toxins. Taken together, agriculture, fisheries and food processing did not contribute more than 4 % to total emissions of any of these substances in 2009. This does not necessarily mean that Norwegian food consumption contributed little to emissions of the substances covered – only that national statistics given no evidence to the contrary. They give no information on emissions related to imported food, to imports of inputs to agriculture or to distribution of food.

Substances that contribute to eutrophication, acidification or ozone formation are precisely the three categories of pollutants, besides GHGs, that are most frequently covered by life cycle assessments of food products (Tukker et al. 2006). To characterise these emissions, acidifying gases are commonly converted to SO_2 equivalents, ozone precursors to NMVOC equivalents (other units do occur) and nutrients variously to equivalent amounts of NO_3^- or PO_4^{3-} or to oxygen depletion potentials. It is generally possible to convert the differing units that may occur in LCAs into each other (CML 1992). Precisely the availability of recognized "common denominators" may be a reason why many life cycle assessments choose to focus on acidification, eutrophication and tropospheric ozone, along with global warming. Some studies do also provide figures on emissions of toxins, but the ways in which these are presented vary widely. While some stop short of any attempt at characterisation and merely present data on individual substances, others make separate assessments, say, of terrestrial and aquatic toxicity, while still others may roll these into one indicator of eco-toxicity, sometimes alongside an indicator of human toxicity, as in case of Tukker et al. (2006).

Although many LCAs of food products do cover acidification, eutrophication and ozone formation, this applies only to one of the macro-level studies of the consequences of national or EU consumption cited above, namely Tukker et al. (2006). Jones (2005) does however estimate US emissions of SO_2, NO_x, NMVOC (and particulates) by consumption category.

The table below shows the share of food in some environmental loads of household consumption in the EU-25 according to Tukker et al. (2006). The shares of hotel and restaurant services and alcohol and tobacco - parts of whose impacts are included in those of food consumption in several other studies – are also shown.

[6] http://statbank.ssb.no/statistikkbanken/Default_FR.asp?PXSid=0&nvl=true&PLanguage=0&tilside=selecttable/hovedTableHjem.asp&KortnavnWeb=nrmiljo

Table 1.6 Shares of some environmental impacts of household consumption in the EU-25 in 2000 attributed to food, alcohol and tobacco and services of hotels and restaurants by Tukker et al. (2006)

Impact	Food, %	Alcohol and tobacco, %	Services of hotels and restaurants, %
Eutrophication	58.1	1.6	12.6
Acidification	29.7	1.5	9.6
Ozone formation	25.5	1.9	8.8
Eco-toxicity	31.6	2.2	9.0
Human toxicity	23.6	1.9	8.4

Roughly, these results would suggest that food (including drink and eating out) is responsible for around two-thirds of the eutrophication generated by EU household consumption, and from more than a quarter to somewhat over a third of the other environmental loads. Recall that this source attributed a somewhat larger share of GHG emissions (as global warming potential) to food than did other sources cited in table 1.1. It is possible that some of these would have arrived at somewhat lower figures than Tukker et al. in some other cases as well.

Jones (2005) attributed about one-fifth of the SO_2 emissions and one-sixth of the NO_x emissions caused by US household emissions to food. He gives no figures for ammonia emissions, but we can safely assume that most of these would have been related to food consumption, and guess that if Jones had assessed total contributions to acidification, food would probably have been responsible for some 20-25 %. Jones also found that food was responsible for about one-fifth of the NMVOC emissions due to US household consumption.

Both of these studies thus suggest that food consumption contributed significantly to acidification and ozone formation, while Tukker et al. (2006) also found it to be the main contributor to eutrophication – hardly a surprise. It is reasonable to assume that much of the food chain's contribution to acidification (beyond that due to ammonia) and to ozone formation is due either to transport or to stationary energy use, including upstream fuel chains. Some of these emissions may be smaller in an *absolute* sense in the case of Norway than of the EU or the USA, simply because the electricity used in Norwegian agriculture, food processing and trade is mainly hydro-electricity, at least from an attributional perspective. But since this applies equally to the electricity used for all other purposes in Norway, it is far from certain that the *share* of food in Norwegian consumers' contribution to acidification or ozone formation would turn out to be less than in the EU or the USA. We have no way of guessing whether food is responsible for more or less of the *toxic* load of consumption in Norway than in the EU. Tukker et al. (2006) included hundreds of substances in their assessments of eco-toxicity and human toxicity. Norwegian statistics cover only eight, and are in any case unable to tell all about the impacts of Norwegian consumption.

Biodiversity

The mere fact that food production occupies a very large share of the most biologically productive land area on Earth places it indubitably among the human activities that have the greatest impacts on biodiversity. It is also unlikely that any human activity at sea has a greater impact on marine biodiversity than fishing. In spite of this, few studies of the life cycle of food products, and to this author's knowledge no studies of the environmental impact of food consumption in aggregate, have attempted to quantify impacts on biodiversity. Reasons for this are easy to find. Many of the data that would have been relevant for such assessments are usually unavailable, and even where data are available, the problems involved in transforming them to simple, robust indicators of 'impact on biodiversity' are daunting.

Nevertheless, several authors have proposed methodological solutions (Lindeijer (2000), Brentrup et al. (2002), Koellner and Scholz (2007), Geyer et al. (2010)). The proposals converge on the idea of classifying *land* in terms of ecosystem or habitat types, which can be rated by their value to biodiversity. The impact of a human activity on biodiversity may then consist of converting land from one type of habitat to another, or of maintaining land in a state different from a natural 'reference' habitat, which might otherwise have existed in the same place. Obviously, it is easier for most practitioners of life cycle analysis to categorize – say - a piece of arable farmland (or its conversion to farmland) in such a system than to go into the field and count the number of species present, not to mention analysing their intra-specific diversity.

So far, few LCAs of food products have applied habitat-type analysis to quantify impacts on biodiversity. One exception is a study of organic and conventional milk production in Germany by Haas et al. (2001). .

2 Where in its life cycle does food have the greatest impacts?

As we saw in the previous chapter, many studies from European countries and the USA have shown that food contributes significantly to their carbon footprints, although the estimated contributions range from about one-sixth to one-third of the aggregate footprint. Several studies also indicate that the food chain is responsible for a significant share of resource consumption as well as emissions of a range of pollutants other than GHGs.

If environmental policies are to address the impacts of food consumption, it is obviously important to know at what stages of the life cycle they mainly arise. Are the greatest impacts caused by primary production – agriculture and fisheries – or in upstream or downstream processes? Such insights can be important even if the predefined objective should be limited to reducing impacts from a particular link in the chain, say agriculture. Changes in the agricultural sector – for example in the mix of production, in agronomic practices or in the geographical distribution of various kinds of farming – may have secondary effects both upstream and downstream.

Unfortunately, some analyses of environmental impacts of food consumption do not break it down by stages of the life cycle. This is true for instance of several of the studies of carbon footprints that were cited in the previous chapter, especially those based on input-output or hybrid input-output analysis. This includes Hertwich and Peters (2009), Tukker et al. (2006), Jones (2005) and Vringer et al. (2009). All of these studies provide results for accumulated global warming potentials up to one point and one only in the life cycle of food. In the case of Hertwich and Peters, that point is where food products leave the processing stage (trade, including distribution, is not included). In the other three studies, the results refer to accumulated impacts up to the point where the products pass the retail outlet's cash register. At least from the published data, it is impossible to tell the shares of these impacts that arose in agriculture, in processing, in trade and so on.

Process life cycle analyses are by definition built up of data that concern individual stages in the life cycle. If we are fortunate, the results will also be presented so that the contributions from each stage can be identified (not all studies are equally transparent). But process analyses generally concern only one particular product or a small group of products. The results of any single study of this kind cannot be generalised. However, some authors have attempted, by combining results from large numbers of process analyses covering a wide range of foods, to estimate impacts from food consumption in general and to break them down by stages. Since there are still many products for which no life cycle analyses are available, and since those which are available vary in methodology, reference period and relevance to the country whose food consumption impacts are to be estimated, this approach requires a willingness to make more or less hazardous assumptions.

The section below gives some results from studies that have attempted to break down environmental impacts of food consumption by stages.

Carbon footprints by life cycle stages

Garnett (2008) attributed 45 % of the carbon footprint of food consumption in the UK to primary production, 5 % to upstream processes, 21 % to processing, 15 % to distribution and 14 % to trade, including restaurants etc. In these percentages, the contributions of storage and preparation of food in homes and of the waste stage have been left out of the total, although Garnett also estimated these. (Taken together, their contribution was about equal to that of trade). It should be noted that Garnett only counted one upstream process, namely fertiliser production. Fuller coverage of upstream processes would obviously have increased their share somewhat.

The study of the carbon footprint of food by Weber and Matthews (2008) represents an exception among input-output analyses, as they do give a partial breakdown by life cycle stages. Their main focus was on transport. According to Weber and Matthews, only 4 % of the carbon footprint of food in the US was due to distribution, and 5 % to trade. They did not specify the contribution from food processing, nor of processes upstream of primary production. However, they did estimate the contribution from transport across all stages, including inter alia transport of inputs to farms and of food from farms to processing. In all, transport contributed 16 % to the life cycle carbon footprint.

Hille et al. (2008) did not explicitly split GHG emissions from food consumption by life cycle stages, but they did break down energy use by stage and energy carrier. These data can be combined with conversion factors in the report and with estimates of non-energy GHG emissions, which Hille et al. also present, to calculate total emissions by life cycle stage. Only a few minor contributions to the carbon footprint cannot be split in this way. These all belong by definition either to the upstream or to the downstream processes, and would therefore slightly have increased their shares, and slightly decreased that of primary production, if they had been included in the breakdown.

Table 2.1 Breakdown of GHG emissions and of primary energy use caused by food consumption in Norway in 2006, according to Hille et al. (2008). Per cent

Process stage	Assuming Norwegian mix of electricity*		Assuming European mix of electricity*	
	GHG emissions	Energy use	GHG emissions	Energy use
Production of capital goods and inputs to agriculture and fisheries	17	24	15	20
Primary production	57	19	51	22
Food processing	9	24	14	26
Transport, downstream of primary production	15	22	13	17
Trade in food	2	11	8	16
Total	100	100	101	101

** Refers to electricity used for processes occurring in Norway.*

Based on the physical reality, which is that almost all the electricity used in Norway was hydropower, we can see that food processing (much of which occurs within the country) and trade contribute very much less to the Norwegian food carbon footprint than to energy demand. This is an untypical situation, due to the fact that Norway has a highly untypical energy system. The columns to the right, in which the carbon intensity of electricity consumption has been converted to the average for European OECD member countries, show that with such a mix of electricity the downstream processes would have contributed 35 % to the Norwegian food carbon footprint. This is still less than the 50 % found in the UK by Garnett (2008), though much more than Weber and Matthews (2008) found in the USA. Part of the difference between these results and Garnett's is simply that Hille et al. included more *upstream* processes in their analysis, so that this stage becomes much more significant.

Resource use and non-GHG pollutants

Irrespective of assumptions about the mix of electricity, Table 2.1 indicates that the breakdown of energy use by stages in the life cycle of food can be quite different from that of GHG emissions. The central point is that primary production is responsible for a much smaller share of life cycle energy use than of GHG emissions, as most of the GHG emissions from primary production – specifically agriculture – are not energy-related. If one is interested in ways of limiting energy use in the food chain, each of the five stages in the table may deserve almost equal interest.

We have no similar data on land use or consumption of material resources split by life cycle stages, but in the case of land it is fairly obvious that most is used by primary production. It is also reasonable to assume that primary production makes a major, and very likely the biggest, contribution to the material flows associated with food consumption, due to large flows of soil and biotic material. A comparison of Läteenoja's (2007) figure for the total material requirement (TMR) of the Finnish food sector with Risku-Norja's (2011) estimate for the TMR of agriculture alone certainly suggests as much. On the other hand, neither agriculture nor fisheries *directly* involve extraction of abiotic resources, as commonly defined in life cycle analyses. Extraction of minerals may take place either to provide raw materials for capital goods and inputs to agriculture or fisheries –and if so belongs to the upstream processes – or to provide capital goods and inputs to the downstream processes.

Concerning pollutants other than GHGs, we can be quite certain that primary production and not upstream or downstream processes is the main source of substances that contribute to eutrophication. To get an idea of where the most important contributions to other impacts come from, we must turn to life cycle analyses of specific food products.

3 Which foods have the greatest impacts?

Life cycle analysis of foodstuffs is a rapidly growing field. Hundreds of articles and reports containing analyses of one or more environmental impacts of one or more food products have been published in the past few years. A handful of input-output analyses of the aggregate impacts of consumption in countries or regions also specify the contributions, not merely of broad consumption categories such as food but of different kinds of food. As the number of primary sources available has grown, so too has the secondary literature in the shape of literature reviews and meta-analyses of the evidence concerning various aspects of the environmental impacts of the food chain. The aspect that has attracted most attention is food's contribution to global warming.

3.1 Emissions of greenhouse gases: Some findings from previous secondary studies

If we want an idea of the relative contributions of different foodstuffs to carbon footprints, the existing secondary literature may seem to offer an attractive short-cut. However, some caveats are called for. One is that the question of relative footprints can be answered in several different ways. Do we want to compare GHG emissions per kg of product, per kilocalorie, per gram of protein or per currency unit, i.e. with price as the denominator? Or do we want to know which foods are responsible for the largest *absolute* emissions, given the average composition of diets in the country in question? All of these ways of posing and answering the question occur in the literature, some of them in several variants (e.g. emissions "per kg" of meat may mean per kg live weight, per kg carcass weight, per kg dressed weight or per kg after preparation in the kitchen). It is usually possible to convert one denominator into another, though such recalculations often entail a certain margin of error. A more important problem is perhaps that it is not always self-evident which denominator is the most relevant.

A more specific problem in the case of Norway is that no secondary studies yet available that are based on Norwegian primary sources. This is an unsurprising consequence of the fact that not very many life cycle analyses of foods have yet been carried out in Norway.

A third problem is that the primary sources – from whatever country – vary widely in their methodologies. Griesshammer et al. (2010, cited in table 3.1), wishing to estimate the carbon footprint of German food consumption, found 177 apparently relevant primary sources, between them containing some 500 results regarding GHG emissions from production of individual food products. They threw out 100 of the 177 on the grounds that they did not conform to international standards for life cycle analysis, nor to the PAS-2050 standard for carbon footprinting. Even after eliminating over half of the original material, they felt obliged to comment on the rest: *'In assessing the studies it became clear that their geographical system boundaries varied widely and that there were major…differences regarding which parts of the production chain they covered…There were hardly any studies with identical system boundaries, so that we did not find it meaningful to present specific figures [on GHG emissions per unit product] for individual foodstuffs'.* Griesshammer et al. opted instead to present a range of possible average specific emissions for each group of foodstuffs, but were still forced to add that *"strictly speak-*

ing, these are neither true ranges nor true averages". Nymoen and Hille (2010) reviewed some 80 (mainly Northern European) studies with a view to estimating the carbon footprints of foods consumed in Norway. They noted the same problems of varying system boundaries and varying cut-offs, as well as the many problems of drawing inferences from foreign studies to Norwegian conditions. They ended up with a solution similar to that chosen by Griesshammer et al. (2010), viz. to surround their estimates with broad and explicit margins of error. Several combinations of high, medium and low values for various product groups were then used to test the central hypothesis of the study.

The most obvious differences in *system boundaries* among LCAs – and carbon footprinting studies - of food products concern the number of upstream and downstream processes covered. Some analyses stop at the farm gate or when fish are landed; some stop at processing, some at delivery to a wholesaler (or regional distribution centre), some at delivery to the retailer, some at the retailer's cash register and some only at the dinner plate. If one is not interested in the downstream processes, but only wants to compare different foods at the farm-gate stage, this need be no problem. If the studies are transparent and specify emissions stage by stage, it is simply a matter of eliminating the post-farm-gate emissions in cases where they are covered. Unfortunately, such transparency cannot be taken for granted. If one is interested in comparing emissions up to and including retailing, however, then studies which stop short of this stage are only useful if some other basis can be found for estimating the likely contributions of processes downstream of the study's system boundary. There are also some LCAs of food products that cover *only* processes downstream of primary production, but not very many. On the other hand, a rather common *upstream* cut-off is to leave production of capital goods outside the system boundary. Other significant methodological differences one encounters include differing treatment of emissions along the production chain of energy carriers used in food production, differing allocation principles in cases where by-products arise, and varying treatment of carbon emissions from or sequestration by the soil-vegetation system. We shall return to the significance of the last point. In studies cited below, carbon exchange between soils/vegetation and the atmosphere has been disregarded unless otherwise stated.

Primary sources that include analyses of several different products have the advantage that the authors will normally have applied a uniform methodology to all the products studied. However, this also means that methodological weaknesses will apply to all products, and such weaknesses need not affect all products equally. So a uniform methodology is not a guarantee of the fairest possible comparison

The table below shows some of the estimates that Griesshammer et al. (2010) and Nymoen and Hille (2010) arrived at for GHG emissions per kg of various products. The downstream system boundary which Griesshammer et al. sought to apply was ex retail, whereas Nymoen and Hille sought to align results one stage further upstream, i.e. they apply in principle to products as delivered *to* the retailer, or to kitchens in care institutions, which were the special focus of their study. As we have seen, neither wholesale nor retail trade contribute much to GHG emissions in Norway, so for *Norwegian* purposes the distinction makes little difference. Nymoen and Hille also calculated emissions per 1000 kcal of nutritional energy. The values for meat refer to dressed weight, while those for fish refer to fillets. Nymoen and Hille (2010) presented separate figures for vegetables in and out of (the Norwegian) season; those shown in the table are averages.

Table 3.1 Estimates of GHG emissions per unit of various foodstuffs by Griesshammer et al. (2010) (Germany) and by Nymoen and Hille (2010) (Norway)

Product	**g CO_2e/kg (Griesshammer et al. 2010)**	**g CO_2e/kg (Nymoen and Hille 2010)**	**g CO_2e/1000 kcal (Nymoen and Hille 2010)**
Bread	600 (400-1,300)	850 (595-1,360)	325 (228-520)
Flour, whole or rolled cereal grains	-	800/750 (525-1,280)	Wheat flour 246 (172-294) Rolled oats 193 (135-309)
Rice	-	4,000 (2,800-5,200)	1,170 (820-1,520)
Potatoes	-	430 (215-645)	514 (257-771)
Cabbage, root vegetbles and onions	Cabbage 200 (100-200) Roots 200 (200-700)	Cabbage, roots 320 (160-480) Onions 540 (270-810)	Carrots 888 (444-1,330) Onions 1,688 (844-2,530)
Other vegetables	Leafy vegetables 200 (100-200) Others 500 (100-2,200)	560 (280-840)	Cauliflower 2,260 (1,130-3,390)
Greenhouse vegetables	-	Tomatoes, cucumbers 2,300 (1,610-2,290)	Tomatoes 17,680 (12,380-22,980)
Tinned or frozen vegetables	Tinned carrots 700 (no range)	1,000 (500-1,500)	-
Dry legumes	-	700 (350-1,150)	Dry peas 244 (122-366)
Fruit and berries	Strawberries 300 (200-400)	Norwegian fruit 270 (135-505) European imports 600 (300-900) Other imports 1,300 (650-1,950) Strawberries 220 (110-330)	Norwegian apples 584 (292-876) European imports 700-1,500 (350-2,250) Other import 1,500-3,000 (750-4,500)
Juice	Cold drinks 600 (200-900)	Orange juice 1,000 (500-1,500) Apple juice 400 (200-800)	Orange juice 2,330 (1,150-3,500) Apple juice 930 (465-1,400)
Sugar	500 (no range)	1,200 (840-1,560)	300 (210-390)
Vegetable oils	-	1,300 (910-1,690)	Rapeseed oil 148 (104-192)
Margarine	-	1,500 (1,050-1,950)	203 (135-271)
Milk	800 (500-1,300)	Whole milk 1,200 (840-1,920)	Whole milk 1,818 (1,273-2,909)
Cheese	8,100 (6,800-9,000)	9,000 (6,300-14,400)	2,700 (1,910-4,320)
Butter	25,600 (23,500-27,600)	14,000 (9,800-22,400)	2,700 (1,910-4,320)
Eggs	2,900 (no range)	1,800 (1,260-2,880)	1,270 (889-2,032)
Beef, veal	15,500 (7,400-28,000)	20,000 (14,000-32,000)	11,800 (8,260-18,880)
Mutton, lamb	15,400 (15,300-15,500)	17,000 (11,900-27,200)	8,090 (5,660-12,940)
Pork	4,200 (3,600-5,000)	4,500 (3,150-7,200)	2,140 (1,500-3,420)
Chicken	-	3,000 (2,100-4,800)	2,780 (1,950-4,450)
Pelagic fish	Figure for pickled herring only, 7,900	Herring, frozen 850 (670-1,020) Mackerel, frozen 950 (760-1,140)	Herring 362 (290-434) Mackerel 500 (350-650)
Demersal fish*	Cod, frozen 2,800 (no range)	Cod 2,800 (2,240-3,360) Saithe 2,600 (2,080-3,120)	Cod 3,730 (2,980-4,480)
Farmed salmon	1,900 (1,430-2,290)	3,190	1,595 (1,280-1,910)

* "Demersal" here includes benthopelagic fish such as cod.

With a few exceptions, the ranges of emissions per kg given by these two sources overlap, but the estimates in Griesshammer et al. (2010) tend to be a bit lower than those of Nymoen and Hille (2010). It is quite possible that emissions from production of some foodstuffs actually *are* smaller in Germany than in Norway. On the other hand, Nymoen and Hille concluded that emissions from production of many foods, especially livestock products, must actually be close to the top end of the ranges shown, if the results were to square with available statistics at the macro level. Statistics on GHG emissions from Norwegian agriculture suggested that for meat, milk and eggs, this production stage alone must account for emissions about equal to those one would arrive at by multiplying the central estimates for emissions per kg of such products with total Norwegian consumption of the respective foods. (This check can be made in Norway because, on the one hand, Norway is almost self-sufficient in meat, dairy products and eggs, and on the other, 88 % of agricultural land in Norway is used to produce fodder for animals. Also, methane emissions from agriculture can be attributed almost entirely to animal husbandry). So to allow for upstream and downstream emissions, total emissions per kg of meat, milk and eggs must be a good deal higher than the first estimates that could be deduced by comparing LCA results. The same may be true of other products, but in the case of plant products Norway imports most of its consumption, and in the case of seafood Norway exports most of its production, so in these cases national statistics cannot be used even for a rough check.

Both of the sources in table 3.2 indicate that animal products generally have a larger carbon footprint per kg than plant products, although there are a few exceptions on both sides. Among animal products, ruminant meat tops the league for emissions per kg, followed by other meat, demersal fish, farmed salmon and eggs, while milk and (according to the Norwegian source) and pelagic fish have the smallest emissions. (We left butter and cheese aside.) Fats, greenhouse vegetables and fruit from other continents compete with milk, at the high end of the scale for plant products vs. the low end of the scale for animal products

Comparisons of emissions per kg of various foods are very common, but not necessarily very useful. If the average Norwegian were to swap his consumption of meat, eggs and cheese for an equal weight of cabbage and carrots, he would probably end up feeling hungry. Water makes up over 90 % of the weight of those vegetables. A comparison of emissions per unit of nutritional energy presents a rather different picture. Using nutritional energy as the denominator, the difference in specific emissions between animal and plant products becomes smaller, with considerably more overlap. Ruminant meat is still at the head of the league, but must share the distinction with vegetables from heated greenhouses (cucumbers and tomatoes are about 97 % water). Dairy products end up on a par with pork – and with many vegetables and fruits. Pelagic fish become as carbon efficient as potatoes, while the products with the smallest emissions per kcal are those plant products that consist almost entirely of fat (oils and margarine) or of carbohydrate (sugar).

This also goes to show that even comparisons by unit of nutritional energy do not tell the whole story. Cereals, sugar and plant fats and oils together make up half of the Norwegian calorie intake, but are probably only responsible for 10-12 % of the food carbon footprint. Changing to a diet exclusively of bread, margarine and sugar might look like an eminent climate strategy, but few doctors would recommend it. A more refined approach to assessing the relative climate impacts of foods might be to divide emissions by a composite indicator of their nutritional value, as Smedman et al. (2010) have proposed. We shall not discuss the merits of that particular proposal further, but it is important to remember that no comparisons based on simple denominators can tell all.

Quite another question is: which kinds of food have the largest absolute carbon footprint? The table below shows how the food carbon footprint breaks down in the UK according to Audsley et al. (2009), in Denmark according to Saxe et al. (2010) and in Norway according to Nymoen and Hille (2010, using the central estimates shown in table 3.1). There are some minor differences in downstream system boundaries between these studies, but they are unlikely to affect the relative emissions substantially. Nymoen and Hille (2010) did not estimate carbon footprints for nuts, cacao products, coffee, tea, spices or alcoholic drinks. To make comparison easier, we have omitted these products from the table as far as possible, although Saxe et al. (2010) provide figures that include all of these. Saxe et al. also have a separate category for soft drinks, which has been merged with sugar below (in line with Nymoen and Hille's study, in which sugar used to produce soft drinks is counted as sugar). The omitted product groups were responsible for 21 % of the Danish food carbon footprint according to Saxe et al. Nuts, coffee, tea and spices have also been omitted from the UK figures. According to Audsley et al. (2009), these products were responsible for only 2 % of the UK food carbon footprint. However, it has not been possible to eliminate alcoholic drinks or cacao products from the UK figures. These were not specified by Audsley et al.; cereals used for brewing or distilling are included with other cereals in their analysis, and grapes used for winemaking or distilling are included in a category for "grapes and wine" (3 % of the total food carbon footprint). Cacao is presumably included in their category "oil-based crops" (5 % of the total footprint).

Table 3.2 Estimated breakdown of the carbon footprint of food consumption by groups of foodstuffs in the UK, Denmark and Norway, c. 2009, and estimated shares of dietary energy intake in Norway. Percentages

Note: Coffee, tea, spices and nuts are not included. UK figures for cereals and fruit include raw materials for alcoholic drinks, which are not included in Danish or Norwegian figures. UK figures UK figures for (plant) oils and fats presumably includes cacao, which is excluded from Danish and Norwegian figures

	UK (Audsley et al. 2009)	Denmark (Saxe et al. 2010)	Norway (Nymoen and Hille 2010)	Shares of dietary energy in Norway (Nymoen and Hille 2010)
Cereals, other than rice	12	5	6	27
Rice	2	1	2	2
Vegetables	11	11	7	9
Fruitand juices	8	6	7	6
Sugar	1	1	3	15
Oils and fats	5	2	1	10
Dairy products	20	22	29	20
Eggs	2	1	2	2
Meat	36	49	41	10
Seafood	3	2	2	1

It appears that animal products are responsible for most of the food carbon footprint in these countries. According to both the Danish and the Norwegian figures, from which some plant-based products were excluded as mentioned above, the share of animal products was 74 %. According to the British figures, from which only a marginal contribution from plant-based products (2 % of the original total) has been omitted, animal products were responsible for 61 % of emissions.

There are some differences in diet between the three countries. For instance, consumption of vegetables is actually higher in the UK and in Denmark than in Norway, which may explain some of the difference in vegetables' share of the food carbon footprint. Also, meat consumption is higher in Denmark than in the UK or Norway, which may explain some of the differences in that case. On the other hand, consumption of dairy products in Norway is just slightly higher than in the UK and slightly lower than in Denmark. The higher contribution of dairy products to the food carbon footprint in Norway is at least partly due to (a) less consumption of *other* carbon-intensive products in Norway than in the other countries, particularly Denmark and (b) the inclusion of more plant products in the UK figures than in those for Norway (and Denmark).

According to Weber and Matthews (2008) animal products were responsible for some 57 % of the food carbon footprint in the USA. That result is just moderately less than what we might have arrived at for the three European countries in table 3.3, had all plant products been included in that table.

All three of the sources for table 3.3 include estimates of how changes in diet could reduce carbon footprints. Audsley et al. (2009) calculated the effects of various possible changes, such as reducing meat consumption by 50 %, 66 % or 100 %, or eliminating ruminant meat from the diet in favour of pork and poultry, or eliminating rice. Combining all of the most radical proposals – which would mean adopting an almost vegan diet (except for seafood) with no rice - would reduce the UK food carbon footprint by more than half. Nymoen and Hille (2010) proposed a nutritionally sound diet for residents of care institutions. Its carbon footprint was compared with that of the average Norwegian diet by scaling the amounts of food up to correspond to the average dietary energy intake, rather than the requirement of elderly people with a low level of physical activity. The proposed climate-friendly diet would result in 35 % less GHG emissions than the current average diet. (It was neither vegetarian nor vegan.) A sensitivity analysis in which it was assumed that all of the foodstuffs which the climate-friendly diet contained more of than the average diet had carbon footprints at the upper end of their possible ranges (cf. table 3.2) and vice-versa, still showed a reduction of 23 %. Saxe et al. (2010) proposed a "New Nordic diet", which involved more moderate changes from the current (in this case Danish) average than the diet proposed by Nymoen and Hille (2010). Meat consumption was reduced by 31 %, 2/3 of which was compensated by an increased intake of fish. Intake of dairy products was also increased. The result by their calculations would be a reduction of only 7 % in the food carbon footprint.

3.2 Comparison of results from 5 primary sources

Leaving these secondary studies, we shall now turn to some primary sources, i.e. original LCAs that cover several kinds of food. Table 3.3 includes one such study from each of five countries: Denmark, Sweden, Germany, the Netherlands and the UK. All of these studies were among the sources used by Nymoen and Hille (2010), while the British study (Williams et al. (2006)) was carried out by researchers at the same institution that delivered many of the primary data used by Audsley et al. (2009), and Wiegmann et al (2008) represent the same institution as Griesshammer et al. (2010). Therefore, the results in tables 3.1 and 3.2 are not, or not likely to be, altogether independent of some of those in table 3.3. An important point that emerges from table 3.3, however, is that different studies, each applying a uniform or near uniform methodology across all foodstuffs analysed, can nevertheless come up with quite different results regarding the *relative* carbon footprints of these foodstuffs. Now this could to some extent be due to real differences between countries: it is conceivable, say, that production of potatoes is more GHG-efficient in Germany than in Sweden, while the reverse may be true of chicken meat production. However, the magnitude of some of the differences makes it unlikely that they are due only to real differences in production systems among North-Western European countries. It is likely that different methodologies have affected the results for different foodstuffs in different ways. The study that differs most from the rest in methodology is that by Kok et al. (2001) who used hybrid input-output analysis. The others all used some form of process analysis, or borrowed their results from previous process analyses. The downstream system boundary in LCA Food (2003) and Wiegmann et al. (2008) is ex retail. LRF (2002) includes the consumer stage (transport from shop to home, storage and preparation of food), while Williams et al. (2006) stop at the farm gate. Therefore, one might expect a tendency for the figures in Williams et al. (2006) to be somewhat lower than in the other sources and for those in LRF (2002) to be higher, but no such tendency is visible.

Table 3.3 GHG emissions from production (and in some cases downstream activities) of various foodstuffs, according to 5 studies. g CO_2eq/kg product. (Meat recalculated where necessary to approximate dressed weight; figures for fish refer to fillets.)

	Source and downstream system boundary				
Product	**LCA Food 2003/2006 (Denmark) Ex retail**	**LRF 2002 (Sweden) Consumer**	**Wiegmann et al. 2008 (Germany) Ex retail**	**Kok et al. 2001 (The Netherlands) Wholesale**	**Williams et al. 2006 (UK) Farm gate**
Bread	White bread 840	Hamburger buns 1,017	820		
Flour, whole or rolled cereal grains	Wheat flour 1,130 Rolled oats790			Wheat flour 850	
Potatoes	220	C. 270 (338 peeled)	c. 200	261	240
Cabbage and root vegetables	Carrots 150		c. 150	Cabbage 116	
Other vegetables		Lettuce 518			
Greenhouse vegetables	Tomatoes 3,450			1,990	Tomatoes 5,900
Fruit			c. 500	Apples138 Grapes 373	
Sugar	960			1,210*	
Plant oils	Rapeseed or soybean oil 3,630			Sunflower oil 2,780	
Margarine					
Milk	1,010	987	950	2,267	1,060
Cheese	11,300	8,800	8,350		
Eggs	2,000		1,950		5,500
Beef and veal	2,220-68,000 (!)	14,000	All meats average 5,200	18,000	21,600
Mutton and lamb					23,300
Pork	4,650	4,250		11,400	8,400
Chicken meat	4,900	1,800	4,600	8,170	7,100
Pelagic fish	Frozen herrings 1,800			Herrings 1,000	
Demersal fish	Frozen cod 3,200			4,900	
Farmed trot	Frozen 4,470				

** Ex refinery*

The figures on bread and flour, potatoes and milk are quite similar, apart from the anomalously high figure for milk from Kok et al. (2001). There are much greater differences between the results for some other products. While the ratio between emissions per kg of eggs and of milk according to LCA Food and to Wiegmann et al. is about 2:1, it is 5:1 according to Williams et al. The corresponding ratio between chicken meat and milk is about 2:1 according to LRF, but about 4:1 according to Kok et al., 5:1 according to LCA Food and Wiegmann et al., and 8:1 according to Williams et al. Chicken meat has a much lower carbon footprint per kg than any other meat according to LRF, while other studies found only a small difference between chicken and pork, and Wiegmann et al. even found only a small difference between chicken and an average of other meats (unfortunately the basis for this average is not transparently presented in the source). Greenhouse tomatoes have a significantly lower carbon footprint according to LCA Food than according to Williams et al. One reason why Williams et al. (2006) arrived at higher figures than other sources for products from animals fed almost entirely on concentrates (pork, chicken and eggs) appears to be that this source has much higher figures for emissions in the production of cereal crops.

Those able to read Norwegian will find a comparison of results for more food products and from a wider range of sources tabulated in Nymoen and Hille (2010). The five sources used for table 3.3 were those that covered the widest ranges of products, among some 80 sources used by Nymoen and Hille.

3.3 Findings in recent literature

As mentioned above, environmental analysis of food products is a rapidly growing field. In the c. 18 months that have elapsed between the time Nymoen and Hille (2010) gathered their data and the writing of this report, dozens of new studies have appeared with new results of possible relevance to an assessment of the likely carbon footprints of foods consumed in Norway. This section presents results of some studies published in 2010-2011. Some recent studies are of interest especially for their results regarding the relative importance of various links in the production chain, for comparative results regarding alternative production systems or for what they tell about the potential consequences of expanding system boundaries. We shall return to some of these later. All results in this section concern conventional (non-organic) production systems.

Cereals

The table below shows some recent results regarding carbon footprints of cereal products.

Table 3.4 GHG emissions per kg of cereal products, according to some recent studies

Source	Product, country	Downstream system boundary	Carbon footprint, gCO_2eq/kg
Espinoza-Orias et al. 2010	White bread, UK	Ex retail*	c. 820
	Brown bread, UK		c. 790
Fazer 2010	Ruispuikula crisps, Finland	Ex factory	1,100
Barilla 2010a,b, EPD	Wasa Rågi crispbread, Sweden	To retail	817
	Wasa Solruta Sesam crispbread, Sweden		1,389
Lindenthal et al. 2010	Bread, 5 types, Austria	To retail	579-1,014
Lantmännen 2010, EPD	Macaroni, Sweden	Ex retail	720
Barilla 2010c, EPD	Penne, Italy	Ex retail	1,332
Blenghini and Busto 2009	Rice, Italy	Ex factory	2,900
Kägi et al. 2010	Rice, 5 cases, USA, Italy, Switzerland	Ex retail	c.1,300-3,400

** The analyses include the consumer stage. Including this stage emissions were 1061-1131 g CO_2eq/kg of white bread and 1036-1078 g CO_2eq/kg of brown bread. In both cases, the consumer stage was found responsible for some 25 % of emissions (read from figure), which has been subtracted above.*

None of these results differs radically from those found in many previous studies of cereal products, with the possible exception of the low-end result for rice in Kägi et al. (2010). However, this concerns rice from a dry cultivation system in Switzerland, which results in much lower emissions of methane than the globally more common wet (paddy) systems. Neither of the LCAs of rice above includes results from any of the Asian countries which are the sources of most of the rice consumed in Norway. The low-end result for bread in Lindenthal et al. (2010), which refers to white wheat bread, although not *radically* lower than some previous results for bread, is nevertheless somewhat surprising, since these authors assume that there are net carbon emissions from soil to atmosphere in conventional cereal growing. As previously mentioned, most LCAs of food products have hitherto disregarded such emissions (or net sequestration as the case might be). Taken together, however, these recent results do not substantially alter the picture regarding typical footprints of cereal products.

Vegetables and fruit

Two recent studies from the Netherlands and Austria each cover a range of vegetables and fruits. A recent doctoral thesis from Denmark includes an in-depth LCA of orange juice, while several recent studies have revisited the carbon footprint of greenhouse tomatoes.

Table 3.5 Carbon footprints of some open-field vegetables and fruits, according to recent studies

Source, country	Product, country of origin	Downstream system boundary	GHG emissions, g CO_2eq/kg
Blonk et al. 2010 Netherlands	Cauliflower, Netherlands	To retail	c. 250-350
	Green beans, Netherlands		
	tinned		c. 1,300
	in jars		c. 1,650
	Mushrooms, Netherlands		c. 1,650-1,950
	Apples, The Netherlands		c. 230
	Apples, New Zealand		c. 420
	Strawberries, Netherlands		c. 800
	Pineapples, Costa Rica		c. 450
	Bananas, Ecuador		c. 470
Lindenthal et al. 2010 Austria	Potatoes, Austria	To retail	c. 180
	Onions, Austria		161
	Carrots, Austria		97
	Lettuce, Austria		124
	Zucchini, Austria		c. 220
	Cucumbers, Austria		c. 220
	Tomatoes, Austria		c. 200
	Strawberries, Austria		c. 280
Trydeman Knudsen 2010 Denmark	Orange juice, from Brazilian concentrate	To distribution centre	424

Most of these results are within the ranges of those found for vegetables and fruits in previous studies. However, the figure for lettuce in Austria is markedly lower than that of LRF (2002) for lettuce in Sweden, and also much lower than those in another Swedish study (Wallén and Mattson 2002) as well as a British study (Milà i Canals 2007). Apart from this, the most surprising results are perhaps the high figure found by Blonk et al. for Dutch strawberries, and the figures the same source gives for fruit imported from other continents, which are significantly lower than some previous estimates (cf. Audsley (2009) and Nymoen and Hille (2010)). The difference between Blonk et al.'s result for Dutch apples and apples imported to the Netherlands from New Zealand is for instance less than half of the difference found by Stadig (1997) between Swedish apples and imports from New Zealand. If Blonk et al. are right on this point, it implies that emissions from long-distance transport of fruit by ship must be significantly less than several previous authors have estimated. The two results for preserved green beans, on the other hand, are respectively slightly below and slightly above the top end of the range estimated by Nymoen and Hille (2010) for preserved vegetables.

Some recent studies of tomato production in heated greenhouses suggest that typical GHG emissions may either be, or be set to become, significantly lower than most previous studies from Northern Europe have indicated. Note that of the sources for the table below, Högberg (2010) is partly a scenario study, while Nordenström et al. (2010) is a straight scenario study.

Table 3.6 Real or potential carbon footprints of tomatoes grown in heated greenhouses, according to some recent studies

Source, country	Heat source	Downstream system boundary	Carbon footprint, g CO_2eq/kg
Blonk et al. 2010	Natural gas	To retail	ca. 1,700
Netherlands	CHP		ca. 1,100
Högberg 2010	Natural gas	Ex retail	ca. 1,750
Sweden	Bio-energy		ca. 480
	Waste heat		ca. 120
	Natural gas + CHP, tomatoes imported from Netherlands to Sweden		ca. 880
Vermeulen 2010	Natural gas	Farm gate	ca. 1,750
Netherlands	CHP		ca. 850
Nordenström et al.	Heating oil	To retail	5,310
Norway	Biomass CHP		770-1,070*

** Depending on assumptions about electricity (Nordic mix og European marginal generation).*

Nymoen and Hille (2010) cited nine previous studies of greenhouse tomato production – from Norway, Sweden, Denmark, Finland, the Netherlands and the UK. The results were quite evenly spread across a range from 2,300 to 5,900 g CO_2eq/kg of tomatoes. Given the current mix of natural gas and CHP – or more precisely district heat – in Dutch greenhouses (Blonk et al. (2010)), it appears that Dutch tomatoes, even after export to Norway, may now have a carbon footprint significantly below the bottom end of that range. In Sweden, a large fraction of production is already based on bio-energy. It is possible that tomatoes produced in Nordic countries as well as imports from the Netherlands will soon have lower carbon footprints than imports from Spain, whose carbon footprint after delivery to Central Norway was estimated at 1,800 g CO_2eq by Nordenström et al. (2010) and 1,450 g CO_2eq by Hille (1998), though as little as 440 g CO_2eq by Bertelsen (2010). The GHG efficiency of greenhouse production is rapidly improving, a fact due not only to changes in energy use but also – at least in Norway – to increasing yields, which mean that less energy is needed for lighting and heating per unit of production.[7]

Sugar

Renouf et al. (2010) analysed cane sugar refining in Australia. Using mass allocation to split the environmental impacts between sugar and its by-products, they found that GHG emissions per kg of raw sugar were between 442-546 g CO_2eq, depending on other assumptions about the system. British Sugar (2010), who produce sugar from beet, have carbon footprinted their product in co-operation with The Carbon Trust. Emissions were found to be 600 g CO_2eq/kg of sugar (ex refinery). Both these results suggest that life cycle emissions from sugar may be somewhat less today than previous studies have indicated. This may be due to real changes in technology, including more utilization of the cane by-product bagasse as an energy source in the case of cane sugar refining, and more use of natural gas and CHP rather than electricity from condensing power plants and direct use of coal as a fuel in sugar beet refining (at least in Europe). Increasing yields in primary production may also be playing a part in reducing life cycle emissions.

[7] The area of tomato grrenhouses in Norway fell from 33.2 ha in 1996 to 32.0 ha in 2009, yet production grew from 8,565 to 10,923 tons, meaning that yield increased from 26 to 34 kg/m^2. (Statistics Norway, StatBank, Table 06046).

Margarine

Nilsson et al. (2010) carried out a comparative LCA of butter and margarine as sold by retail outlets in the UK, Germany and France. The carbon footprint of margarine varied between 1,100 and 1,660 g CO_2eq, but this concerned products with a higher water content than the standard margarine (as opposed to "light" margarine) sold in Norway. At a standard fat content of 80 %, the emissions would have varied between some 1,500 and 2,300 g CO_2eq. This range overlaps but is somewhat higher than the range estimated by Nymoen and Hille (2010) for standard margarine in Norway.

Dairy products

Several new LCAs of dairy products have been published in 2010-2011. The table below shows some of the results for milk. For other dairy products, results will tend to vary almost in proportion to the results for milk, since the lion's share of their life cycle emissions is due to primary production or upstream processes.

Table 3.7 Carbon footprint of liquid milk, according to some recent studies

Source	Country or region	Downstream system boundary	Carbon footprint, gCO_2eq/kg
FAO 2010	Western Europe	To retail	ca. 1,650*
Leip et al. 2010	EU-27	Farm gate	1,400
Lindenthal et al. 2010	Austria	To retail	1,186
Corson and Aubin 2010	France	Farm gate	1,037
Sheane et al. 2011	Scotland	Ex retail	1,400

** C. 1.500 g CO_2eq/kg to farm gate + 155 g CO_2eq/kg from downstream processes.*

Two of the sources above are of particular interest. FAO (2010) is a comparative analysis of dairy production across regions of the world, while Leip et al. (2010) is a major study of emissions from all kinds of livestock production, ordered by the EU Commission, using a calculation model with the acronym CAPRI. Both arrived at a somewhat higher carbon footprint than most previous Western European studies, though neither result is beyond the range of earlier results. The result in FAO (2010) includes emissions of 90 g CO_2eq/kg milk from land-use changes related to feed production, a factor disregarded in most earlier studies. On the other hand, it does not include emissions from production of capital goods, and some inputs are also disregarded. Leip et al. (2010) estimated emissions not only from land use change, but also from ongoing land use. Without these contributions, emissions from milk production in the EU-27 would have dropped to some 1,000 g CO_2eq/kg. It is notable that despite the fact that much of the feed for dairy cows comes from grassland, land use for milk production according to the CAPRI model leads not to net sequestration but to net emissions of CO_2. While this factor pulls the carbon footprint upwards – compared to that found in many previous European studies – another methodological refinement in Leip et al. (2010) has the opposite effect. The CAPRI model makes CH_4 emissions slightly lower and N_2O emissions significantly lower than those reported by EU governments under guidelines issued by the IPCC in 1996. Internationally recognised standards for life cycle assessments (ISO 2006) and for carbon footprinting (PAS 2050) – BSI(2011) still recommend using the IPCC 1996 guidelines, which lead to higher estimates for N_2O emissions than later IPCC guidelines. We shall return to this issue.

Among Nordic countries, Leip et al. found the carbon footprint of milk production in Sweden to be almost equal to the EU average, while it was higher in Denmark (c. 1,600 g

CO_2eq/kg of milk) and especially in Finland (c. 1,800 g CO_2eq/kg of milk). Carbon release from soils explains the high Finnish figure – the country has a large fraction of organic soils.

Eggs

Leip et al. (2010) estimated the carbon footprint of eggs in the EU-27 to be c. 2,900 g CO_2eq/kg, of which nearly 40 % was due to net release of carbon from soils used to produce fodder – a factor disregarded in most previous studies. Their estimates for Sweden and Denmark are somewhat lower (2,100-2,200 g CO_2eq/kg). This is just slightly higher than estimated in a previous Danish study (LCA Food 2003), but significantly higher than two previous Swedish estimates (Sonesson et al. 2008, Cederberg et al. 2009). In the case of Finland, Leip et al. estimate the carbon footprint of eggs at more than 6,000 g CO_2eq/kg. Once again, the high Finnish figure is due to carbon emissions from soils.

Meat

The table below shows estimates by Leip et al. (2010) of GHG emissions from production of meat in the EU. The source gives figures per kg of carcass weight. In the last column, these have been recalculated to approximate emissions per kg dressed weight, for ease of comparison with figures in previous tables. The following factors were used:
Cattle, sheep and goat meat: Dressed weight = 73 % of carcass weight
Pig meat: Dressed weight = 76 % of carcass weight
Poultry: Dressed weight = 65 % of carcass weight.

Table 3.8 Carbon footprints of meat in the EU-27, according to Leip et al. (2010). Downstream system boundary at farm gate

Type of meat	Carbon footprint, g CO_2eq/kg of carcass weight	Carbon footprint, g CO_2eq/kg of dressed weight (est.)
Cattle meat	22,200	30,400
Sheep and goat meat	20,300	27,800
Pig meat	7,900	9,900
Poultry meat	4,500	6,900

These figures are all markedly higher than those a majority of previous European studies have arrived at. Much of the reason is – once again – the inclusion of CO_2 emissions from land use and land use change. For all four kinds of meat, emissions from Danish production were found to be slightly lower than the EU average, and emissions from Swedish production a little lower again.

3.4 Carbon footprints of different foodstuffs

There is broad agreement among sources that animal foods generally have higher carbon footprints per unit weight than plant foods, although there may be a few exceptions. Also, there is broad agreement that ruminant meat has a higher carbon footprint per kg than other important animal foods. Among animal foods, other kinds of meat, demersal fish and eggs have lower carbon footprints than ruminant meat, but it is far from clear how pork, poultry meat, eggs and various species of demersal fish should de ranked in relation to each other. Milk and pelagic fish – at least if fresh or frozen, and not pickled in jars –

appear to have the smallest carbon footprints per kg among animal foods. Among plant foods, only rice, sugar, oils and fats and vegetables from heated greenhouses appear generally to have carbon footprints *per kg* equal to or higher than milk. However, fruit and (other) vegetables that have been highly processed or have travelled over long distances may also end up on a par with milk.

However, if we consider emissions per unit of dietary energy, the ranking changes noticeably. Plant products with high energy densities – sugar, fats and cereals – have the lowest carbon footprints by this measure, while fruit and vegetables, which consist mainly of water, have carbon footprints per kcal that surpass those of some animal foods. Ruminant meat remains very GHG intensive, but vegetables from heated greenhouses may in some cases be on about the same level. Pork, which has a fairly high energy density, may drop down to about the same GHG intensity as dairy products - while some fruits and vegetables, even if fresh and not extremely well-travelled, come into the same league as pork and dairy products. This does not apply to all fruits and vegetables – for instance, potatoes have a fairly small carbon footprint per kcal. However, fatty pelagic fish species such as herring and mackerel compare well even with potatoes.

Studies from Norway as well as Denmark, the UK and the USA indicate that animal foods make the largest *absolute* contribution to the carbon footprint of food consumption in these countries – a little more or less than 60 % of the total. The contribution of animal foods to dietary energy intakes is significantly less. Broadly speaking, animal foods are responsible for significantly more GHG emissions, even per unit of dietary energy, than plant foods. There are just more exceptions to the rule by this measure than if we compare emissions per kg.

Research on the carbon footprint of foods is a rapidly growing field. Recent work by FAO (2010) and Leip et al. (2010) suggests that the carbon footprints of animal foods may be rather higher than most previous studies have indicated. A study by Blonk et al. (2010) suggests that long-distance transport of fruit and vegetables may contribute less to their carbon footprint than several previous studies have indicated. Several recent studies indicate that the carbon footprint of greenhouse vegetables in Northern Europe may be shrinking noticeably. So may the carbon footprint of sugar. In both of the latter cases, technological change has contributed to lower emissions. Apart from these points, the recent studies cited in the previous section confirm the broad picture of the relative carbon footprints of foodstuffs that could be inferred from earlier work in the field.

3.5 Which phases of the life cycle contribute most to the footprints of different foods?

Previously, we have seen that et al. (2008) estimated that primary production, along with upstream processes, were responsible for 2/3 of the carbon footprint of food consumption in Norway (up to and including the retail stage). Garnett (2008) attributed only half of the carbon footprint of food consumption in the UK to primary production and upstream processes, but left most of the latter out of the reckoning.

If well over half of the carbon footprint of food is generated before the farm gate, this still does not *necessarily* mean than the greatest potential for reducing it is to be found on the farm or upstream. The downstream emissions, which are largely energy-related, *may* be easier to reduce than process emissions from agriculture itself. Also, the distribution of emissions among phases of the life cycle is not the same for all foodstuffs. In some cases, emissions downstream of the farm gate may surpass those upstream.

The table below shows the percentage of GHG emissions up to (and in some cases including) the retail stage that occurred before the farm gate, for a selection of different foodstuffs according to four European sources. The LCAs in LRF (2002) actually include the consumer stage, but it is possible to subtract retail *plus* consumer stages from processes upstream of these. Unfortunately the transparency of the report does not extend to separate data for the each of these last two stages.

Table 3.9 Percentages of the carbon footprints of foods (to retail or ex retail) that are due to processes up to the farm gate, according to 4 European studies

	Source and downstream system boundary			
Product	LCA Food 2003 Ex retail	LRF 2002 To retail	Wiegmann et al. 2008 Ex retail	Lindenthal et al. 2010 To retail
Bread		74	40	48
Potatoes	73	52	c. 40	c. 55
Other vegetables and fruit		Lettuce 56	$<<50$	Onions 32 Zucchini c. 70 Tomatoes, open field c. 25 Strawberries c. 55
Whole milk	89	c. 90	(Dairy products 82-99)	87
Meat	(>80)*	Cattle c. 90 Pig 84 Chicken 86	Fresh meat 79 Frozen meat 69	

** LCA Food has figures for the carbon footprints of meats both at the farm gate and ex retail, but they can only be approximately compared since the farm gate figures are given per kg live weight of animals while the ex retail figures refer to meat as sold to consumers.*

There is a marked difference between plant products on the one hand, and animal products on the other. For animal products, emissions downstream of the farm gate are minor compared to those that arise on the farm or upstream of it. This also applies to eggs. Pre-farm gate emissions were responsible for some 90 % of emissions in two Swedish production chains for eggs that were analysed by Sonesson (2009). In the case of meat, not even long-distance transport seems to make much difference. Ledgard et al. (2010) found that transport beyond the farm gate was responsible for only 5 % of the carbon footprint of New Zealand sheep meat sold in Europe. If emissions from primary production of animal products need to be revised upwards from the estimates made in the sources of table 3.9, as the results in Leip et al. (2010) suggest, this will push the pre-farm gate share of their carbon footprints still higher. The fact that Wiegmann et al. (2008) attribute a somewhat lower share of the carbon footprint of meat to pre-farm gate processes than other sources do is due to a surprisingly low estimate of emissions from those processes, compared to results from other studies.

In the case of plant foods, however – and not just of a processed food such as bread, but also of fresh vegetables – downstream processes can be responsible for a large share, or even most, of the carbon footprint. The share varies widely among products and production systems. Estimates of how large it is, and of the breakdown among various downstream processes, also seem to depend quite a lot on the methods used. Wiegmann et al. (2008) found distribution responsible for just 15 % of the carbon footprint of fresh vegetables in Germany, while total downstream emissions made up well over 50 %. The biggest contribution came from cool storage. LRF (2002) found distribution responsible for just 9 % of the carbon footprint of lettuce – delivered to retail – in Sweden, but the figure was 31 % for potatoes. Lindenthal et al. found distribution responsible for 37 % of the carbon footprint of onions in Austria – somewhat more than storage and packaging taken together. Weber and Matthews (2008), who estimated that distribution and trade were responsible for only 9 % of the carbon footprint of food consumption in the USA, nevertheless estimated the share at 25 % for fruit and vegetables. This is still surprisingly low, compared with the European estimates for fresh vegetables in table 3.9.

All of the figures above refer to products grown in the country where they are destined to be eaten, and with the partial exception of Weber and Matthews (2008) they refer only to fresh products. For bulky products such as fruit and vegetables, long-distance transport can make a very large contribution to carbon footprints, especially if that transport is by road (not to mention by air, but distribution by air is not very common). Carlsson-Kanyama (1998) found that Italian carrots sold in Sweden had a carbon footprint of 700 g CO_2eq/kg, compared with 200 g CO_2eq/kg for Swedish carrots. Angervall et al. (2002) found that transporting frozen broccoli, from the far south of Sweden to Stockholm (~500 km) resulted in emissions of some 175 g CO_2eq/kg, which made up 35 % of the product's carbon footprint. Road transport from Spain produced five times more emissions, and made up 62 % of the carbon footprint of Spanish broccoli sold in Stockholm. However, if the frozen broccoli was produced in Ecuador and transported to Sweden by ship, the transport emissions became significantly smaller than for the Spanish broccoli. The last point is also illustrated by Trydeman Knudsen's (2010) study of orange juice. She found that transport between various points along the production chain was responsible for no less than 57 % of the carbon footprint of Brazilian orange juice, as delivered to a regional distribution centre in Denmark. Yet shipping the orange juice concentrate from Brazil to Europe was responsible for only 3 %. The rest was due to road transport within Brazil, and more road transport from Rotterdam to a tapping plant in Germany and on to Denmark. As mentioned above, Blonk et al. (2010) found that long-distance transport of fresh fruit contributed less to its carbon footprint than some previous studies have estimated. Nevertheless, transporting pineapples and bananas from farms in Costa Rica and Ecuador to shops in the Netherlands was responsible for about 40 % of the products' carbon footprints, while the corresponding share was all of 80 % for apples from New Zealand. (The finding that New Zealand apples nevertheless had just over twice as big a carbon footprint as Dutch apples was due to lower emissions from growing them in New Zealand).

Norway now imports over half its consumption of vegetables (excluding potatoes) and almost 90 % of its consumption of fruit, including berries. In 2010, 58 % of these imports came by road vehicle[8], and of those imports, half came from Mediterranean countries. Having crossed the border – or entered via a ferry harbour at the southern end of the country – some of the products then travel very long distances via a distribution centre and on to their destination in Norway. At the extreme, the distance within Norway may be almost equal to the distance from Spain to the Norwegian border. Transport is therefore likely to make up a large share of the carbon footprint of fruit and vegetables consumed in Norway (Hille 1998). For compact plant products such as flour, sugar, fats and oils, the transport component is probably much less important. Høgaas Eide (2002) also found that transport makes a very small contribution to the carbon footprint of a compact animal product such as milk, even in Norway. On the other hand, several Swedish studies (e.g. Andersson et al. 1998, Sundkvist et al. 2001, Thomsson 2005) indicate that transport can be responsible for a large fraction of the carbon footprint of bread, which is a much

[8] Statistics Norwat, StatBank, Table 03064: http://statbank.ssb.no/statistikkbanken/Default_FR.asp?Productid=09.05&PXSid=0&nvl=true&PLanguage=1&tilside=selecttable/MenuSelP.asp&SubjectCode=09

less compact product than flour (and often distributed by smaller, less fuel-efficient vehicles).

Downstream processes such as cool storage and the running of retail outlets doubtless contribute much less to the carbon footprint of plant foods in Norway than, say, in Denmark or, provided one takes an attributional view of the Norwegian mix of electricity.

For products that have small carbon footprints up to the farm gate, but are then industrially processed, the processing may make up a large or even the largest share of the carbon footprint at the retail stage. The figures for tinned carrots in table 3.1 and for preserved green beans in table 3.5 illustrate this. Mattsson (1999) calculated that the carbon footprint of Swedish carrot purée in jars was 1,490 g CO_2eq/kg, more than seven times the footprint of fresh Swedish carrots according to Carlsson-Kanyama (1998). Wiegmann et al. (2008) found that the footprints of preserved vegetables in Germany were some 3-4 times higher per unit weight than those of fresh vegetables. A more extreme case was that of frozen potato chips, with a carbon footprint 40 times higher than that of fresh potatoes. (Since the latter was mainly due to the use of fossil-generated electricity for freezing and storage, the picture would have been rather different in Norway.) Wine is a rather different example of a processed plant product. Soja et al. (2010) found that processes downstream of harvesting grapes were responsible for 69 % of the carbon footprint of Austrian wine as delivered to retail. For products in tins, jars or bottles, the packaging often contributes very significantly to carbon footprints, and this factor is rather less sensitive to mixes of electricity generation than is freezing and cold storage.

Some animal products also have rather small carbon footprints in a fresh state. Whereas Norwegian as well as Danish sources have estimated the carbon footprint of fresh herrings at some 5-600 g CO_2eq/kg, Ritter et al. (1999) estimated the footprint of pickled herrings in jars in Denmark at over 2.000 g CO_2eq/kg. This is still modest compared to the estimate by Griesshammer et al. (2010) of 7,900 g CO2eq/kg for pickled herrings in Germany.

LCAs of *ready meals*, along with some other complex food products, have so far been rather few. One such study was carried out by Berlin and Sund (2010). The main subject of this study was a comparative analysis of two ready meals – a "hunter's meal" actually produced by Snellmann Kokkikartano Oy in Finland, and a chicken risotto from Fjordland of Norway, which was merely on the drawing board as a recipe when the study was carried out. Berlin et al. explicitly attributed 31 % of the carbon footprint of the "hunter's meal" (consisting of pork, mushroom sauce, potatoes and carrots) to post-farm gate processes. However, this did not include all processing, only the part carried out by the Snellmann company. Emissions from slaughtering pigs and transporting them to slaughter, for instance, are lumped together with primary production under the heading of "ingredients". The results for the hypothetical chicken risotto are rather more surprising. While packaging was estimated to contribute 10 % of the carbon footprint ex retail, processing otherwise contributed zero. This was not merely due to the Norwegian electricity mix, as energy use for processing was also estimated to be negligible. The authors do *not* comment further on this rather surprising result.

In addition to their comparative analysis of these two ready meals, Berlin and Sund also carried out a simpler "screening" analysis of six ready meals from other countries. They found that processing, including packaging, was responsible 7, 18, 19, 24, 28 and 45 per cent of the carbon footprint of these meals. The lowest percentage was for a meal consisting mainly of lamb and rice, i.e. both an animal and a plant food with relatively large carbon footprints before the farm gate. The highest figure was, unsurprisingly, for a vegetarian meal.

Virtanen et al. (2010) analysed six Finnish ready meals and found that the share of processing in their carbon footprints varied from about 12 % to 23 % (read from figure).

Both these results and those of Berlin et al. (2010) suggest that processing may be responsible for a significant share, but not the largest share, of the carbon footprint of ready meals consisting of a mix of animal and plant foods.

Another interesting issue – at least internationally - is whether or not emissions from industrial processing of food into ready meals is compensated for by smaller emissions at the consumer stage than would be involved in preparing the meals from fresh ingredients at home. (The question may seem of less interest in Norway, where cooking as well as industrial food processing is mainly by hydro-electricity. Again, it depends on one's perspective on the consequences of electricity consumption.) Davis and Sonesson (2008) carried out a comparative LCA of a semi-ready chicken meal and a meal prepared from the same ingredients at home in Sweden. They calculated that the carbon footprint of the semi-ready meal at serving was some 730 g CO_2eq, compared to 650 g CO_2eq for the other meal. Virtanen et al. (2010) compared several Finnish ready meals to similar meals prepared from basic ingredients at home, as well as in school kitchens. They found that the home-made meals systematically had higher carbon footprints than the ready meals, whereas the relationship between school kitchen meals and ready meals varied from case to case. However, the main issue turned out *not* to be energy use for preparing food at home. Instead, the lower emissions from ready meals was due to the fact that the primary producers who delivered ingredients to producers of ready meals were more GHG efficient than the average of those supplying retail outlets. This may be a peculiarly Finnish situation. It is impossible to draw any general conclusions from these two studies, which differ markedly not only in their results, but also in their methodology.

3.6 How much do different foods contribute to other environmental impacts?

Land use

Hille and Germiso (2011) estimated the amount of cultivated land used to produce 1 t of each kind of food consumed in Norway in 2006. The table below shows the results for main categories of foodstuffs, along with estimates of the corresponding land use per GJ of dietary energy. Note that the categories refer to foodstuffs "at wholesale level": highly processed foods are not specified, but are in principle counted as equivalent amounts of more basic ingredients. The figures per GJ were calculated using a dietary energy table from the Norwegian Institute of Agricultural Economics Research (NILF) (Mads Svennerud, personal communication). Some of the categories below do not exactly match those in NILF's table. See notes for explanations of how these cases have been treated.

Table 3.10 Land required by Norwegian consumption of food in 2006, according to Hille og Germiso (2011), and estimated land requirements per GJ of nutritional energy

Product	Consumption, kt	GJ dietary energy/ton	Land use km^2	Land use, ha/ton	Land use, ha/GJ
Cereals (including cereal products and rice)	407	14.5[1)]	1,214	0.30	0.021
Potatoes	264	2.9	202	0.07	0.026
Other vegetables	226	0.92	135	0.06	0.065
Fruit	632	1.24[2)]	550	0.09	0.070
Dried legumes	10.6	14.5	90	0.85	0.058
Nuts and oilseeds	22.2	No estimate	221	1.07	No estimate
Cacao and products	27.3	21.7[3)]	467	1.71	0.080
Sugar	160	16.3	179	0.11	0.007
Plant oils and fats	43.0	37[4)]	205	0.48	0.013
Sheep meat	27.1	8.4	736	2.72	0.323
Cattle meat	96.5	6.2[5)]	3,327	3.45	0.556
Pig meat	122	11.0	1,327	1.09	0.099
Poultry meat	66.8	4.5	506	0.76	0.168
Eggs	51.4	6.0	366	0.71	0.116
Dairy products except butter and cheese (as whole milk)	692	2.88	1,498	0.22	0.075
Cheese	77.5	15.4[6)]	1,348	1.74	0.114[7)]
Butter	14.0	31.2	200	1.43	0.046[7)]

1) Cereals: NILF's figures are 15 GJ/t for rice 14.5 GJ/t for other cereals. The consumption-weighted average is 14.54.

2) Fruit: NILF, figure is 1.92 GJ/t, but the denominator includes concentrated fruit juices, counted as they are at import and not converted back to equivalent tonnages of fresh fruit. The figure in the table has been calculated by dividing NILF's estimate of the absolute energy content of Norwegian fruit consumption in 2006 (cf. Utviklingen i Norwegian kosthold 2007, published by the Norwegian Directorate of Health), which was 783 TJ, by Hille and Germiso's estimate of fruit consumption, in which fruit juices were converted to primary equivalent.

3) Cacao products: The figure in the table is a weighted average of cocoa and chocolate products.

4) Plant oils and fats: NILF's figure for margarine has been multiplied by 1/0.8, as standard margarine contains 80 % fat while the tonnage figure in the table refers to virtually pure fats.

5) Cattle meat: NILF's figures are 4.18 GJ/t for veal and 6.27 GJ/t for beef. The share of veal in Norwegian cattle meat consumption is very small, but the average here is a guess.

6) Cheese: NILF's figures are 14.71 GJ/t for ordinary cheese, 19.23 GJ/t for whey cheese and 16.97 GJ/t for processed cheese. The figure above is an approximate weighted average.

7) Cheese and butter: The smaller amount of land per GJ of butter than of cheese follows from the allocation between dairy products used in the study (production of butter yields skimmed or low fat milk as an important by-product, whereas production of cheese usually yields whey only) combined with the higher energy density of butter.

The pattern of land requirements per unit of dietary energy is not dissimilar to that of carbon footprints per unit of dietary energy (table 3.1.) Sugar, cereals and plant-based fats demand little land in relation to their energy content. They make up half of the Norwegian dietary energy intake, yet require only 12 % of the land demanded by Norwegian food consumption, if alcoholic drinks, coffee, tea and spices – not shown in the table – are included in the 100 %. Vegetables and fruit demand considerably more land per unit of dietary energy; dairy products, eggs and pork demand still more; and ruminant meat demands the most, just as it is responsible for the largest GHG emissions. In general, production of foods that demand a lot of land also generates large N_2O emissions. In many cases, it will also entail large emissions of CO_2 from energy use in primary production and in upstream processes. Ruminants do not only release considerable amounts of methane from enteric fermentation – they are also less efficient than pigs or poultry at converting feed intake to weight gain.

Energy use

The table below shows estimates from various Swedish sources – which are more numerous than relevant Norwegian sources – of energy use for production and distribution of a selection of foods. The sources have been chosen because they give results not only for energy use but also for carbon footprints, so that the relationships between these can also be compared. The downstream system boundary is at delivery to retail, unless otherwise indicated. Most of the sources provide figures on end-use of energy only; in cases where they do provide figures on primary energy use, these are shown in parentheses. Of course the ratio between end use in the food production chain and primary energy use will be sensitive to the mix of energy carriers used as well as the mix of electricity generation. Up to the farm gate, most energy use in Sweden as in other countries is fossil (for most food products), and much of the difference between end use and primary energy will consist of energy use in extraction, refining and (partially) distribution of fossil fuels. Where Swedish-generated electricity enters the picture, the difference is affected by the fact that this is largely a 50/50 mix of hydropower (with small losses in generation) and nuclear (with large losses in generation). At the same time, the fact that fossil generation plays a very minor part in Sweden – as in Norway – means that GHG emissions from production chains where electricity plays an important part will tend to be lower than in many other European countries.

Table 3.11 Energy use in production and distribution of various foodstuffs in Sweden, and relationships between carbon footprints and energy use

Product	Source	Energy use, MJ/kg	GHG emissions, g CO_2eq/kg	g CO_2eq/MJ
Bread	LRF 2002 (hamburger buns)	14.8	940	64
	Barilla 2010 (Wasa Rågi)	(23.5)	817	
Potatoes	LRF 2002	3.1	193	63
	Mattsson 2001			
Carrots, fresh	Lagerberg-Fogelberg and Carlsson-Kanyama 2006*	2.4	69	29
Carrots, frozen	Lagerberg-Fogelberg and Carlsson-Kanyama 2006*	7.6	267	35
Carrot purée	Matsson 1999	24	1,490	62
Lettuce	LRF 2002	6.1	413	68
Tomatoes	Lagerberg-Fogelberg and Carlsson-Kanyama 2006*	51	2,700	53
	Möller Nielsen 2007**	28	860	31
Strawberries	Hagberg 2009	4.2-6.6	206-229	49-35
Rapeseed oil	Cederberg and Flysjö 2007**	6.9 (12.1)	1,000	145
Milk	LRF 2002*	5.26	ca. 950	181
Eggs	Sonesson et al. 2005	8.2-9.5 (17-18)	1,590-1,820	192-194
Cattle meat	LRF 2002*	53.4	ca. 13,900	265
Pig meat	LRF 2002*	32.8	4,659	142
Cod fingers	Sund et al. 2010*	52	3,400	65

* *Downstream system boundary at delivery to distribution centre.*

** *Downstream system boundary is ex processing or packaging plant.*

The table suggests that carbon footprints are higher in relation to energy use for animal products than for plant products. (The exception among plant products is rapeseed oil, in whose case refining demands a good deal of energy.) A main reason for this is that non-energy-related GHG emissions are higher in production of animal products, partly because of CH_4 and partly because the animal products generally demand more land per kg and are therefore associated with more N_2O emissions. Apart from the striking difference between animal and (most) plant products, we cannot draw any robust conclusions from this table. As usual, the methodologies of the various studies vary considerably. The differences between the two sets of figures for tomatoes may, however, reflect real ongoing changes, as discussed previously. Lagerberg-Fogelbergs and Carlsson-Kanyamas study had 2004 as its reference year, whereas Möller Nielsens figures were projections for 2008.

Had the retail stage been included in the figures in the table above, the percentage add-ons to energy use would almost certainly have been larger than those to carbon footprints, since retail trade in Sweden – as in Norway – uses little fossil energy.

The difference in energy use per kg between animal and plant foods is less than the corresponding difference in carbon footprints. By weight, consumption of plant foods is much larger than that of animal foods. If it is a goal of environmental policy to limit energy use, then measures that target the life cycle of plant foods may be important.

Emissions of nutrients, acidifying substances and toxins, and depletion of abiotic resources

We saw that the amounts of energy demanded by production and distribution of different foods do not necessarily vary in proportion to their carbon footprints. There is perhaps even less reason to assume at the outset that there must be a close relationship between carbon footprints and emissions of pollutants other than GHGs.

The table below shows estimates of the contributions to eutrophication and acidification from production of 1 kg of various foodstuffs. These are from LCA Food (2003) and LRF (2002), apart from the figure for eggs (not covered by either of these sources), which was taken from Sonesson et al. (2008). The different foods' contributions to these environmental impacts and to global warming are also compared. Nijdam and Wilting (2003) did not present figures for contributions to eutrophication and acidification per kg of product, only per Euro of the retail price. Since this source also estimates global warming potential per Euro, it is nevertheless possible to compare the figures, in the same way as the figures per kg in the other sources. The results from LCA Food (2003) concern Danish conditions, while LRF (2002) and Sonesson et al. (2008) concern Swedish and Nijdam and Wilting (2003) Dutch conditions. The downstream system boundary is ex retail in all cases except LRF (2002), in which the retail stage cannot be separated from the consumer stage. Therefore, the figures from this study are those at delivery to retailers. Given the mix of energy sources in the Swedish service sector, the contribution of retailing to acidification as well as eutrophication in Sweden is likely to be small.

Table 3.12 Contributions to acidification and eutrophication from production of various foods, and relative contributions to these impacts vs. global warming

Product	Source	Contribution to acidification per kg of product and relative to carbon footprint		Contribution to eutrophication per kg of product and relative to carbon footprint	
		g SO_2eq/kg	g SO_2eq/kg CO_2eq	g PO_4^{3-}eq/kg	g PO_4^{3-} eq/kg CO_2eq
Bread	LCA Food 2003	5.0	6.0	5.6	6.7
	LRF 2002	8.7	9.3	2.5	2.7
	Nijdam and Wilting 2003	-	6.7	-	7.2
Potatoes	LCA Food 2003	1.5	6.8	1.4	6.2
	LRF 2002	1.5	7.8	1.8	9.5
	Nijdam and Wilting 2003	-	6.0	-	15
Fresh vegetables	LRF 2002 (lettuce)	1.9	4.6	2.9	7.0
	Nijdam and Wilting 2003	-	5.0	-	7.5
Fresh fruit	Nijdam and Wilting 2003	-	9.4	-	7.1
Plant oils and fats	LCA Food 2003 (rapeseed/soybean oil)	31	8.5	42	11
	Nijdam and Wilting 2003	-	7.0	-	7.0
Liquid milk	LCA Food 2003	10.4	10	4.9	4.8
	LRF 2002	15.5	17	5.0	5.3
	Nijdam and Wilting 2003	-	12		9.5
Eggs	Sonesson et al. 2008	22-28	14-15	11.4-14	7.2-7.6
Cattle meat	LCA Food 2003 (forequarters of beef)	248	10	220	9.0
	LRF 2002	284	20	94	6.7
Pig meat	LCA Food 2003	75	16	39	8.5
	LRF 2002	62	13	36	7.8
Fresh meat	Niljdam and Wilting 2003	-	12	-	9.0
Fish	LCA Food 2003 (frozen cod)	32	10	5.3	1.7
	Nijdam and Wilting 2003	-	19	-	1.1

According to all the sources, but most markedly according to the figures from LRF (2002) there is a tendency for animal products to contribute even more strongly to acidification than to global warming, relatively to plant products. The reason is not hard to find: this is due mainly to emissions of ammonia in primary production of animal products. According to LRF (2002), ammonia is responsible for 78 % of the contribution to acidification from pig meat, but for only 28 % in the case of potatoes. In the case of fish, contributions to acidification come mainly from the fuel used by fishing vessels. They depend on the sulphur content of the fuel and technological factors that govern NO_x emissions per unit of combusted fuel.

The next table shows a comparison of contributions from various kinds of food to global warming, acidification, eutrophication, eco-toxicity and depletion of abiotic resources, according to results in Tukker et al. (2006). In this case, the source data refer to the percentage contribution the various categories of food make to the total environmental loads of household consumption in the EU-25. They have been indexed, so that the share of each category in the total global warming potential (GWP) due to household consumption equals 1. For example, meat and meat products are responsible for 11.99 % of the GWP from all household consumption, but only 6.96 % of the contribution to abiotic resource depletion. The relative importance of meat and meat products to abiotic resource depletion is therefore 6.96/11.99 = 0.58.

Table 3.13 The contribution of major categories of foodstuffs to other environmental loads of household consumption in the EU-25, relative to their contribution to global warming potential. Share in GWP = 1. Source: Tukker et al. (2006)

	GWP	Abiotic rersource depletion	Acidification	Eutrophication	Eco-toxicity
Brerad, cakes etc.	1	0.84	0.93	3.72	1.24
Fresh vegetables	1	0.60	0.66	0.51	1.51
Fresh fruit	1	1.03	0.63	1.51	1.42
Frozen or preserved fruit and vegetables	1	0.82	0.87	1.05	1.19
Plant oils and fats	1	0.68	0.74	1.38	1.29
Liquid milk	1	0.72	1.11	2.06	1.10
Meat and meat products	1	0.58	1.12	1.88	0.85
Fresh or frozen fish	1	0.85	0.78	0.79	0.59

The contribution of foods to abiotic resource depletion is generally less than their contribution to global warming. This is not surprising. In contrast to many other products consumed by households, food is not in itself made from abiotic resources. On the other hand, and partly for reasons that are peculiar to food, it makes a major contribution to global warming – a *very* major contribution according to Tukker et al. In fact, it is rather surprising that the differences between foodstuffs' contributions to GWP and to abiotic resource depletion are not greater: the figures imply that deliveries to enterprises along the food chain make rather large demands on abiotic resources, including of course fossil fuels and fertiliser minerals, but a wide range of other materials as well. Meat makes the smallest contribution to abiotic resource depletion *relative to its contribution* to GWP, since its contribution to GWP is high, partly for reasons unrelated to resource depletion.

Plant products contribute somewhat less to acidification than to global warming, while the opposite is true of animal products. This is in line with results from the previous table and no doubt due to higher ammonia emissions in production of animal products.

All but two categories of food in the table contribute more to eutrophication than they do to GWP. That fish are an exception is unsurprising – it is agriculture, not fisheries, that is a major source of nutrient runoff. However, the low figure for vegetables in the eutrophication column, especially compared with the extremely high figure for bread and cakes, has no obvious explanation. Indeed, it is so surprising that one may reasonably suspect some error in the model used by Tukker et al. That the contribution of frozen and preserved vegetables to eutrophication is just slightly greater than their contribution to GWP is less surprising. As we have noted previously, a large share of their contribution to GWP may be due to processing and packaging. These processes are not likely make a corresponding contribution to eutrophication.

Finally, we can see that plant products contribute more to eco-toxicity than they do to GWP, while this is less true of animal products. This is probably due to pesticide use, which is generally most intensive in production of fruit and vegetables, followed by other arable crops, and least on grassland. No pesticides are applied to fishing banks.

The differences between different kinds of food regarding the pattern of their contributions to various environmental loads are generally smaller according to Tukker et al. (2006) than we might deduce from process analyses (cf. the results from LCA Food (2003) and LRF (2002) in table 3.12). Input-output analysis, the method used by Tukker et al, tends to show smaller differences between products – in the pattern of their environmental impacts – than process analysis. One reason for this is that input-output analysis in principle covers *all* activities that contribute to the production chain, whereas process analysis has to make cut-offs. For instance, emissions caused by market gardeners' accountants and dairy farmers' accountants and the accountants' consumption of inputs are included in input-output-based results for vegetables and milk respectively, but are not likely to be included in process analyses. Since there is no reason to assume that accountants serving market gardeners cause more or less emissions of any substance than those serving dairy farmers (and no input-output analysis would be able to register the difference if it existed), this will tend to lessen the differences in emissions patterns between vegetables and milk. On the other hand, input-output analysis, since it is based on generalized data for economic sectors (usually broader than "accountancy" or "dairy farming"), may well miss information on factors peculiar to production chains for particular kinds of food, which process analysis is more likely to reflect.

Neither the results in Nijdam and Wilting (2005) nor those in Tukker et al. (2006) can be broken down by life cycle stages. However, those on acidification and eutrophication in LCA Food (2003), LRF (2002) and Sonesson et al. (2008) can be split between pre- and post-farm gate processes. The table below shows the percentages of total contributions to acidification and eutrophication that were due to processes up to the farm gate, according to these sources. Again, "total" in the case of LRF (2002) means all processes until delivery to retail, whereas the totals in the other sources include the retail stage.

Table 3.14 Shares of the contributions of various foodstuffs to acidification and eutrophication that were due to processes up to the farm gate, according to Scandinavian studies

Product group	Source	Acidification	Eutrophication
Bread	LCA Food 2003 (rye bread)	87	98
	LRF 2002 (hamburger buns)	c. 84	c. 94
Potatoes	LCA Food 2003	77	97
	LRF 2002	54	95
Other fresh vegetables	LRF 2002 (lettuce)	59	98
Liquid milk	LCA Food 2003		
	LRF 2002	>94	98
Eggs	Sonesson et al. 2008	c. 97	c. 98
Cattle meat	LCA Food 2003 (beef fore-quarters)	100	100
	LRF 2002	98	99
Pig meat	LCA Food 2003	100	100
	LRF 2002	93	98
Fish	LCA Food 2003 (frozen cod) (before landing)	97	98

Contributions to eutrophication are overwhelmingly due to processes before the farm gate, irrespective of the kind of food in question. This also applies to the contributions to acidification from animal products, but not as unequivocally to those from plant products. In the case of plant products, the contributions to acidification are due more to use of fossil fuels than to emissions of ammonia from manure. If processing and/or distribution are responsible for a significant fraction of life cycle consumption of fossil fuels, then these processes may make a significant contribution to the acidification load of plant products.

4 Environmental impacts of alternative production and distribution systems

4.1 Introduction

Many life cycle analyses of foodstuffs include comparisons of the environmental impacts of different production systems. Most of these comparative analyses focus on alternative systems of primary production, and very many of these again compare organic and conventional systems. Some studies have compared alternative systems of animal husbandry within the conventional school, for instance intensive vs. extensive dairy farming (Haas (2001)), different pig-rearing systems (Basset-Mens and van der Werf (2005), Williams et al. (2006)), egg production by battery vs. free-range hens (Williams et al. 2006) or production of beef from dairy vs. sucker herds (Casey and Holden 2006, Williams et al. 2006, Hirschfeld et al. 2008). Regarding plant products, we have already cited several studies of greenhouse vegetable production that include comparisons of alternative energy systems. There are also examples of comparative LCAs of open-field crop production by two or more non-organic systems, focusing for example on the effects of different tillage practices (Zaher et al. 2010) or different rotations (Hayer et al. 2010).

Alongside the comparative studies that focus on alternative primary production systems, there are also some that have focused on distribution systems. We have already cited several studies that compared the environmental impacts of fruits or vegetables from near and distant sources. Saxe et al. (2010) generalised the issue by estimating the effect on the Danish food carbon footprint of substituting Danish foodstuffs for imports, as far as practically possible. (Rather surprisingly, they found that this would increase the food carbon footprint by 5 %. However, if a "New Nordic diet" were adopted, the import substitution would make no difference.) So far, not very many studies appear to have analysed the consequences of alternative logistics at the national or local level. Examples of such studies – all of which indicate that local or regional supply systems can reduce GHG emissions from distribution – include Pirog et al. (2001), Wallgren (2005), Kulak (2010) and Marletto and Silling (2010).

A few authors have also carried out comparative analyses of processing systems (meaning systems that lead to essentially identical products – as opposed for instance to comparisons of fresh vs. frozen or tinned products.) The most frequent focus of such studies appears to be the impact of large-scale vs. small-scale processing. One example is Høgaas Eide (2002), who compared the impacts of liquid milk production by a small, a medium-sized and a large dairy in Norway, and found the largest to be the most environmentally efficient. Braschkat et al. (2003) and Andersson et al. (1998) compared bread from small and large-scale bakeries in Germany and Sweden respectively, and came to opposite results – in the German case, the large bakery had the least impact, while in the Swedish case, that honour fell to the small enterprise. Thomsson and Wallgren (2005) analysed the impacts of small-scale processing as well as the associated distribution systems in the vicinity of Stockholm. We shall return to the latter study.

The fact that the comparative environmental performance of conventional and organic farming systems has attracted particular attention is unsurprising. Environmental concerns are in many cases the prime motive for those who choose to farm organically or to buy organic food. It is reasonable to ask whether, or under what circumstances, organic production actually results in smaller environmental impacts. The following sections will focus on that issue.

4.2 Carbon footprints of organic vs. conventional foods

In a recent conference presentation, Hermansen et al. (2010) summarised the results of 11 comparative analyses of conventional and organic foods, containing 28 pairs of results for individual products. In 20 of the 28 cases, the organic products were found to have smaller carbon footprints than their conventional counterparts, while the conventional products came off best in 8 cases. Out of 15 comparisons of plant products, there were only two in which the conventional products had been found most climate friendly. The results for animal products were more evenly split.

In a meta-analysis of comparative studies of organic and conventional production, Mondelaar et al. (2009) concluded that the material did not indicate a clear difference either way between the climate friendliness of the two systems. The authors pointed out that the avoidance of artificial fertilisers and pesticides in organic production, along with less use of feed concentrates, had a downward influence on GHG emissions in organic production. However, higher methane emissions from ruminants – due to a smaller fraction of concentrates in their feed – and more fuel consumption for mechanical weed control were among factors with an opposite effect. It is worth adding, although Mondelaers et al. do not make the point explicitly, that if yields are lower, as they tend to be in European organic agriculture, then fuel consumption for other operations such as tillage and harvesting will also tend to be higher per kg of product, other things being equal. Refsgaard et al. (1998) found that diesel consumption *per hectare* in organic and conventional agriculture in Denmark was almost equal, with organic farmers using somewhat more fuel to spread manure and conventional farmers somewhat more for tillage.

Lynch et al. (2010) reviewed 120 studies of relevance to the issue of carbon footprints of conventional vs. organic products. Many of the studies reviewed were actual comparative LCAs, but most were not. They included many studies in which the research questions were limited to aspects of agronomic practice or emissions of individual GHGs from particular processes. Frequently, results were presented only as emissions per unit area, not per kg of product. However, Lynch et al. found that a majority of the studies indicated that organic production led to smaller GHG emissions than conventional, although the tendency in the case of GHG emissions was less pronounced than it was for energy use. They also pointed out that while most studies have found that emissions per hectare are smaller in organic than in conventional systems, the tendency for emissions per kg of product also to be smaller is clearer in North American than in European studies. The reason for this is that differences in yields are smaller in North America than in Europe, where conventional farming is often more intensive. In contrast to Mondelaer et al. (2009), Lynch et al. (2010) found that differences in fuel consumption for tractor operations did not much affect the results regarding emissions per unit of product.

In their review of the literature, Nymoen and Hille (2010), like Hermansen et al. (2010), found that a majority of comparative LCAs of plant foods indicated that organic products had lower carbon footprints than conventional products, although this did not apply to vegetables. In the case of milk, they found that most studies showed only small differences between organic and conventional products, whereas results for meat diverged more markedly, some studies indicating that GHG emissions from organic production were significantly higher and others the opposite. The latter findings are also in agreement with Hermansen et al. (2010), which is not very surprising, since there is a high degree of overlap in the primary sources used.Several new comparative LCAs of organic and conventional foods have been published during 2010 and early 2011. The table below draws together results from 24 comparative studies, which contain 72 pairs of results for individual products. (In cases where studies include results for several variants of a product, e.g. more than one kind of bread or cheese, these are counted as a single result.) All of the studies represented in the table below are from North-Western or Central Europe. A number of comparative LCAs of conventional and organic products are available from other regions, especially North America, but are not included in the table as differences inter alia in climate and farming systems may lead to different tendencies in the results, as noted by Lynch et al. (2010). As always, it is important to note that downstream system boundaries vary, which can influence the magnitude of relative differences in carbon footprints. This applies particularly to plant products, since emissions downstream of the farm gate may make up a large share of the carbon footprint in their case. If the downstream emissions per kg of product are identical for conventional and organic products, and LCAs include these emissions, then any difference in emissions up to the farm gate will become relatively smaller the further downstream the analysis goes. The effect of including processing is illustrated for example by Fritszche and Eberle (2007) who found that fresh organic potatoes had a 30 % smaller carbon footprint than conventional potatoes, and other fresh organic vegetables a 15 % smaller footprint, whereas the differences for frozen and preserved products were well down into the single digits.

There are also other methodological differences between the studies, including varying assumptions about N_2O emissions from primary production and the treatment of net carbon exchange between the soil/vegetation system and the atmosphere, which is disregarded in most studies but included, more or less, in a few of the most recent. Most of the studies are based on direct process analysis, but a few are mainly modelling exercises. Both Mondelaer et al. (2009) and Lynch et al. (2010) discuss the issue of how such methodological differences may affect comparisons between organic and conventional products, and advise that results from the existing literature should be interpreted with caution.

In addition to results for individual food products, the table includes one set of results for a whole national diet. Saxe et al. (2010) estimated the effect of increasing the share of organic food in the Danish diet from about 6 % to 80 %. The effect was estimated under the assumption that the composition of the diet would otherwise remain unchanged, and also under two alternative assumptions.

Table 4.1 Relative carbon footprints of organic vs. conventional food products, according to various sources
Note: Downstream system boundaries vary

	> 10 % smaller for organic products	0-10 % smaller for organic products	0-10 % larger for organic products	> 10 % larger for organic products
Cereals	LCA Food 2003 (wheat, rye, barley, oats, Denmark) Hirschfeld et al. 2008 (wheat, Germany) Fritzsche and Eberle.2007 (wheat, Germany) Lindenthal et al. 2010 (bread, Austria)	Williams et al. 2006 (wheat, UK)		
Vegetables, and fruit	Cederberg et al. 2005 (parsnips, Sweden) Fritzsche og Eberle. 2007 (potatoes, open field tomatoes, other vegetables, Germany) De Backer et al. 2009 (leeks, Belgium) Lindenthal et al. 2010 (potatoes, onions, carrots, open field tomatoes, zucchini. tomatoes, cucumbers, strawberries, Austria)	Williams et al. 2006 (potatoes, UK)	Kok et al. 2001 (potatoes, Netherlands) Blonk et al. 2010 (pineapples from Costa Rica to the Netherlands)	Mattsson et al. 2001 (potatoes, Sweden) Cederberg et al. 2005 (potatoes, Sweden) Halberg et al. 2006 (greenhouse tomatoes, carrots, Denmark) Bos et al. 2007 (potatoes, leeks, salat, Netherlands) Blonk et al. 2010 (cauliflower, greenhouse tomatoes, Netherlands) Vermeulen 2011 (greenhouse tomatoes, Netherlands)

	> 10 % smaller for organic products	0-10 % smaller for organic products	0-10 % larger for organic products	> 10 % larger for organic products
Legumes	Bos et al. 2007 (peas, beans, Netherlands)			
Sugar	Bos et al. 2007 (sugar beet, Netherlands)			
Oilseeds	LCA Food 2003 (rapeseed, Denmark)	Williams et al. 2006 (rapeseed, UK)		
Dairy products and eggs	Kok et al. 2001 (milk, Netherlands) Fritzsche and Eberle. 2007 (eggs, Germany) Lindenthal et al. 2010 (milk, yoghurt, cheese, butter, Austria) Hortenhuber et al. 2010 (milk, Austria)	Cederberg and Flysjö 2004 (milk, Sweden) Cederberg et al. 2007 (milk, Sweden) Hirschfeld et al. 2008 (milk, Germany) Bos et al. 2007 (milk, Netherlands) Fritzsche and Eberle. 2007 (milk, cheese, butter, yoghurt, eggs, Germany)	Haas 2001 (milk, organic vs. intensive conventional farming, Germany) Thomassen et al. 2007 (milk, Netherlands) Corson and Aubin 2010 (milk, France)	Haas 2001 (milk, organic vs. extensive conventional farming, Germany) Williams et al. (milk, eggs, UK)
Meat	Williams et al. 2006 (pig meat, sheep meat, UK) Hirschfeld et al. 2008 (pig meat, Germany) Halberg et al. 2010 (pig meat, Denmark, including net C sequestration) Fritzsche and Eberle 2007 (cattle meat, chicken, Germany)	Hirschfeld et al. 2008 (cattle meat, sucker herds, Germany) Fritzsche and Eberle 2007 (pig meat, Germany)	Cederberg and Darelius 2000 (cattle meat, Sweden)	Basset-Mens and van der Werf 2005 (pig meat, France) Williams et al. 2006 (cattle meat, chicken, UK) Hirschfeld et al. 2008 (cattle meat, dairy herds, Germany)
Whole national diet			Saxe et al. 2010 (current diet, officially recommended diet and «New Nordic diet"): 5-7 % more emissions in all cases after organic substitution.	

All of the studies of cereal products and of oilseeds indicate that organic production entails less GHG emissions per unit product than conventional. In these cases, the higher emissions from conventional production are mainly due to (a) production of artificial fertilisers and possibly (b) elevated N_2O emissions, which result from higher total applications of nitrogen per hectare, though not *necessarily* per kg product. The importance of the latter factor is also sensitive to assumptions about the soil nitrogen cycle. On-farm fuel consumption is likely to be higher per unit product in organic than in conventional production of cereals and oilseeds, at least in Europe. Williams et al. (2006) seem to have estimated fuel consumption per hectare to be the same in organic as in conventional wheat production, but they estimated the organic yield to be only one-third of the conventional. Hirschfeld et al. (2008) estimated that fuel consumption per hectare was some 20 % less in organic systems, but that the yield in "average" organic systems was barely half of the conventional, while Lillywhite et al. (2007) estimated that fuel consumption was some 13 % less in UK organic wheat production than in conventional systems. If yield differentials are as large as those estimated by Hirschfeld et al. or Williams et al., then both fuel consumption and N_2O emissions per hectare must obviously be *very much* less in organic systems if these factors are to exert a downward influence on emissions per kg of product. However, the avoidance of artificial fertilisers in organic systems appears to decide the issue for cereals and oilseed rape in favour of organic production.

For other plant products, results are spread across the whole scale from much lower emissions from organic production to much lower emissions from conventional production. All of the three studies that include results for greenhouse tomatoes show significantly higher emissions from organic production. For this product, the main contribution to GHG emissions comes from energy use for lighting and heating greenhouses. If this is the same per m2 in organic as in conventional systems, and the energy sources are also the same, then emissions per kg will essentially be an inverse function of yield levels, which decide the issue in favour of conventional production. In analyses of open field crops, data or assumptions about the amounts of manure applied in organic systems, and the resulting emissions of N_2O, appear to have a strong influence on the divergent results. Halberg et al. (2006) assumed that total nitrogen applications were much larger in organic than in conventional carrot production, resulting in four times higher (!) emissions of N_2O per kg of product in the organic case. Obviously, some authors who have concluded that organic production causes lower emissions have done so at least partly because their figures for total nitrogen applications were lower in the organic than in the conventional cases. In the conventional parsnip operation that Cederberg et al. (2005) analysed, very large amounts of slurry were applied, and the total application of N was much larger than in the organic system. Likewise, the conventional leek system analysed by De Backer et al. (2009) involved much larger applications of N and well over twice the emissions of N_2O per hectare of the organic system. Emissions per kg were almost twice as high for conventional leeks as for organic. The systematically better results for a range of organic products than for conventional counterparts in the study by Lindenthal et al. (2010) are due partly to another factor, namely the assumption that there was net sequestration of carbon in soils in the organic systems, while the opposite was true of conventional systems.

For dairy products, many studies have found the difference in emissions per unit product between conventional and organic systems to be quite small. Hortenhuber et al. (2010), who analysed eight different organic and conventional systems, found that the ranges of emissions per kg overlapped. This study has been placed in the far left column of the table above only by a hair's breadth: the average for organic milk was 11 % less than the average for conventional milk. In the case of milk production, methane emissions from enteric fermentation tend to be higher per unit product in organic than in conventional systems, while emissions from production of feed tend to be lower.

The results for meat vary widely. Given that comparative studies of cereal and oilseed production indicate that organic production causes less GHG emissions per kg than conventional, one might perhaps expect that results for products from animals fed largely or entirely on concentrates – i.e. eggs, pig and chicken meat – would also turn out to the advantage of organic production. Nevertheless, Basset-Mens and van der Werf (2005) reached the opposite conclusion in their study of French pig meat systems, as did Williams et al. in their analyses of egg and chicken meat production (but not of pig meat) in the UK. In complete contrast to these results, Fritzche and Eberle (2007) found that organic production of pig meat and *especially* of chicken meat caused less emissions of GHGs than conventional production in Germany. Halberg et al. (2010) also found that emissions from organic production of pig meat were smaller than from conventional production, but only if net sequestration of carbon in soils was included in the analysis, which has not been the case in most studies hitherto. Without this factor, the organic systems analysed by Halberg et al. would have caused 7-22 % higher GHG emissions per kg of meat than a conventional system. The factor that counterbalances smaller emissions per unit of feed in organic pig and chicken meat and egg systems appears to be that the animals require more feed for a given weight gain than in conventional systems. This is presumably an effect of stricter animal welfare requirements in organic systems: the animals expend more energy through physical activity and live longer before being slaughtered. However, the magnitude of the difference in feed-to-weight-gain ratios between organic and conventional systems varies considerably from one study to another.

The signs and sizes of differences between the carbon footprints of organic and conventional products at the *retail* stage could be affected not just by differences in methods of primary production, but also by differences in downstream processes. At present, there are much fewer organic than conventional primary producers in all countries, and the number of processing plants with organic product lines may also be quite small. This could have the effect that organic products have to travel further, on average, from the farm to processing and from there to retailers. Meisterling et al. (2009), in a comparative study of wheat flour production in the USA, found that although organic wheat at the farm gate had a smaller carbon footprint than conventional, the difference would disappear if organic flour had to travel 420 km further than the conventional. For fruit and vegetables, the sensitivity to transport distance is greater than for cereals. On the other hand, it could be that organic farmers are more likely than conventional farmers to want to market their products locally, and/or to process them on or near the farm, than are conventional farmers.

An extreme case of localized organic production was analysed by Kulak (2010). This concerned organic ”community supported agriculture” in a district of London. The products were distributed from the field to consumers within a radius of a few kilometres, for good measure by an electric vehicle, which did a weekly run of 25 km. For all 13 products analysed in Kulak’s study (12 vegetables and apples) GHG emissions per kg were lower – and in many cases much lower – than for corresponding products delivered to supermarket in a conventional reference system. A slightly less extreme case was studied by Thomsson (2005). This concerned transport of grains+bread, milk, meat and vegetables from organic farms around Järna (south of Stockholm), via local processing (except for the vegetables, which were not processed) and on to consumers, mainly in Greater Stockholm. Overall transport distances were several tens of kilometres for most products, and longer in the case of meat (live animals were transported some 80 km to slaughter and the meat back before it was locally distributed). In the Swedish case, transport was by diesel vehicles. Nevertheless, GHG emissions from processing and distribution were significantly smaller than in conventional systems that had been analysed in previous Swedish studies – except in the case of meat, where there was little difference. However, energy use for *transport alone* (as opposed to processing) was higher in the organic-localised system than in conventional reference systems, not just in the case of meat but also of bread. The main reason for this was that small quantities were transported at a time in the organic-localised system. The use of small vehicles, and poor capacity utilisation even in spite of this, led to much higher energy use (and therefore higher emissions) per ton kilometre than in large-scale systems. So far, no environmental analyses of the logistics of alternative distribution systems such as farmer's markets or community supported agriculture, nor of the downstream logistics of organic products in general, appear to have been carried out in Norway.

The finding by Saxe et al. (2010) that increasing the organic share of Danish food consumption to 80 % would slightly increase its overall carbon footprint deserves a comment. As Audsley et al. (2009) pointed out with reference to the UK, a transition to wholly or mainly organic diets in Europe would probably have to be accompanied by a reduction in the share of animal foods in the diet, if net imports of food were not to increase. This follows from the fact that yields are smaller in organic agriculture. A somewhat larger share of an all-organic diet would have to consist of foods that are "land-efficient", if the available agricultural land were still to produce enough dietary energy. The scope for expanding agricultural land in Europe is small, and any expansion is likely to come at an environmental cost. Saxe et al. (2010) present a scenario in which consumption of animal foods is only moderately reduced (meat consumption per capita falls by 31 % from the current Danish level, which is among the highest in the world, but some of this is compensated by increased consumption of dairy products). If a transition to this "New Nordic diet" were *combined* with a transition to 80 % organics, then the Danish food carbon footprint would be reduced by 2 % from the current level, according to calculations in Saxe et al. (2010). However, the study does not account for land use. Had a requirement been introduced that the amount of land demanded by Danish food consumption must not increase, then consumption of animal products would probably have had to be reduced somewhat more than the "New Nordic diet" allows for, and the reduction in the food carbon footprint would also have been larger.

If one accepts the proposition that any transition to organic diets must be accompanied by other adjustments in diet, so that land requirements do not increase, then comparisons of GHG emissions per hectare in organic vs. conventional systems become more meaningful than they may otherwise seem. If the number of hectares is given, then absolute emissions depend simply on emissions per hectare. With very few exceptions, comparisons on that basis show smaller emissions from organic than from conventional systems. Only 3 of the 72 results in table 4.1 imply higher emissions per hectare from organic than from conventional production.

4.3 Other environmental impacts of organic vs. conventional production

Energy use

Just over half of the studies in table 4.1 present results for energy use, either in terms of end use or of primary energy. Based on the studies that do so, there is a stronger tendency for organic products to require less energy per kg, than for the same products to have smaller carbon footprints per kg. Vegetables are the exception. According to Williams et al. (2006), organic wheat production in the UK demands significantly less energy per kg than conventional. (Meisterling et al. (2009) arrived at the same result in a study of US wheat production, as did Hoeppner et al. (2006) in a Canadian study.) However, Mattsson (2001) found that energy use per kg in Swedish organic and conventional potato production was almost equal, whereas Cederberg et al. (2005) found that organic potatoes required significantly more energy. They also found that organic production of parsnips required somewhat more energy than conventional production. Williams et al. found that organic potato production required just marginally (2 %) more energy per kg than conventional. Bos et al. (2007) arrived at results regarding energy requirements that were in line with those for carbon footprints – in other words, Dutch organic production of potatoes, leeks and lettuce required more energy than conventional production, whereas the opposite was true of peas, beans and (especially) sugar beet. In the case of greenhouse tomatoes, there can be little doubt that comparisons of energy requirements would exhibit the same tendency as those of carbon footprints – i.e. to show higher figures for organic products – since the carbon footprints are overwhelmingly due to energy use. The major differences between results for carbon footprints and for energy requirements concern animal products. In the case of milk production, not only Cederberg and Flysjö (2004), Cederberg (2007) and Bos et al. (2007), but also Williams et al. (2006), Haas (2001) and Thomassen et al. (2007) found that organic production required less energy per kg than conventional. The difference in favour of organic production varied from 10-40 %. The reason why organic dairy production does not compare quite as favourably with conventional for GHG emissions as for energy use is that it tends to entail higher methane emissions from enteric fermentation. Cederberg and Darelius (2000) also found that energy requirements for organic beef production were less than for conventional, although they arrived at the opposite result regarding GHG emissions. In the case of pig meat, Basset-Mens and van der Werf (2005) found that energy requirements, like GHG emissions, were higher in organic than in conventional production. In contrast, Williams et al. (2006) found that organic production required the least energy not only in case of pig meat, but also of cattle and sheep meat. Only organic chicken meat required more energy than the conventional product.

Land use

Without exception, all of the studies in table 4.1 show that organic products demand more land per kg than their conventional equivalents. The differences are generally greatest in the case of cereals and animal products based on feed concentrates, and somewhat smaller for vegetables, legumes and products from ruminants. That the differences in yields between organic and conventional vegetable crops are less than for cereal crops is at least partly due to the fact that organic vegetables get larger applications of manure and/or compost. Vegetables occupy a very small fraction of Europe's agricultural land, so it is possible to obtain the necessary quantities of compost and manure for high application rates on this limited area. Because they are high-value crops, it also pays to do so. It would be impossible to duplicate such high application rates across the vastly larger area devoted to cereals. Organic cereal crops usually have to make do with less nutrients, which results in lower yields compared to their conventional equivalents, but also in lower N_2O emissions. Legumes can fix their own nitrogen, so the avoidance of synthetic fertilisers in organic systems makes less difference to them than to other crops.

Eco-toxins

Only a few of the studies in table 4.1 present results for eco-toxins. Since avoidance of synthetic pesticides is a basic requirement in organic agriculture, it seems reasonable to expect that organic production will cause less toxic pollution per unit product than conventional. Williams et al. (2006) and Cederberg (2007) limited their inventory of eco-toxins to pesticides, so it is hardly surprising that organic production compared very favourably with conventional in their analyses. Haas et al. (2001) analysed contributions to eco-toxicity from organic, intensive conventional and extensive conventional beef production systems in Allgäu i Germany, but found them so negligible in all three cases that they did not publish the results. Basset-Mens and van der Werf (2005) rather surprisingly found that organic pork production contributed more to terrestrial eco-toxicity than conventional, a result they attribute mainly to feed production, but do not explain in more detail. De Backer et al. (2009) found that organic leek production contributed 200 times less to terrestrial eco-toxicity than conventional, and four times less to human toxicity.

Eutrophication

One might expect that organic production would generally contribute less to eutrophication than conventional, since the avoidance of artificial fertilisers means that less nitrogen and phosphorous can be applied to the soil and later turn up as nutrient runoff. However, some factors may exert an influence in the opposite direction, particularly if an organic system is compared to a conventional system in which *only* artificial fertilisers are applied. Nutrients in animal manure or compost do not become available to growing plants as fast as those in artificial fertilisers. It can be more difficult to spread applications of organic fertilisers optimally (in time as well as space) and leakages from storage of such fertilisers are also more likely. In organic systems that utilize "green manure", large surpluses of nitrogen may arise in the first year – i.e. nitrogen which will not be take taken up by the next crop. Mondelaers et al. (2009), in their meta-analysis of comparative studies of organic and conventional production, found a significant tendency for nitrogen runoff to be less per hectare in organic than in conventional systems, but no significant difference in runoff per kg of product. In the case of phosphorous runoff, there were fewer sources. Although most of these indicated that runoff *per hectare* was less from organic farms, the uncertainty even on this point was greater than in the case of nitrogen. One study not cited by Mondelaers et al. is Granstedt et al. (2005), who compared nitrogen and phosphorous surpluses in conventional agriculture in seven Baltic countries with those from a sample of organic mixed farms, producing animal as well as plant products for sale, i.e. farms that were able to utilize manure optimally. On average, the mixed organic farms had a slightly *negative* surplus of P – i.e. a deficit – and less than half the N surplus per hectare of conventional farms.

Only seven of the comparative studies in table 4.1 give results for contributions to eutrophication per kg of product, and these are highly divergent. This is partly due to differences in data or assumptions regarding the quantities of N and P applied to crops and the ways in which they are applied, in conventional as well as in organic systems, but especially in the latter. In addition, there are differing assumptions about subsequent processes, quite apart from the fact that real differences in soils, climate and drainage conditions among the farms or regions studied will obviously influence the results. LCA Food (2003) indicates that organic wheat and barley in Denmark contribute less to eutrophication than conventional equivalents, while the reverse is true of oats, rye and rapeseed. Williams et al. (2006) found that not only organic rapeseed, but also organic wheat in the UK contributed much more to eutrophication than conventional equivalents. This result was based on the assumption that the organic rotations included green manure. For potatoes, Williams et al. found that the contribution to eutrophication from organic production was just slightly higher than from conventional. Halberg et al. (2006) found that organic carrots in Denmark contributed four times as much to eutrophication as conventional carrots. This is the same differential that they arrived at for N_2O emissions, and likewise mainly due to the fact that the applications of manure in the organic system were very large (175 kg N/ha). De Backer et al. found that contributions to eutrophication from organic and conventional leek production in Belgium were almost equal per unit product. They assumed that total applications of N were smaller in the organic system, though applications of N in organic fertilisers were about the same. Results concerning animal products also diverge. Williams et al. (2006) found that contributions to eutrophication per unit product were higher from organic than from conventional animal products in all cases except pig meat, in whose case organic systems performed significantly better than conventional. Cederberg et al. (2007) found a weak indication that organic milk production in Northern Sweden might contribute slightly more to eutrophication than conventional production, and Basset-Mens and van der Werf reached a similar result for pig meat production in France. Thomassen et al. (2007) and Haas et al. (2001) found that organic milk, from the Netherlands and Southern Germany respectively, contributed significantly less to eutrophication than its conventional equivalent.

Acidification

Results regarding the relative contributions to acidification per unit of organic and conventional products are just as divergent as those regarding contributions to global warming or eutrophication. LCA Food (2003) indicates that there are only minor differences between the contributions to acidification of organic and conventional cereal and rapeseed crops. The signs also vary. Williams et al. (2006) found that conventional production of rapeseed, potatoes and pig meat in the UK contributed significantly more to acidification per unit product than organic production, whereas there was little difference in the case of wheat, and conventional production performed significantly better in the cases of milk and eggs, as well as of cattle, sheep and chicken meat. The results for ruminant products were partly due to high proportions of clover in organic meadows, resulting in higher emissions of ammonia. Halberg et al. (2006) found that organic carrots in Denmark contributed significantly more to acidification than conventional carrots , while de Backer et al. (2009) found that organic leeks in Belgium contributed less to acidification than conventional leeks. The difference in signs between these two studies repeats itself – it is the same for acidification as for global warming potential and eutrophication, and in all cases mainly a consequence of very different data regarding manure applications. Both Cederberg et al. (2007) and Thomassen et al. (2007) found contributions to acidification from organic milk production to be insignificantly higher than those from conventional production, while Haas et al. (2001) found that emissions from conventional milk production were slightly higher than from organic production. Basset-Mens and van der Werf (2005) found that organic pig meat production contributed less to acidification than one of the two conventional systems they analysed, but more than the other conventional system.

4.4 Summary

All of the sources reviewed agree that organic production of cereals and of rapeseed contribute less to global warming, per unit product, than conventional production. For other plant products as well as for animal products, however, the results diverge widely, with almost equal numbers showing that organic systems do better than conventional and the opposite.

Only one study was found that includes an estimate of the effect on GHG emissions of changing to a largely organic diet. This finding in this Danish study was that it would lead to slightly increased emissions, if the pattern of food consumption in Denmark otherwise remained the same. However, there are grounds for contending that a general transition to organic food consumption in Europe would have to be accompanied by reduced consumption of animal foods. The combined effect could be a reduction in GHG emissions.

A majority of results in the studies reviewed show that organic systems require less energy per unit of production than conventional, though vegetable production may be an exception. On the other hand, these European studies provide no exception to the rule that organic systems require more land per kg of product.

Few of the studies include analyses of contributions to eco-toxicity, and the available results give no grounds for general conclusions beyond the obvious point that synthetic pesticides are not permitted in organic agriculture. The comparative studies provide more results on contributions to eutrophication and to acidification, but these are highly divergent in both cases. They give no grounds for general conclusions about whether organic or conventional systems tend to contribute most per unit product to these environmental impacts.

As a corollary to the fact that yields are lower in organic than in conventional agriculture, organic production consistently performs better when environmental impacts are compared per unit area, than if they are compared per unit of product. An overwhelming majority of results show smaller GHG emissions per hectare from organic than from conventional systems. If one wishes to assess the consequences of a general transition to organic production, and assumes that this must involve a change in diets rather than an expansion of agricultural area, then comparisons of environmental impacts per unit area become relevant to the assessment.

5 Will new insights about process emissions from agriculture change our understanding of food carbon footprints?

Of the many methodological differences among analyses of the carbon footprints of foodstuffs, one group deserves particular attention in 2011. This is because we can speak of ongoing trends, meaning that authors of some of the most recent studies have made methodological choices which were rare or unknown in the earlier literature, and that we can expect to see these new approaches applied more often in the future. The trends concern choices of assumptions and physical system boundaries that affect results for process emissions from primary production. They may lead to adjustments in our understanding not only of the size of the aggregate carbon footprint of food consumption, but also of the relative contributions of different foods, and/or the relative impacts of different production methods.

5.1 Process emissions of methane and nitrous oxide

In many LCAs of food products, estimates of nitrous oxide emissions, and sometimes of methane emissions, are calculated by using models based on guidelines for national reporting issued by the Intergovernmental Panel on Climate Change (IPCC[9]), or on the way in which these guidelines have been adapted by authorities in the country the study concerns. This practice is recommended in international standards for life cycle analysis (ISO 14040, 2006) as well as carbon footprinting of products (PAS 2050) (BSI 2011). The IPCC guidelines allow countries to estimate emissions from various processes in several ways: they may either apply standard (Tier 1) emission factors, or factors that take more account of specific conditions in the country and sector concerned (Tier 2 or 3), as available information permits. Countries have in fact practiced the guidelines in somewhat different ways, and the more refined methods have been applied somewhat more frequently as knowledge has improved. The methods used in life cycle analysis and in carbon footprinting do not always adhere to IPCC guidelines. In the cases of CH_4 emissions from enteric fermentation and from manure management, many authors have used estimation procedures based on independent sources (e.g. Haas et al. 2001; Bos et al. 2007; Cederberg et al. 2007; Thomassen et al. 2007; Hirschfeld et al. 2008) rather than those used in national reporting of GHG emissions. If estimation procedures in the most recent studies are based on better information than those used in earlier studies, then this may of course lead *either to higher or to lower* estimates of N_2O or CH_4 emissions from particular processes in a given country, even if nothing should have changed in the real world.

Another point is more relevant to the question of *trends* in estimates of process emissions from agriculture. The IPCC guidelines for estimation of both CH_4 and N_2O emissions, as well as the standard (Tier 1) emissions factors for some processes, were revised in 2006[10]. In spite of this, national reports have hitherto continued to follow the 1996

[9] http://www.ipcc-nggip.iges.or.jp/public/gl/guidelin/ch4wb1.pdf
[10] http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

guidelines (Leip et al. 2010), and the ISO 14040 and PAS 2050 standards still recommend adhering to those guidelines (Blonk et al. 2010). However, the 2006 guidelines and emissions factors have been applied in some recent environmental analyses of food products, and it is reasonable to expect that this will become increasingly common, at least until the IPCC issues further revisions. In 2007, the IPCC also changed the recommended weighting factors for converting CH_4 and N_2O emissions to CO_2 equivalents, or more precisely CO_2-equivalent global warming potential in a 100-year perspective (GWP100), the measure that is almost invariably applied in life cycle analyses. Although the last-mentioned change does not affect estimates of emissions of individual gases from agricultural processes, it does affect assessments of their contributions to global warming.

Methane

The dominant source of methane emissions from agriculture, in Norway as elsewhere in Europe, is enteric fermentation in ruminants. The IPCC (Tier 1) emissions factors of 2006 lead to somewhat higher estimates of emissions from enteric fermentation in cattle (though not in sheep) than the corresponding 1996 factors. For cattle other than dairy cows, estimated emissions increase by some 20 %; for dairy cows the increase is somewhat less. However, most LCAs apply more refined methods than the IPCC's Tier 1 to estimate emissions from enteric fermentation, taking account inter alia of feed composition (as in IPCC Tier 2), and often relying on independent national sources for emissions factors, rather than IPCC recommendations. Therefore, it is not obvious that revised IPCC factors concerning enteric fermentation will have any discernible effect on future estimates of the carbon footprint of milk or cattle meat production. However, new conversion factors for methane to CO_2 equivalent may. Until 2001, the IPCC estimated the GWP100 of CH_4 as 21 times that of CO_2; in 2001 this factor was revised to 23, and in 2007 to 25. Until around 2007, most environmental analyses of foodstuffs nevertheless seem to have applied the factor from which the IPCC departed in 2001 (CH_4= 21 times CO_2). In the most recent studies, the factor of 25 has been more frequently applied. According to many studies, methane is responsible for about half of the life cycle emissions of ruminant products. In that case, changing its CO_2 equivalent factor from 21 to 25 increases the total contribution of such products to global warming by about 10 %. Because methane, according to most comparative studies, is responsible for a somewhat higher share of the total GHG emissions from organic than from conventional production of ruminant meat and milk, the change could also have a (minor) effect on comparisons between organic and conventional systems, to the advantage of the latter. Methane contributes much less to the overall carbon footprints of most other foodstuffs. Rice is the main exception. Pig meat *can* also be an exception, if manure is kept in open lagoons, as is often the case in the USA (Thoma 2010) and in Australia (Price 2010), but not in Europe.

Nitrous oxide

Nitrous oxide plays a much bigger part in the contributions to global warming of most food production systems – virtually all except production of rice and ruminant products - than does methane. In estimating emissions of N_2O, most studies do rely on IPCC recommendations, and not on independent sources as often happens in the case of CH_4. As mentioned above, these recommendations were revised in 2006. The table below, which was taken from Blonk et al. (2010), compares the latest IPCC recommended (Tier 1) factors relevant to estimating N_2O emissions, with the corresponding factors issued by the IPCC in 1996. The four first lines show the amounts of N that are assumed to be compounded to N_2O per kg of N available from the source in question. The other lines concern factors that indirectly affect N_2O formation.

Table 5.1 IPCC (Tier 1) factors for calculating N_2O emissions. 1996 and 2006 versions. Source: Blonk et al. (2010)

Emissions from:		IPCC 1996	IPCC 2006
Applications of artificial fertilisers, manure and crop residues	kg N_2O-N/kg N	0,0125	0,010
Biological N fixation	kg N_2O -N/kg N	0,0125	–
Urine and excrement deposited by animals	kg N_2O -N/kg N	–	0,0200
Use of peat soils (temperate climate)	kg N_2O -N/ha/år	5	8
Volatilisation from artificial fertilisers	kg NH_3-N/kg N	0,10	0,10
Volatilisation from manure	kg NH_3-N/kg N	–	0,20
Volatilisation from urine and excrement	kg NH_3-N/kg N	0,20	0,20
Nitrate leaching	kg NO_3^--N/kg N	0,30	0,30
Emission factor for volatilisation	kg N_2O -N/kg N	0,0100	0,0100
Emission factor for nitrate leaching	kg N_2O -N/kg NO_3^--N	0,0250	0,0075

For most modern farming systems, using the 2006 factors is likely to lead to somewhat lower estimates of nitrous oxide emissions than the 1996 factors would have led to. Most analyses based on the latter have found the main sources of N_2O emissions to be fertiliser applications and leached nitrate. As the table shows, the 2006 emission factor for the former is down by 20 % on the 1996 figure, while the 2006 factor for the latter is down by all of 70 %. In addition, biological nitrogen fixation is no longer assumed to be a source of N_2O emissions (so there will be smaller estimated emissions from cultivation of legumes or rotations with clover). Apart from the factor for peat soils, the revisions that have an opposite effect on N_2O emissions concern direct deposition of urine and excrement by grazing animals, and volatilization (formation of ammonia) from manure, which leads at the next turning to some formation of N_2O. Both of these factors will tend to make estimated emissions from farms with livestock, particularly grazing animals, increase somewhat relatively to those from farms growing crops only, particularly if the latter do not import manure, but rely on artificial fertilisers and/or biological fixation for their supply of nitrogen. What effect using the 2006 factors rather than those of 1996 will have on comparisons of organic and conventional systems is not so obvious. To the extent that organic farms make more use of biological nitrogen fixation, the change on this point will work to their advantage. However, we have observed that there is no agreement among sources on whether organic or conventional systems cause the most nitrate leaching per unit product, and therefore should have most to "gain" from the lower N_2O emission factor for nitrate leaching.

While most environmental analyses of food products do seem to adhere to IPCC methodology in estimating N_2O emissions, it is worth noting that some go beyond Tier 1, for example by using higher emission factors for slurry than for other animal manure.

As in the case of methane, the IPCC revised its estimates of the 100-year global warming potential of nitrous oxide both in 2001 and in 2007. However, the adjustments for N_2O have been smaller, from an original CO_2 equivalent factor of 310 to 296 (2001) and then to 298 (2007). Until about 2007, most studies appear to have used the first and highest factor. Using 298 instead reduces the contribution of N_2O emissions to global warming by 4 % - obviously a minor effect, but one with the same sign (for most farming systems) as the effect of using 2006 emissions factors in preference to those of 1996.

Leip et al. (2010) implemented the 2006 IPCC recommendations in their analysis of emissions from the EU livestock sector, and combined them with a new model for estimating its nitrogen balance, including nitrate leaching. Compared to the figures in national reports under the Kyoto protocol, Leip et al. found emissions of N_2O from all sources in table 5.1 to be smaller – by 89 % in the case of nitrate leaching and between 7-19 % for other sources, except for emissions from use of peat soils, which were 3 % smaller than those shown in national reports, despite the upwardly revised emission factor for this source. This study did not cover food crop production, so we do not know how corresponding nitrogen balance calculations would have worked out for this sector. As mentioned above, however, it seems likely that the contribution of revised IPCC factors to lowering estimates of N_2O emissions could be greater for straight crop production than for the livestock sector.

5.2 Net flows of carbon between soils/vegetation and the atmosphere

Before 2008, few LCAs of food products included estimates of transfers of carbon between soils and the atmosphere, i.e. net releases or sequestration of carbon, as the case might be. Effects of land use change, including changes in carbon stocks in vegetation, have only occasionally been considered in European studies of this kind. However, the picture is rapidly changing. Over the past few years a growing number of studies have included one or both of these factors in their analyses. This trend is not primarily due to new recommendations from the IPCC, but rather to the fact that the fate of carbon stocks in soils and vegetation has attracted growing scientific as well as political interest. The PAS 2050 standard for carbon footprinting (BSI 2008) recommends including effects of land use change in analyses, but not releases or sequestration of carbon that result from ongoing land use.

Where land use change enters into environmental analyses of food products, it is almost invariably as a source of carbon emissions. If it is assumed that some or all of the land used to produce the product will be made available by expanding farmland, then it becomes relevant to account for the carbon stocks in the existing vegetation that needs to be cleared and for any once-off changes in soil carbon stocks that result from the conversion. (In theory, a consequential analysis could involve a *reduction* in agricultural area – that is, if the food in question were assumed to be a substitute for another food that required more land. In that case, the substitution could free up land for afforestation. However, the author knows of no analyses based on such assumptions). While estimating the amount of carbon stored above ground in a forest to be cleared can be fairly straightforward, changes in soil carbon stocks are not. If one assumes that these stocks are in a state of equilibrium under natural vegetation and will reach a new state of equilibrium some time after conversion to farmland, and if one has some way of estimating the size of the stock "before" and "after", then the change can be attributed to production by distributing it over an appropriate period.

However, soil carbon stocks will not necessarily be in equilibrium – neither before the conversion to farmland, nor after a limited transitional period. Soils may act as long-term sinks or sinks for carbon, without any change in land use. Long-established farmland as currently used may be releasing or sequestering an amount of carbon per hectare each year. Some recent studies try to take account of this, but it is not obvious how it should be done, even if one should have a good basis for estimating the annual rate. The simplest method, of course, is simply to divide the amount annually sequestered or released per hectare and attribute it to the crop by dividing it by the yield. An alternative method is to compare the current situation to a reference situation with natural vegetation in place. If it can be assumed that the net annual release or (more probably) sequestration of carbon in that reference situation would have differed from 0, then it could be more relevant to attribute the difference between the current and the "natural" rate of carbon transfer to ongoing production. These questions of principle, and some of the more practical problems that arise at the next turning, are discussed more thoroughly by Blonk et al. (2010).

Where the authors of recent environmental analyses of food products have incorporated carbon flows resulting from land use change and/or land use, they have actually chosen a wide variety of approaches. The table below shows some of them.

Table 5.2 Estimation of net carbon emissions resulting from land use change and land use. Some examples of methodologies in recent environmental analyses of foods

Source	Product, country	Emissions from land use change	Emissions from land use	Important results
Audsley et al. 2009	Whole diet, UK	Estimated from global increase in land used for commercial agriculture. UK share of this equal to current share of global agricultural land that is required to supply UK. Global emissions from land use change to agriculture estimated at 8.5 Gt CO_2eq/yr.	Constant emission factor of 1.43 t CO_2eqha/yr for all agricultural land	Carbon footprint of UK food consumption increases by > 50 % when land use change is included
Blonk et al. 2010	Fruit and vegetables, Netherlands	Recommendation: Either estimate emissions from actual expansion of farmland to produce the product in question, or distribute total annual expansion of farmland in producer regions among crops that contribute to expansion of agricultural area.	Recommendation: Emission factor of 0.4 t CO_2eq/ha/yr for all agricultural land (=loss of sink compared to reference situation with natural vegetation) plus 0.45 t CO_2eq/ha/år for conventional arable land, 0.3 t CO_2eq/ha/år for organic arable, 0 for grassland,	Results not clear as the recommendations were not implemented in analyses of individual products.
Brandão et al. 2010	Wheat, rapeseed,, energy crops, UK	Emissions caused by conversion from reference vegetation (temperate forest) are included – even if actual conversion was several centuries ago – and distributed over number of years for which new crop is expected to be produced.	Net emissions of 0.4 t CO_2eq/ha/yr from conventional production of wheat and rapeseed, but net sequestration of 0.25 t CO_2eq/ha/yr in organic production,	
FAO 2010	Dairy products, global	Only new land to produce soybeans for feed included. Emissions written off over 20 years.	–	Inclusion of land use change increases emissions per kg of milk in Europe by c. 7 %

Source	Product, country	Emissions from land use change	Emissions from land use	Important results
Halberg et al. 2010	**Pig meat, Denmark**	–	Carbon sequestration (0.24-0.39 t C/ha/yr) imputed from nitrogen accumulation in soils.	Inclusion of C sequestration improves relative performance of organic production.
Leip et al. 2010	Livetock products, EU	Estimated for countries where total agricultural area is expanding. Effect distributed among crops in relation to their contribution to expansion. 3 scenarios for magnitude of once-off emissions, based on different reference situations. Emissions written off over 20 years.	Estimated as difference between net emissions in reference situation (natural grassland) and under current land use.	Estimated carbon footprint of livestock products becomes higher than in most previous European studies.
Lindenthal et al. 2010	Fruits, vegetables, bread, dairy products, Austria	–	Net sequestration of 0.4 t CO_2eq/ha/yr in all organic systems and emissions of 0.202 t CO_2eq/ha/yr in all conventional systems.	Improves performance of organic vs. conventional products.
Trydeman Knudsen et al. 2010	Orange juice, Brazil to Denmark	–	Only difference between organic and conventional system estimated, i.e. emissions from conventional system implicitly set at 0. Alternative scenarios for global warming effect (20 or 100 year horizon).	Assumptions reduce emissions from organic oranges by 22-33 % (20 year horizon) but only 5-8 % (100 year horizon). Effect on total life cycle emissions for orange juice much less.
Trydeman Knudsen et al. 2010	Soy beans, China to Denmark	–	Net emissions of 0.1 t CO_2eq/ha/yr in organic system and 0.12 t CO_2eq/ha/yr in conventional .system	Assumptions lead to slightly greater increase in emissions from conventional product, but greater relative increase in emissions from organic product.
Zaher et al. 2010	Wheat, USA Different tillage options	–	Net emissions modelled from factors for fertiliser applications, availability and decomposition rates of plant residues.	Near 0 emissions in case of conventional tillage, net sequestration in all of 4 reduced tillage systems.

In no two of the ten cases in the table was the same – or even approximately the same – methodology chosen. Not even the two papers that are included in a single source (Trydemann Knudsen et al. (2010) on orange juice and soybeans) use the same methods. Analysis of net carbon emissions from land use change and land use is a field still in its infancy, with no robustly established standards.

Just as the methods in LCAs and other environmental analyses of products differ, so do the results of empirical research on carbon fluxes. There is a tendency in the literature to date, as Leip et al. (2010) and FAO (2010) point out, for measurements to have shown smaller net emissions of carbon from soils under grassland than under arable land (smaller net emissions may of course mean more net sequestration). However, the magnitude of this effect differs a great deal from one study to another, as may the signs of the fluxes in both cases. Given that such an effect is real, it is still not clear what consequences it may have for assessments of the relative carbon footprints of different food products. Since grassland mainly serves to produce feed or provide grazing for ruminants, one might imagine that incorporating net carbon emissions from land use in environmental analyses would make ruminant meat and milk fare better in comparison with other products. But this is far from obvious if a large share of the ruminants' feed intake is in the shape of concentrates, produced on arable land. If, in addition to this, the production of concentrates requires a significant amount of land use *change*, as FAO (2010) assume in the case of dairy cows, then the case becomes even more moot.

There is also a tendency, reflected in the assumptions made in several of the sources for table 5.2, for empirical studies to find smaller net emissions of carbon from land under organic than under conventional management. In their meta-analysis of comparative studies of organic and conventional production, Mondelaers et al. (2009) found 19 studies which indicated such an effect, and 7 which did not. After eliminating some studies for methodological or other failings, they still found a significant indication that net carbon emissions from soils were less per hectare under organic than under conventional management. Whether lower yields in organic systems lead to a weaker or a stronger effect *per kg of product* than per hectare, depends on the signs of the net emissions. If the sign of net emissions per hectare is negative in both organic and conventional systems, then the lower the yield, the more each kg of product contributes to reducing net carbon emissions. So in that case organic products have the advantage. If the signs are opposite (net positive emissions from conventional systems and net sequestration in organic systems, as assumed by Brandão et al. (2010) and Lindenthal et al. (2010), then taking account of this will work to the advantage of organic systems, irrespective of whether the denominator is hectares or kg. But if net emissions per hectare are positive in both cases (as assumed by Blonk et al. (2010) in the case of arable land, and by Trydeman Knudsen et al. (2010) in the case of soybeans) then lower yields in organic systems may make emissions per kg of product higher than in conventional systems, even though emissions per hectare are lower. This is the same effect that applies to other (positive) GHG emissions than those from land use.

5.3 Albedo

A stimulating contribution to the debate on food production systems and global warming was recently made by Muñoz et al. (2010), who analysed tomato production in unheated Spanish plastic greenhouses. They found that the carbon footprint of the tomatoes, up to the farm gate, would be almost halved – from 303 g CO_2eq per kg to 168 g CO_2eq – if one took into account that the greenhouses had a much higher albedo, i.e. reflected more solar heat back out to space, than the natural reference vegetation would have done.

Greenhouses are an extreme case in this respect. In the case of greenhouses heated by fossil fuels – in climates where there is less solar energy to reflect than in Spain - the relative effect of incorporating albedo in calculations of the carbon footprint of tomatoes would also have been very much smaller. However, the basic point made by Muñoz et al. can be generalised beyond greenhouses. The albedo of agricultural land will usually be different from that of natural vegetation in the same place. It will also depend on what is being grown and how. It may change a great deal through the year, not least in Norway, where a cornfield may be white with snow in winter (extremely high albedo – but not much sunlight to reflect), black in early spring, green in late spring and early summer, yellow in late summer, then black after harvest before it again turns white. The annual average, insolation-weighted albedo of agricultural land is very likely to differ from that of natural vegetation, and also to vary from one crop to another. Although incorporating this factor in carbon footprint calculations would be unlikely to have a major effect for most products, it might in some cases be more than negligible.

5.4 Summary

Scientific understanding of process emissions of methane as well as nitrous oxide and carbon dioxide from agriculture is gradually improving. New insights, some of which have been enshrined in new guidelines and recommendations from the IPCC, have recently been applied to environmental analyses of foodstuffs, and led to results that differ noticeably from those that would have resulted from following previously established standards in the field. More research in the next few years may well force us to revise some of the tentative conclusions that can be drawn from the body of literature hitherto published. This applies to the aggregate carbon footprint of food consumption as well as the relative contributions of different foodstuffs and the relative climate friendliness of different methods of production.

Until recently, most environmental analyses of food products have relied on guidelines issued by the IPCC in 1996 in calculating process emissions of nitrous oxide and methane, unless they have taken emission factors from independent sources. Until about 2007, most studies also appear to have calculated the global warming potentials of these gases using the conversion factors originally established by the IPCC, although these were revised in 2001 – and again in 2007. Taking account of the most recent IPCC guidelines and conversion factors is likely to result in moderately lower estimates of nitrous oxide contributions to the global warming load of most foods, but moderately higher estimates for methane, than in earlier work. This has already been illustrated by some recent studies.

Changes in carbon stocks in soils and vegetation as a result of land use change as well as of ongoing land use were disregarded in most environmental analyses of food products until very recently. Over the past few years, however, a number of studies have appeared that do attempt to account for these factors. Accounting for land use change makes the carbon footprint of food consumption at the global level greater than if this factor is disregarded, and will do so for as long as global agricultural area goes on expanding. The effect on estimates of carbon footprints in specific countries or from specific food products will depend on how the global expansion is allocated. In some recent studies, including analyses of the aggregate footprint of food consumption in countries such as Denmark and the UK, the inclusion of land use change has made a very large difference to the results. The effect of including net carbon emissions from soils on existing agricultural land may affect results in either direction, since there may be net sequestration. It is too early to be certain of what the tendency in studies that take account of this factor will be. The estimation procedures used so far in studies that have attempted to take account of this factor vary widely, as do results of empirical research in this field.

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PART 2: Climate change and food in Norway – production and consumption perspectives

1 Background

This paper discusses the current state of knowledge regarding the carbon footprint of food consumption in Norway, with particular emphasis on results from some recent studies in which either of the two authors have been involved. So far, the Norwegian literature on food and GHG emissions from a life cycle perspective has been rather limited. However, many recent studies and policy documents have analysed aspects of the problem – in particular how the Norwegian agricultural sector contributes to GHG emissions and how its emissions might be reduced.

In a document entitled *"Klimastrategi – jordbruk"* (Climate Strategy – Agriculture) (2011) the Research Council of Norway mentions a number of relevant reports and ongoing activities. These include:

- Eltun et al. (2010), *Kunnskapsstatus – Bedre agronomi* (Knowledge status – Better agronomics) - a report by Bioforsk, The Norwegian University of Life Sciences and Hedmark University College.
- *Biogass – kunnskapsstatus og forskningsbehov* (2010) – a report by Bioforsk, SINTEF and the Norwegian University of Life Sciences.
- Report no. 39 to the Storting (2008-2009): *Klimautfordringene – landbruket en del av løsningen* (Climate change – Agriculture as part of the solution).
- *Klimakur* 2020 (Climate Cure 2020), Report no. 16 (Agriculture) – published by the Norwegian Climate and Pollution Agency.
- The Research Council's own revised plan for its Food Research Programme (2010-2015).
- International activities, including:
  - The EU Joint Programming Initiative on Agriculture, Food Security and Climate Change
  - The EU Framework Programme for Research (Theme 2)
  - The work of the Global Research Alliance on agricultural greenhouse gases
  - Nordic co-operation
  - Bilateral co-operation between Norway and other countries.

Other recent and important contributions in the field include a report from the Norwegian University of Life Sciences on GHG emissions from agriculture and forestry (Trømborg et al 2008) and a report published by Oikos – Organic Norway on organic agriculture and climate change (Swensen 2010).

Much of the published and ongoing work has a "producer perspective", meaning that the analyses relate GHG emissions to enterprises in the food production chain, and most often to agricultural enterprises. However, many actors and activities outside the agricultural sector affect the contribution that food makes to global warming. For some purposes, it is therefore useful to extend the analytical perspective to the entire food chain, including the industries that deliver inputs to agriculture or process its products, those who distribute or trade in food -and not least the consumers. If one also switches vantage points, to view the food chain from the consumers' end, further issues become apparent, including the importance of international trade. Norwegian food consumption leads to greater GHG emissions than Norwegian food production, since Norway is a net importer.

Changes in Norwegian production *and/or* in Norwegian consumption may lead to greater or smaller emissions elsewhere, depending on how they affect net imports. Since the impact of GHG emissions is independent of where they occur, it is important to consider this effect in assessing strategies to reduce emissions.

2 The food sector's overall impact on greenhouse gas emissions – from a production perspective

2.1 Food production and GHG emissions

Norway's total GHG emissions in 2009 were 51.2 million tons of CO_2 equivalents, according to official statistics[11]. Of this, 4.7 Mt was from agriculture, which was thus responsible for 9 per cent of emissions from Norwegian territory. The agricultural emissions (as CO_2 equivalents) were comprised of 48 per cent CH_4, 41 per cent N_2O and 10 per cent CO_2. Emissions from agriculture have fallen slightly since 1990, both in an absolute sense and as a share of total emissions. GHG emissions from agriculture as such are thus mainly in the shape of nitrous oxide – due to applications of organic and artificial fertilisers, cultivation of peat soils and reactions following nitrate leaching and ammonia emissions – or in the shape of methane, mainly from enteric fermentation and manure management. CO_2 emissions, mainly from fuel for tractors and other machinery, play a minor part – at least until losses of soil carbon, particularly from countries is almost the same as in Norway (9.2 % in 2007, according to Eurostat). .

2.2 From the agricultural sector to the whole food chain

In 2006, the GHG emissions from agriculture as reported by Statistics Norway were 4.8 Mt. Trømborg et al. (2008) published the following "corrected" estimate of emissions related to agriculture in Norway in the same year.

Table 2.1 Corrected estimate of emissions related to agriculture in Norway in 2006

Source	Emissions, 1000 t CO_2 equivalents			
	CO_2	CH_4	N_2O	Total
Official emissions figures	407	2,212	2,137	4,757
Fertiliser production	316		326	642
C losses from arable land	500			500
C losses from peat soils	1,900			1,900
Totals	3,123	2,212	2,464	7,799

Source: Trømborg et al. (2008).

Including production of artificial fertilisers and carbon losses from soils increases the total emissions from Norwegian agriculture by 3 Mt, according to this estimate. (It is worth notion that since 2006, emissions of N_2O from fertiliser production have been sig-

[11] http://www.ssb.no/english/subjects/01/04/10/klimagassn_en/tab-2011-05-25-04-en.html (emissions by industry), cf. http://www.ssb.no/english/subjects/01/04/10/klimagassn_en/tab-2011-05-25-02-en.html (emissions by source).

nificantly reduced through cleaning. On the other hand, the figure for CO_2emissions from fertiliser production in the table above only includes direct emissions from manufacturing, not emissions from transport, mining activities or production of the fossil fuels used in all of the processes.) There is some considerable uncertainty about the size of net carbon emissions from soils in Norway as in other countries, cf. the discussion in the previous paper by Hille. The issue is also discussed by Swensen (2010) and UNCTAD (2011).

The figures in table 2.1 still only include one contribution from an activity «upstream» of agriculture. There are many more inputs to be accounted for than artificial fertilisers, and production of capital goods such as buildings and machinery for agriculture also generates emissions. For the total GHG emissions caused by food products, activities downstream of agriculture – transport, processing and trade - may be even more important than those upstream (cf. table 2.1 in the paper by Hille). If emissions at the consumer stage – from shopping, storage and preparation of food and the waste phase – are included, this becomes all the more likely. Strategies that aim to reduce emissions from agriculture may have consequences (positive or negative) for downstream emissions, or they may presuppose changes in consumption patterns. This is also discussed for example by Coley et al. (2009), by Swensen (2010) and by Refsgaard et al. (2011). It is therefore a questionable merit to limit analyses of emissions related to agriculture strictly to those directly generated by the sector itself.

In comparing different production systems for foods of agricultural origin, it is important to include not only the complete range of emissions from agriculture itself, but also contributions from sources outside this sector. This is also important if we wish to make realistic estimates of the potential for reducing emissions.

2.3 GHG emissions in a consumption perspective

Quite apart from the fact that sectoral statistics on GHG emissions from agriculture exclude upstream and downstream emissions, national emissions statistics exclude emissions from production of goods the country imports, but include those from production of goods it exports. In other words, statistics are production-based and not consumption-based (Peters et al, PNAS 2011). In the particular case of Norway, this has a very lopsided effect, since the country imports large quantities of food and feedstuffs, but exports very small quantities, if we consider only agriculture and not fisheries and fish farming. Totally 35 – 50 % of the dietary energy consumed in Norway is imported (St.Meld., 2011). So in Norway's case, emissions related to production of foods from agriculture will be less than emissions related to consumption. (The reverse is true of seafood, of which Norway is a major net exporter.) The import/export factor is important in assessing potentials for emissions reductions. If changes in national agricultural production are likely to lead to greater or smaller imports of some products, this should also be accounted for.

International trade in food, as in other products can be affected by differences in taxes on emissions. Taxes on GHG emissions from the production of goods – other than energy goods - are rarely placed on the finished goods themselves, so that the consumer pays them directly. There are no CO_2 taxes – anywhere - on meat or oranges or bread. However, the price of such goods may be affected by taxes that enterprises along the production chain have to pay. This includes farmers as well as producers of inputs and enterprises engaged in transporting or trading in inputs of food products. At present, countries rarely tax the most important GHG emissions from agriculture as such, but some do tax emissions by other actors along the food chain[12]. If some countries tax emissions and others do not, or if some tax them heavily and others only lightly, then there will be an incentive to shift production to the countries where taxes are least. If taxes were placed on consumption instead, so that the consumer had to pay a tax proportional to all the emissions generated in producing and distributing the product, this incentive would disappear. As long as the «polluter pays principle» is not applied consistently and universally, a "consumer pays" principle could perhaps adjust the balance. However, such a regime presupposes a much better understanding of emissions all along the production chains for all kinds of products, including foodstuffs, than we have as yet.

Since Norway imports about half of its food (on a dietary energy basis), a considerable fraction of its food carbon footprint will be reported not in Norway's emissions statistics, but in those of other countries. The fraction is probably less than half, since Norwegian consumption of the most GHG-intensive foods (animal products) is largely covered by domestic production, albeit with the help of some imported feed. Norway is a net exporter of artificial fertilisers, but a net importer of many other inputs to agriculture. As mentioned in the previous paper, two attempts have recently been made at estimating the actual carbon footprint of Norwegian food *consumption*. Hille et al. (2009) arrived at a figure of 12.5 Mt CO_2eq in 2006, assuming that electricity used in Norway was generated within the country. Hertwich and Peters (2009) arrived at a figure which works out as 10 Mt CO_2eq in 2001. The former estimate includes distribution and trade in food while the latter does not. Neither figure includes carbon losses from soils, yet both are significantly higher than the estimate in table 2.1 of emissions related to (Norwegian) agriculture. In the next section we present an analysis done for some basic food items in Norway on their GHGs in which also the carbon foot print from the total amount of these products is calculated.

2.4 GHG emissions from production and consumption of some foods in Norway

Much of the research that has so far been conducted on potentials for reducing emissions related to agriculture has focused exclusively on direct emissions from primary production. The "Climate Strategy – Agriculture" published by the Research Council of Norway points out the need for studies that cover whole production chains, and mentions life cycle analysis as a relevant approach. Such approaches are certainly necessary if we want to compare different production systems for food products.

The Norwegian Institute for Agricultural Economics Research (NILF) has recently carried out research into GHG emissions from whole production chains for several food items. This research was funded by the Research Council of Norway, as part of a programme entitled "Societal and environmental assessments of organic agriculture". Figures for inputs to and outputs from the «Reference farms» defined by NILF were an important data source.

[12]In Norway, the present situation is that not only emissions of N2O and CH_4 from agriculture, but also CO_2 emissions from fossil fuels used to power tractors and other machinery, are exempt from tax. However, production of fertiliser and lime is subject to the EU Emissions Trading System, while transport and service enterprises – whether they contribute to the food chain or not – are subject to national taxes on fossil fuels.

2.4.1 Life cycle analyses – methods and results

In the study carried out by NILF in co-operation with MiSA – Environmental Systems Analysis (Refsgaard et al. 2011) life cycle analyses of four products from Norwegian agriculture were performed. These were milk, minced beef, brown bread and potatoes. GHG emissions from all phases in the products' life cycle from production of inputs (though not including the manufacturing of the capital goods) to agriculture until they reached the consumer's home were estimated (thus the analyses covered shopping trips, but not storage or processing at home. The analyses were performed for different production systems, including both conventional and organic systems. Emissions of N_2O and CH_4 were converted to CO_2 equivalents. The results shown below are weighted averages for products from various types of farm (NILF's "Reference farms" include both specialized and mixed farms, of varying sizes and in different regions of Norway). In some cases the results varied significantly according to what type of farm the products (at the farm gate stage) were assumed to come from. This is explained in more detail in Refsgaard et al. (2011).

Figure 2.1 shows the estimated emissions of CO_2-equivalents per kg of the four products and per megacalorie (Mcal) of dietary energy. Results are shown for conventional and organic products, except in the case of potatoes, where results are only available for the conventional product. The animal products have very much larger carbon footprints per kg as well as per Mcal than the plant products. Minced beef also has a very much larger carbon footprint per kg than milk, but the difference in emissions per Mcal is much smaller: 6 - 9 kg CO_2eq for the meat vs. approximately 4 kg CO_2eq for milk. This compares with a mere 0.14 – 0.24 kg CO_2eq for bread. In all cases where organic production was analysed, it was found to be more GHG efficient than conventional. The difference was greatest in the case of bread, somewhat less in the case of minced beef and least in the case of milk. On the other hand, organic production required more land in all cases (the difference was least in the case of meat, greater in the cases of milk and bread). Within each type of production system, the land use for animal products is higher than for plant products, with around 16-17 m2 per Mcal of meat compared to around 1 m2 per Mcal of wheat.

Another point to emerge from the study was that production of beef from dairy herds was much more GHG-efficient than production from sucker herds (11-15 kg CO_2eq/kg minced beef from dairy herds, vs. over 30 kg CO_2eq/kg minced beef from sucker herds).

The results may be compared with those from studies in other North-Western European countries that were discussed by in the previous paper by Hille. To do so, we must bear in mind that the results shown in the figure below include emissions from shopping trips, which are not included in most other studies. They made a negligible contribution to total emissions in the case of minced beef, but were responsible for about one-sixth in the case of milk and almost half in the cases of conventional potatoes and bread. If these emissions are subtracted, then the results for conventional potatoes and the average for beef fall well within the range of results from previous studies in nearby countries. However, the emissions from milk and beef from sucker herds were higher than those found in most previous studies, while those from bread were lower. Part of the reason for the latter is that emissions between the farm gate and the consumer, which according to other European studies can be significant in the case of bread, made a negligible contribution to the result in the Norwegian study. This again partly reflects realities (low CO_2 intensity of the Norwegian stationary energy mix)

Regarding comparisons of conventional and organic products, the previous paper shows that previous studies have consistently found GHG emissions from organic production of cereals to be less than from conventional, while most studies have found only a small difference one way or the other in the case of milk – with the data from model farms constructed by NILF. However the variation due to soil structure, feeding regime, etc. may impact on these results creating other differences between systems. In the case of meat results vary widely, some studies showing significantly lower emissions from conventional production and some the opposite. The Norwegian results do not contradict this pattern, though they show organic products performing better in all three cases.

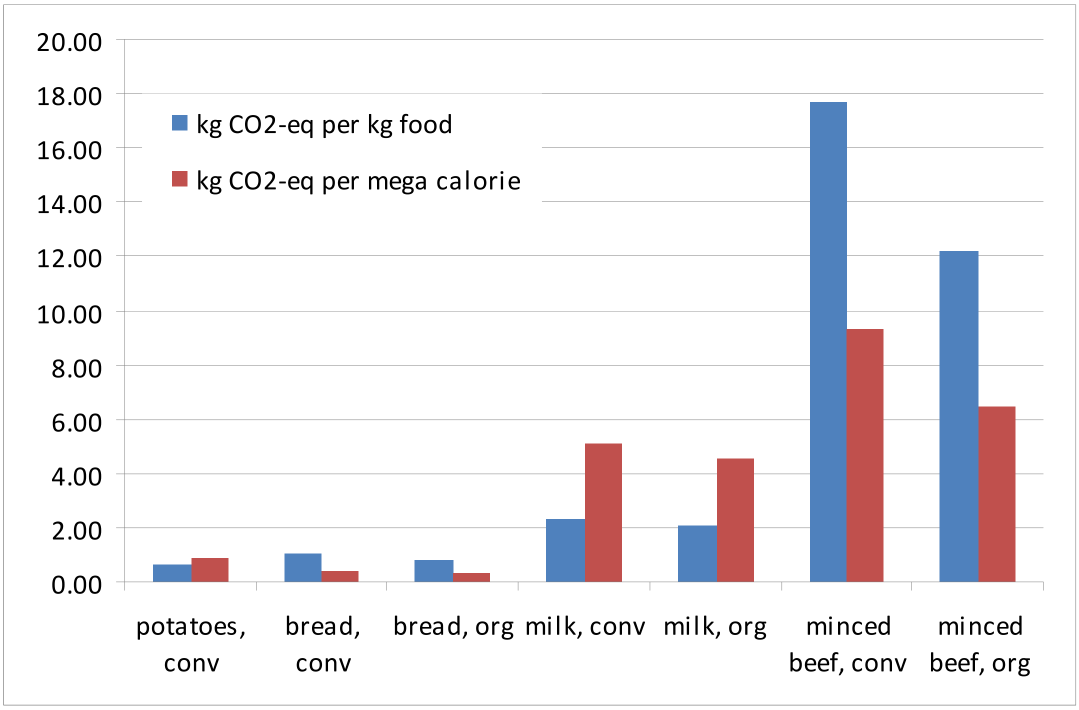

Figure 2.1 Results of Norwegian LCAs of food products. GHG emissions per kg of product and per Mcal of dietary energy

2.4.2 Composition of the total emissions for typical food items

Beef, bread, milk (including dairy products) and potatoes together make up about 50 % of the Norwegian diet by weight and a similar fraction in terms of dietary energy. They may be regarded as basic foodstuffs. The study by NILF and MiSA included an analysis of how much aggregate annual consumption of these four products would contribute to Norwegian GHG emissions, on the assumption that the imports of these products caused the same emissions per kg as domestic production. (Norway is largely self-sufficient in beef and dairy products, but imports large quantities of bread wheat and potatoes. Some of the feed for cattle is also imported.) The results of this analysis are shown in figure 2.2. The carbon footprint of annual consumption of beef, milk, bread and potatoes would be some 5.7 Mt CO_2eq if the products were produced conventionally and 4.8 Mt CO_2eq if they were produced organically. As in the figure above, this includes all emissions from production of fertiliser and other inputs until the products are brought home by consumers.

It is clear that the animal products contribute very much more to annual emissions than the plant products, not only per kg but also in aggregate, given current Norwegian consumption patterns. The largest contributions to emissions from milk and meat are direct emissions of CH_4 and N_2O from enteric fermentation and from the management and application of manure. These are responsible for over 50 % of emissions up to the farm gate in the case of conventional milk and some 70 % in the case of organic milk. That total emissions are higher for conventional products is mainly due to production and application of artificial fertilisers. In the case of conventional bread, most of the emissions up to the farm gate are due to production and application of artificial fertilisers. Note that emissions from the processing and retail stages were estimated to be very small in all cases. This is partly due to the Norwegian mix of energy sources (the main energy carrier used is electricity and electricity has a very low CO_2 intensity).

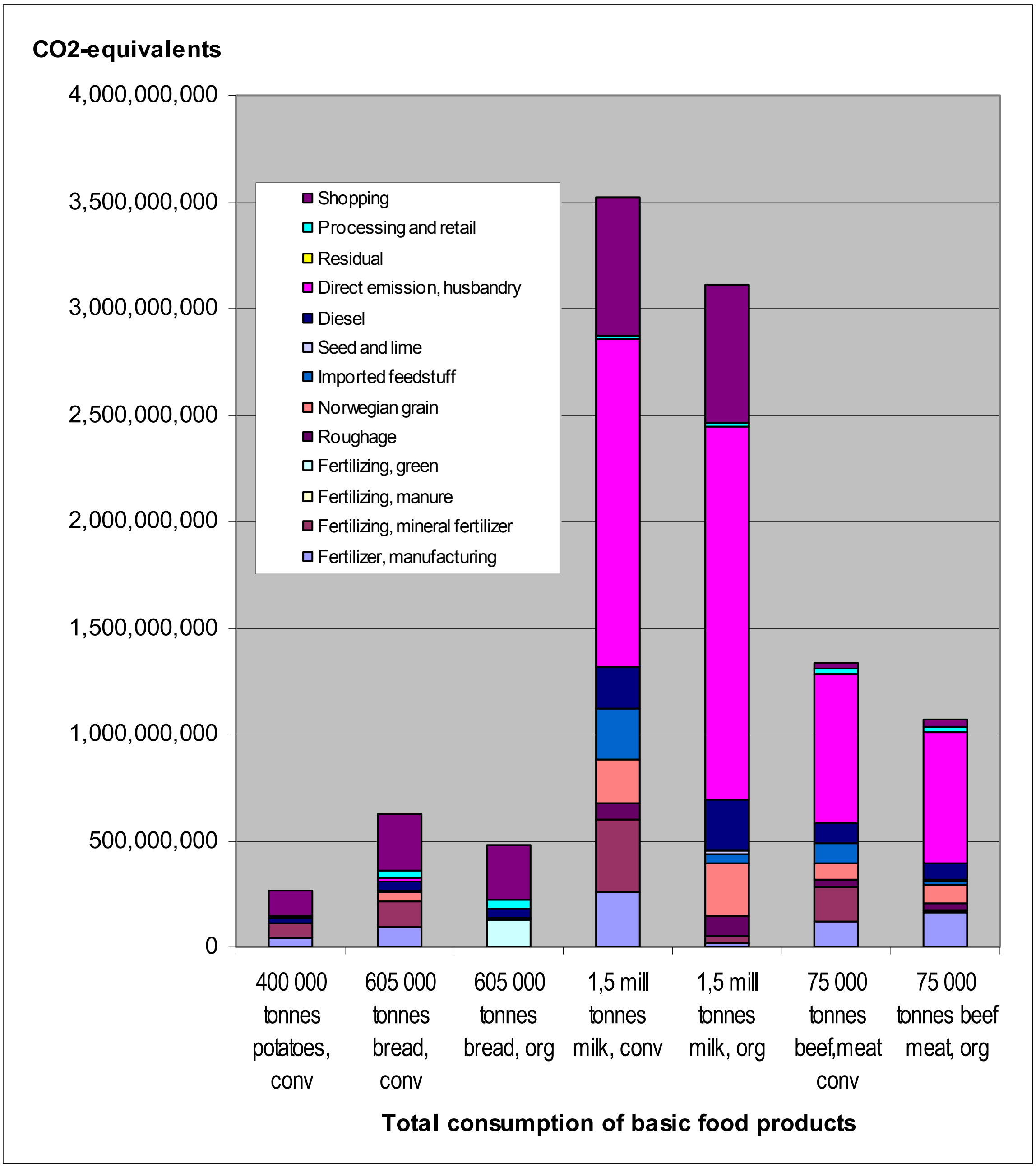

Figure 2.2 The contributions of different inputs and processes to GHG emissions caused by the total annual consumption of four basic food products in Norway

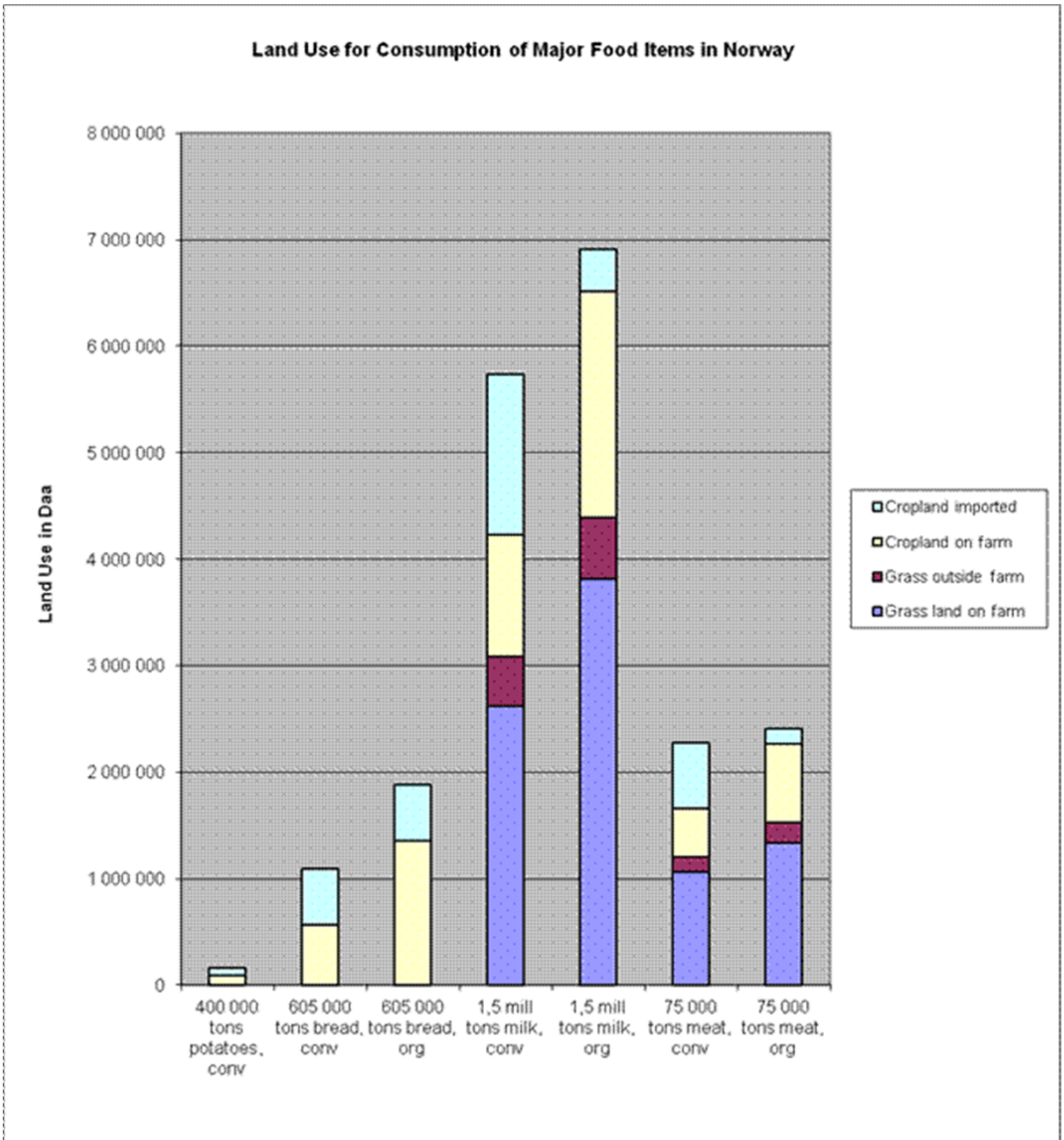

Figure 2.3 Land required in Norway and abroad to produce the aggregate Norwegian consumption of four basic food products

As mentioned above, organic products required more land than conventional, in all three cases where the two type of production systems were compared. However, the need for land in other countries to produce raw materials for feed concentrates was significantly less in the case of organic production, with a lower amount of imported protein in the feedstuffs. This is illustrated by figure 2.3 below. Milk production required 1.5 million decares (150.000 ha) of land abroad in the conventional case but only 0.4 million decares (40.000 ha) in the organic case. For beef, the figures were 0.6 and 0.14 million decares respectively.

2.4.3 Potential reductions in emissions

In the case of beef, the study by NILF and MiSA indicates that GHG emissions could be significantly reduced if organic production became the rule. The estimated emissions from producing 75.000 t of beef annually were 1.33 Mt CO_2eq in the conventional case, but only 0.92 Mt in the organic case. If all of the beef were produced from conventional sucker herds, the emissions would have increased to some 2.5 Mt CO_2eq. Being able to produce much of the beef from dairy herds is an advantage, although the amount of beef they can deliver is not unlimited in relation to demand for milk.

A transition to 100 % organic production of bread and milk as well as beef in Norway would, according to the estimates by NILF and MiSA, reduce emissions by some 0.9 Mt CO_2eq. However, such a transition – without any change in consumption - would also require an extra 2.1 million decares (210.000 ha) of agricultural land, including both grassland and arable land. This would be composed of an increase on 2.1 million decare in domestic cropland, an increase on 1.6 mill. decares in domestic grassland and a decrease in 'imported land' of 1.6 mill. decares. According to Svendgård-Stokk (2011), about 23 million decares of land are available for agriculture in Norway. Today, about 10.8 million decares are cultivated, of which 3.5 million decares are is crop land (The Budget Committee for Agriculture (2010). This leaves 12.2 million decares of available land. However, because of the change in the composition of cropland, the change in carbonsequestration in soil has to be considered. Further, a large amount of the potential land is today either moor or forested area and primarily in the mid- and northern parts of Norway. Therefore, further analyses and calculations of whether the total emissions would decrease or increase by a total change to organic agriculture have to carried out.

Changes in consumption patterns could allow GHG emissions from the food chain to be reduced, without any increase in the demand for farmland. Specifically, two changes could make a major difference: lowering the amount of food waste and increasing the share of plant vs. animal foods in our diet. At present, some 30 % of food ends up as waste in Norway, of which at least 50 % is edible, see Refsgaard et al. (2011) for calculations on this issue. NILF and MiSA estimated that emissions could be reduced by some 0.9 Mt CO_2eq if this waste were eliminated in the case of the four products analysed (assuming conventional production). As illustrated by figure 2.1 and figure 2.3, and shown through the discussion of other European studies in the previous paper, plant foods *not only* tend to generate much smaller GHG emissions per unit of dietary energy than animal foods, but also to require much less land.

2.4.4 Discussion

When considering possible courses of action to reduce GHG emissions, it is necessary also to consider what other consequences they are likely to have. A fair comparison of alternative agricultural systems must include an assessment of how much land they require to produce a given amount of food. Land can substitute for fossil energy and organic agriculture – at least under European conditions – tends to make up for its avoidance of artificial fertilisers and pesticides by requiring more land per unit product than conventional agriculture.

The analyses by NILF and MiSA showed that consumers, as well as the agricultural sector itself, can contribute to reducing the carbon footprint of food consumption. Many recent analyses and proposals for climate strategies have focused on measures within the agricultural sector, including improved management of manure, optimization of fertiliser applications, better drainage, production of biogas and mixing biochar into soils. However, the cost-benefit equation for some measures, such as biogas production, is uncertain, dependent as it is on linkages to sectors outside of agriculture. In such cases, multisectoral analyses are needed. Also, downstream as well as upstream processes within the food chain – including processing and distribution – need to be better explored with a view to identifying GHG reduction potentials. An alternative (or complementary) approach is to start from the consumer's end of the food chain, and ask how changes in diet, represented by alternative «baskets» of food, might contribute to reducing emissions (and perhaps land requirements).

2.5 Other Norwegian sources

2.5.1 GHG emissions by stages in the food life cycle

Hille et al. (2008) estimated the amounts of energy used in Norway and abroad to produce, distribute and market the food consumed in Norway in 2006. These estimates are broken down by stages in the chain from production of capital goods and inputs to wholesale and retail trade. Hille et al. did not explicitly split GHG emissions by stages in the production chain. However, the source includes emission factors for energy carriers used in Norway as well as in other countries, and also separate estimates of process emissions from fertiliser production and from agriculture in Norway and abroad. It is therefore also possible to calculate the implicit breakdown of GHG emissions by stages, with only minor items omitted. Figure 2.4 shows the results. The GHG emissions distributed in this figure sum up to 12.3 Mt CO_2eq, whereas the total estimated by Hille et al. was 12.5 Mt.

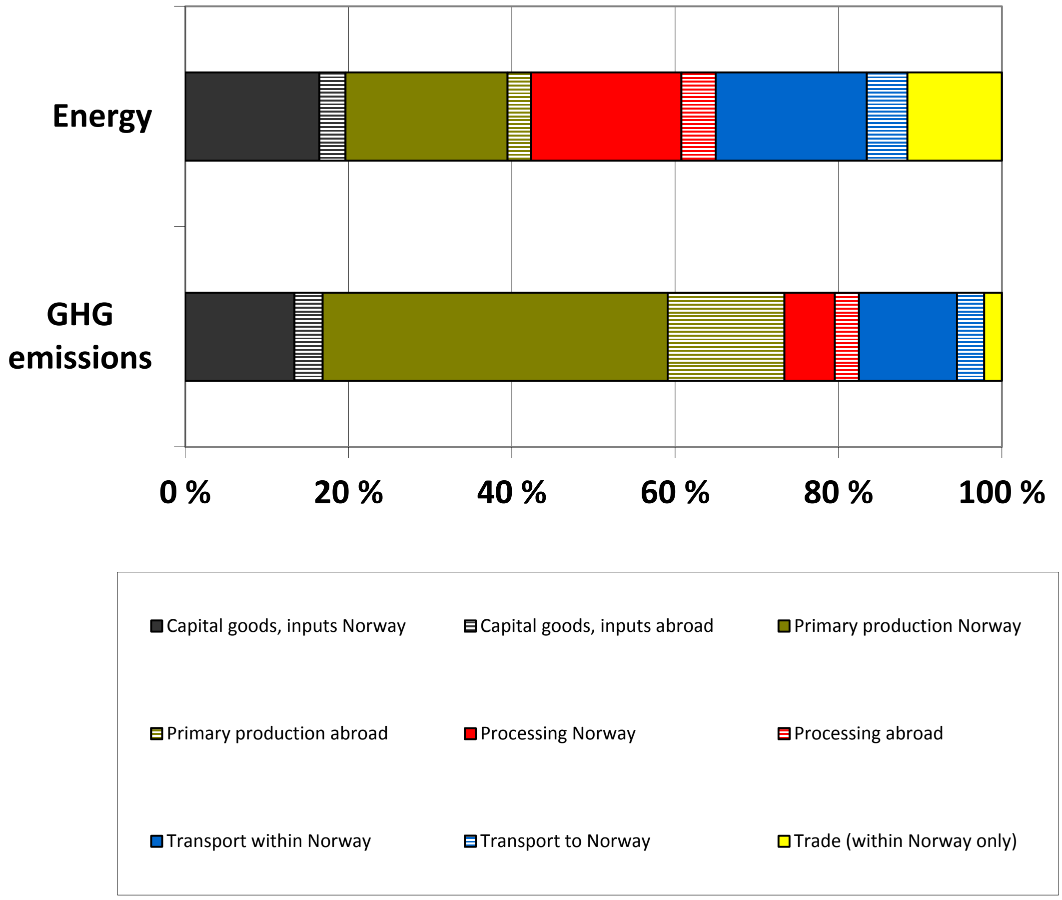

Figure 2.4 Breakdown of energy use and GHG emissions in the production chain for food consumed in Norway in 2006, as estimated by Hille et. al (2008).

Notes:
1) "Capital goods, inputs" covers deliveries to primary production (agriculture and fisheries) only. Production of capital goods and inputs used in downstream processes is included in the totals for these; e.g. emissions from production of packaging materials are included in emissions

from food processing, and emissions from production of vehicles used for transporting food are included in emissions from transport.
2) Transport refers to transport downstream of the farm gate or of the harbour where fish were landed Transport of capital goods and inputs to agriculture and fisheries is included in "capital goods, inputs".
3) In the case of capital goods and inputs, the share marked "Norway" covers all deliveries to Norwegian primary production – whether the inputs were produced in Norway or in other countries – and conversely, the share marked "abroad" only covers deliveries to primary production abroad. In all other cases, "Norway" refers to processes that took place on Norwegian territory and "abroad" to processes that took place elsewhere.
4) Traded feedstuffs are not regarded as an input to agriculture but as cross-deliveries within the agricultural sector. Therefore, emissions from production of imported feed are included in "primary production abroad" and not in "capital goods, inputs".

It needs to be emphasised that the estimates by Hille et al. (2008) of emissions from imported foods, including emissions from primary production as well as from inputs and processing, are based on simplified estimation procedures and can at best give a rough indication of likely orders of magnitude. For instance, N_2O emissions from land used to produce for export to Norway were assumed to be the same per hectare as from land in Norway (although applications of artificial fertiliser on this land were assumed to be smaller per ha than the Norwegian rate). "Imports" of CH_4 were estimated only in relation to imports of rice, since Norway was a minor net exporter of products from ruminants in 2006, and largely self-sufficient in other animal products. For most products of which Norway imports significant quantities in processed form, energy use in processing was estimated on the basis of a Danish study. For most of these foods, the processing takes place in Europe. Estimates of energy use for transporting products to Norway were made on the basis of the actual regions of origin of food imports and estimated average transport distances by ship and by road for products from each region. (The modes of transport on entry to Norway were available from statistics.)

As mentioned previously, Hertwich and Peters's (2009) estimate of the carbon footprint from food consumption *excluding distribution and trade* was about 10 Mt CO_2eq, which agrees very closely with the results from Hille et al. (2008). Out of the 12.3 Mt CO_2eq represented by the GHG emissions column in figure 2.4, 2.1 Mt were due to distribution or trade.

Provided that the estimates shown in figure 2.4 were not very wrong, they indicate, firstly, that processes downstream of primary production were responsible for a much larger share of energy use in the food chain than of GHG emissions. This is especially true under Norwegian conditions, since much of the energy used in downstream processes in this country is non-fossil. However, it is likely to be true to some degree in all countries, because primary production generates major GHG emissions that are unrelated to energy use, whereas downstream processes usually do not. These points are discussed in more detail in the previous paper.

A second point that emerges from figure 2.4 is that while most of the carbon footprint of Norwegian food consumption is due to Norwegian production, the share due to foreign production, including shipment of imports to Norway, is not insignificant. According to figure 2.4 this share is some 25 % *plus* the share due to foreign production of capital goods and inputs to Norwegian primary production (such as tractors). The uncertainty in the figures is such that it is fairer to say that processes outside Norway quite likely contribute somewhere between one-fifth and one-third to the food carbon footprint. The question deserves to be more closely analysed, and consequences for global emissions deserve to be incorporated in future analyses of climate strategies for the Norwegian food sector.

A third point is that while processes downstream of the farm gate contribute less to GHG emissions than to energy use, their contribution to the latter was still significant according to this study (somewhat over one-quarter of the total according to figure 2.4) . On this point, the estimates by Hille et al. (2008) differ somewhat from those by Refsgaard et al. (figure 2.2), who found that the downstream processes (apart from shopping trips, which were not included by Hille et al.) made very small contributions to the carbon footprints of the four products they analysed. The apparent differences mainly concern processing and transport of food. It is clear that the retail stage causes small direct GHG emissions under Norwegian conditions, since the energy used there is mainly electricity from renewable sources. This is also the main source of energy in Norwegian food processing, but the share of fossil energy in processing and especially in production of packaging – much of which is imported – was estimated by Hille et al. to be considerably higher than in retailing. Still, the most important apparent discrepancy concerns emissions from transport of food, in whose case both studies had to rely to a large extent on assumptions. We lack thoroughgoing empirical analyses of fuel consumption and emissions from food distribution in Norway. This is clearly an issue which deserves more attention.

Whatever the current level of emissions from transport of food, there are strong grounds to suspect that they are increasing. The volume of transport of food and feed by lorry within Norway increased by a factor of 2.3 between 1993 and 2007, from 1.8 to 4.1 billion ton kilometres (Hille 2010). Road transport is the dominant mode for most foodstuffs, although ship and rail transport are much more energy efficient. *Imports* of some of the bulkiest foods – vegetables and fruit – have also increased dramatically over the past 20 years Hille 2011), and over half of these also arrive by road vehicle. This is likely to have pushed emissions from transport of food to Norway upwards, along with emissions within Norway.

2.5.2 Which foods have the greatest environmental impacts?

The study by NILF and MiSA that was dicussed above showed that animal products – and (especially) beef – not only had much higher carbon footprints per unit of weight and dietary energy, but also demanded much more land than bread or potatoes.

A Norwegian analysis which covers GHG emissions from a wider range of animal and plant products up to the farm gate stage is presented in the paper by Pettersen and Solli. For the products covered by that study, the results are broadly in line with those inferred by Hille in the previous paper from other European studies. If one compares GHG emissions per unit of dietary energy, there is general agreement that ruminant meat is close to the high end of the range, perhaps alongside greenhouse vegetables. Other animal products, such as pork, chicken meat, eggs and dairy products are in the next echelon, perhaps alongside demersal fish and fruits and vegetables that have been highly processed and/or travelled a long way. Locally produced fresh fruit and vegetables perform better, as do fatty pelagic fish. Energy-dense plant products such as cereals, sugar and plant oils are at or close to the bottom of the list, at least as long as their production does not presuppose land use change.

Results regarding land use tend to show a similar pattern to those for GHG emissions, apart from the fact that primary production of seafood requires no land. Estimates of land requirements per unit of dietary energy for major categories of food consumed in Norway, based on a study by Hille and Germiso (2011) are presented in the previous paper (table 3.10). They indicate that ruminant meat is most land-intensive, followed by chicken meat, eggs, pork and dairy products. Fruits and vegetables, excluding potatoes, are almost as land-intensive as milk by this measure. Potatoes, cereals, plant oils and sugar – in that order – require the least land per unit of dietary energy.

There is a lack of comparative Norwegian studies of other environmental impacts from different foods. Studies from other countries suggest that the difference between animal and (most) plant foods may be less regarding energy use and contributions to eco-toxicity, than it is in the case of GHG emissions. On the other hand, the difference may be greater regarding contributions to acidification, because of the ammonia emissions associated with animal husbandry. In the case of eutrophication the impacts are much more related to the local conditions. We cannot draw any general conclusions about these issues in the Norwegian context based on general tools like LCA..

2.5.3 Diet and emissions

If we are able to compare the contributions various foods make to GHG emissions, then we can also estimate the on emissions of possible changes in diet. This was done by Nymoen and Hille (2010), in a study that focused on diets for elderly people in care institutions. Designing a nutritionally adequate, low-emission diet for people in this group was assumed to be more challenging than proposing such a diet for the general population, because special requirements and limitations apply. Because elderly people with a low level of physical activity require fewer calories, yet largely the same amounts of essential nutrients as other adults, the food must be "nutrient dense". It should also be easy to chew, and not too different from what people have become accustomed to during their long lives. Yet within these restrictions, Nymoen and Hille were able to propose a diet which according to their calculations would result in 35 % less GHG emissions per calorie than the current average Norwegian diet. The proposed diet differed from the current average mainly through including:

- less meat, but more eggs and fish
- less greenhouse vegetables and preserved vegetables, but more root vegetables and other fresh vegetables in season
- more Norwegian fruit and berries, and less from distant sources
- no rice, but more barley and oats
- very few highly processed foods - it was assumed that kitchens could prepare meals from fresh ingredients.

The share of dairy products in the diet was largely the same as the current Norwegian average. Recognising that there was a high degree of uncertainty in their estimates of GHG emissions from various food products, Nymoen and Hille performed a sensitivity analysis. They found that if all of the foods of which they proposed an increased intake generated emissions at the upper end of their likely ranges and vice-versa, then this diet would still lead to 23 % less emissions per calorie than the average Norwegian diet.

We are not aware of other comparable Norwegian studies so far. A Danish and a British study which have recently addressed the issue of diet and GHG emissions are discussed in the previous paper by Hille. The former proposed fairly moderate changes in diet, which led to a minor reduction in emissions, while the latter estimated that truly radical changes could reduce the (UK) food carbon footprint by more than half.

2.6 Discussion – Possible strategies for reducing the food carbon footprint

If we aim to realise as much as possible of the potential for reducing the carbon footprint of food consumption, an understanding of how different foods contribute to it is clearly useful. So is an understanding of how different stages in the food life cycle contribute, and of the interactions between them. At least in the Norwegian context, there are still some significant knowledge gaps on both of these points. Nevertheless, it is already possible to point out a number of possible strategies for reducing GHG emissions, including some strategic choices between alternative approaches.

In a White Paper on agriculture, food and climate policy, (Landbruks- og matdepartementet 2009) the Norwegian Ministry of Agriculture presents a wide range of possible measures to reduce GHG emissions from the food chain. Most, but by no means all of these measures concern the agricultural sector as such. They include:

- Measures to enhance carbon sequestration or prevent carbon release from soils, including a ban on new cultivation of peat soils, reducing autumn tillage, more use of cover crops, increasing the area of meadows and pasture at the expense of cereal crops and mixing inert carbon into soils.
- Better drainage to reduce N_2O emissions.
- Generating biogas from manure to simultaneously reduce CH_4 emissions and provide renewable energy.
- Reducing applications of nitrogen fertiliser.
- Increasing the share of farmland under organic management.
- Increasing beef production from dairy herds in preference to sucker herds.
- Increasing the share of concentrates in cattle feed to reduce CH_4 emissions.

Other measures that are discussed concern downstream links in the food chain – such as cleaner technology and in particular improved logistics in distribution and trade.

The White Paper also points out the potential of measures that directly address consumers, including:

- Promoting sustainable diets through better information about the carbon footprints of different foods.
- Making more seasonal and locally produced foods available in supermarkets.
- Measures to reduce food waste.

There are some apparent conflicts among these various goals. For instance, an increase in demand for concentrates at the expense of coarse fodder would be hard to square with increasing the area of meadows and pasture, or with more organic agriculture. A change in diets that involved less consumption of beef and/or dairy products and more of plant foods would likewise be hard to square with an increase in the area devoted to grass, unless the share of concentrates in feed were to decrease rather than increase. As the study by Refsgaard et al. (2011) highlights, increasing the share of organic production could reduce GHG emissions per unit product from existing farmland, but would require more land if food consumption were unchanged. Some of the land available for cultivation is moorland and would be excluded by the first point above; much of the rest is at present forested, which also means that cultivation would initially lead to a net release of carbon.

However, there are also some potential synergies among the points above. A change in diets combined with a reduction in food waste could make a large-scale transition to organic agriculture possible without requiring much more land. This illustrates the importance of combining producer-oriented and consumer-oriented approaches in food climate policy. So far, the latter have partly been lacking and partly had rather limited effects. Meat consumption has increased strongly over the past 20 years, and while the Government has a target of increasing the organic share of food consumption to 15 per cent by 2020, it is currently stalled at just 1 per cent.

In its White Paper, the Ministry of Agriculture and Food also pointed out the need for more research into the life cycle of foods in Norway, to provide a better knowledge base for policymaking as well as information to consumers. The studies presented in this and the following paper will hopefully contribute to that end, but there is still a need for more work to cover a wider range of foods and improve our understanding of some important processes and linkages.

One issue not addressed in the study by Refsgaard et al. was that of carbon release or sequestration by soils. In the light of a point made above – that there may be trade-offs between, say, increasing the area under grass or avoiding an expansion of agricultural area, and other measures that would reduce emissions – this is an important issue. Not enough is yet known about rates of carbon release or sequestration by soils under grass and annual crops and under conventional and organic management in Norway. There is a need for more original research in this field as well as for studies that incorporate this factor into life cycle analyses of individual foods.

Another field which clearly needs to be further explored is that of emissions downstream of the farm gate. Refsgaard et al. (2011) found that emissions between the farm gate and the consumer were rather small for the four products they studied, which would suggest that the scope for reducing emissions through improved logistics or technology in downstream processes is also limited in Norway. On the other hand, Hille et al. (2008) estimated these emissions to be quite significant on average for all food products. The issue can only be settled through more thorough analyses, perhaps of distribution systems in particular. Although food processing and retailing may make minor contributions to the carbon footprint of food in Norway due to the mixes of energy used, their contribution to energy use is larger. Opportunities for saving electricity in these sectors are also relevant to climate policy if one considers that saved electricity could substitute for fossil fuels in other areas.

Finally, there is a need to translate findings about the carbon footprint of foods into relevant policies that address not only producers but also consumers. This may be the greatest challenge. While nutritional policies motivated by health concerns have met previously achieved noticeable success in Norway, corresponding policies driven by environmental concerns are still largely uncharted territory and likely to demand new approaches.

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PART 3: A model for calculation of total environmental impacts from agriculture in Norway – Linking consumption and production through input-output models – A possibility study

1 Background

This paper documents the work done by MiSA in the NILF project "Environmental and climate analysis for the Norwegian agriculture and food sector and assessment of actions". The project was supported by SNF and NILF in addition to 25 % self-funding by MiSA.

Briefly, the overall aim is to review existing studies and methods for assessing the environmental impacts of the agricultural sector and consumption of products from agriculture and food products. The literature review is presented in Part 1 of this report along with overall recommendations for further work.

Further, the project aims at suggesting and outlining possible ways to develop a model that will improve our knowledge of environmental impacts from Norwegian agriculture specifically. As part of this work, a simplified model of Norwegian agriculture has been developed, partly to test suggested practices, and partly to assess the usefulness of existing data sources for use in a model. The simplified model also gives examples of the kinds of information that could be drawn from such model.

A few characteristics of the desired model are clear: It must be able to present emissions estimates for food products based on life cycle assessments, i.e. it must cover the entire value chain from resource extraction via production and processing, to sale and consumption. Second; the connection between consumption and production in the model should be explicitly modeled. Finally, the model should preferably be able to incorporate economic analysis, or at least connect to econometric models of the agricultural sectors, as has been demonstrated by others (Wolf et al. 2011).

2 Metods

The following sections briefly describe the methodology behind life cycle assessment and input-output analysis.

2.1 LCA

Life cycle assessment (LCA) is the assessment of environmental impacts through the life cycle of product systems. A cornerstone of the life cycle approach is the understanding that environmental impacts are not restricted to localities or single processes, but rather are consequences of the life cycle design of products and services. The product life cycle covers all processes from extraction of raw materials, via production, use, and final treatment or reuse (Wenzel et al. 1997; Guinée 2001; Baumann and Tillman 2004; ISO 2006). The combination of a quantitative approach and a holistic perspective leads to trade-offs being clearly stated in LCA. It is a systems tool well-suited for environmental decision-making.

Referred to by many names through its development (Baumann and Tillman 2004), LCA has in the last four decades evolved from the assessment of cumulative resource requirements into a scientific field that includes emission inventory methods (Heijungs and Suh 2002) and environmental cause-consequence modeling (Udo de Haes et al. 2002). Many of the first applications, including the first Norwegian use of the life cycle concept (Nunn 1980), were related to beverage packaging, although early reviews show that a large range of products were assessed through life cycle approaches (Nord 1992).

The problem of including all significant processes in life cycle inventories is well known in LCA (Norris 2002). Hybrid approaches have been proposed as a method to identify the largest contributing paths and to ensure that all processes are included within the system boundaries (Suh 2004; Suh et al. 2004). Hybrid approaches link process information collected in physical life cycle inventories with monetary flows in economic models. The combination of LCA and input-output models has proved valuable as a complementary tool to traditional inventory methods in LCA (Heijungs and Suh 2002; Strømman 2005; Strømman et al. 2006).

Standardization of LCA methodology has been achieved step by step. The SETAC working groups (e.g., Consoli et al. 1993; Barnthouse et al. 1997; Udo de Haes et al. 2002) and other institutions have been vital in this process (e.g., Nord 1992, 1995). The development of international standards has been an important driver for defining the methods of LCA. The first set of standards was published by the International Organization for Standardization in 1997 (ISO 1997), with a revised version complete in 2006 (ISO 2006). For a more thorough description of the historical development of LCA, see Ayres (1995) and Baumann and Tillman (2004).

General framework

The standardized framework for LCA sets out four consecutive stages, as illustrated in figure 2.1 (ISO 2006). The stages are described in some detail here, but the reader is referred to guidelines and textbooks for a thorough introduction (e.g., Wenzel et al. 1997; Hauschild and Wenzel 1998; Guinée 2001; Heijungs and Suh 2002; Baumann and Tillman 2004; ISO 2006).

Goal and scope

The first stage of LCA consists of defining the aim and boundaries of the assessment, and the choice of methods for inventory and impact assessment. The goal and scope stage includes defining the functional unit (FU). The functional unit is a quantitative measure of the functional requirement(s) that the product or service is designed to fulfill. It is the basis for comparison in LCA, used to evaluate the relative performance of alternative product systems.

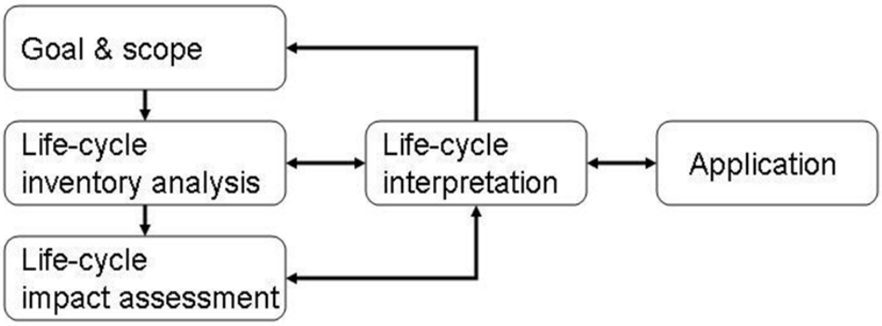

Figure 2.1 Outline of the stages and iterative approach of life cycle assessment (Adapted from ISO (2006)

Examples of FUs are 15 years of passenger transport for transportation systems, 100 m2·years for paints and other surface protectors, and *1 GJ at consumer* for energy supply and distribution systems.

Life-cycle assessment may be applied for various purposes, such as product benchmarking, product declaration, process development or policy support. Study designs set important limitations to the applicability of the study to provide answers. An important issue in this respect is the functional unit. Other issues include the level of inventory completeness, temporal and spatial considerations, and impact and inventory assessment approaches.

Limitations in scope may be caused by resource constraints. Spatial and temporal limitations may be applied to suit policy perspectives. Similarly, a study may be undertaken to investigate a few issues of concern, such as energy efficiency or CO_2 equivalents, or it may aim at a broad impact assessment. While limitation of the scope is a necessary step towards completing any study, it is vital that the principle of reproducibility is maintained; i.e., that any limitations applied do not exclude information that may alter the conclusions.

Life-cycle inventory analysis (LCI)

The second stage consists of establishing an inventory that describes the environmental interventions that arise from the product system. Environmental interventions are inputs of resources from the environment to the product system (i.e., energy and material resources), and outputs to the environment that the product system produces (usually emissions. The inventory is balanced to the functional unit.

Life-cycle impact assessment (LCIA)

Once the inventory of environmental interventions is established, the interventions are translated into environmental impact indicators in the third stage of LCA.

The ultimate purpose of LCA is to provide indication on environmental impact potential. Quantitative scores are achieved by applying characterization factors that describe the relative potential of each intervention to adversely affect "safeguard" objects through defined impact mechanisms. An example is CO_2 equivalents, which are used to aggregate the global warming potential of various atmospheric emissions. Each substance is characterized by its global warming potential relative to that of CO2.

The life cycle impact assessment stage is divided into three consecutive steps. First, environmental interventions are separated according to their cause-and-effect chains, termed impact chains or impact categories in LCA. Interventions may relate to inputs, e.g. energy and materials extracted from the environment, or to outputs, e.g. emissions to the environment. Second, impact scores are aggregated for each impact category by multiplying inventory mass flows with their respective characterization factors and summarizing for each of the impact chains. The last step of life cycle impact assessment is the weighting of impact scores relative to each other. Weighting requires judgments about the relative importance of different environmental issues, such as the importance of acidifying atmospheric emissions relative to consumption of material resources. An inherently subjective process, and a voluntary step in life-cycle impact assessment, weighting is not often applied in the scientific literature.

Weighting methods and the selection of impact categories to be considered in an LCA depend on the stakeholders in the study. Identification of stakeholder attributes, and the matching of these with the results produced by the study, is vital to ensure the relevance of any LCA.

Life-cycle interpretation

The final stage of LCA is the interpretation of results. Vital to the interpretation stage is the consideration of uncertainty. Other aspects include the ability of the selected impact assessment methods to fulfill the stated purpose of the study in a valid manner, and the potential bias introduced by inventory sources and approach. The re-visitation of methodological choices validates the outcome of LCA and increases the relevance of LCA for decision support.

2.2 Environmentally extended input-output analysis and hybrid LCA

Input-output analysis (IOA) was initially developed by Leontief (1936) as a method to study the interrelations between the sectors in an economy. In the beginning of the seventies he formulated a framework with which to extend the analysis with environmental information (Leontief 1970).

The basic idea is to utilize the information contained in national economic statistics, in combination with data on emissions from the various sectors in the economy, to calculate all the (direct and indirect) emissions occurring from an arbitrary final demand placed upon the system.

The economic consequences of spending 1 NOK on, say, gasoline, may be calculated and traced through all the interconnected sectors of the economy in an infinite (but converging) series of demands between the sectors. Once the economic outputs required to support the production of this 1 NOK purchase of gasoline have been calculated, the resulting vector of economic activity in each sector may then be multiplied with emissions intensities for each sector to give the total (life cycle) amount of emissions occurring in the production of 1 NOK worth of gasoline.

Once the emissions have been calculated, the procedure that follows may be the same as in LCA.

The approach has been developed significantly since Leontief, both for stand-alone use (Suh and Huppes 2002), multiregional analyses (Peters and Hertwich 2006a, 2008) and structural studies (Peters and Hertwich 2006b; Guan et al. 2008; Guan et al. 2009).

The structure and compilation of input-output tables for an economy is described in detail in United Nations (1999).

Hybrid life cycle assessment

While process based LCA[13] is relatively specific in the type of data used, it has been criticized for leaving out significant portions of the emissions that occur in the system (Lenzen 2001; Norris 2002; Strømman et al. 2006). This is referred to as cut-off and applies particularly to far upstream processes and service based activities.

On the other hand, input-output analysis is good for including emissions from all types of activities without any cut-offs, since it is based on an aggregated model of all existing sectors of the economy. However, it lacks the detail provided by LCA, so that it may be good for estimating e.g. total emissions from household food consumption, but is not able to distinguish between bread and bananas.

Several authors describe the use of LCA and IOA in a hybrid approach, which tries to utilize the benefits of both approaches, in order to achieve both the completeness associated with input-output analysis, and the specificity offered by process based LCA.. Various variants of these approaches are described by several authors (Treloar 1997; Nakamura and Kondo 2002; Suh et al. 2004; Suh and Huppes 2005; Strømman and Solli 2008) and applied to different case studies (Marheineke et al. 1999; Treloar et al. 2000; Lenzen 2002; Solli et al. 2006; Strømman et al. 2006; Michelsen et al. 2008; Larsen and Hertwich 2009).

Computational structure

The computational structure of life cycle assessment, input-output analysis and hybrid LCA is more or less identical. The idea is to calculate emissions occurring as interconnected processes are instigated by a final demand. Several authors give detailed descriptions of the computational structure of LCA (Heijungs and Suh 2002; Peters 2007). We will give a short description of the computational framework in the following. Beware that notation may differ from other sources as there is no generally agreed nomenclature that applies to all methods.

We start by defining our system of production processes, economic sectors, or both (in hybrid analyses) as a matrix Z, containing the flows of energy, materials, money etc. between the different entities (from now on referred to as "nodes").

$$\boldsymbol{Z} = \begin{pmatrix} \boldsymbol{z_{11}} & \cdots & \boldsymbol{z_{1j}} \\ \vdots & \ddots & \vdots \\ \boldsymbol{z_{i1}} & \cdots & \boldsymbol{z_{ij}} \end{pmatrix}, \quad \boldsymbol{x} = \begin{pmatrix} \boldsymbol{x_1} \\ \vdots \\ \boldsymbol{x_i} \end{pmatrix}$$

Each element zij of the matrix denotes the flow of the product from node i into the production of output from node i. In addition we have information on the total output from the system, x. If the total output from each node is described by the vector x, a normalized system may be constructed by dividing each column in Z by the corresponding total outputs. The result is a matrix A containing the "cookbook recipe" for producing one unit of output from each node in the system.

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[13]The standard/"old fashioned" LCA is often described as "process based LCA" to distinguish it from approaches using input-output analysis or hybrid assessments.

$$A = Z\hat{x}^{-1}$$

Say we want to calculate the total output from each node due to some final demand by an end consumer, for instance a household. The final demand from the various nodes can then be described by the vector y.

$$y = \begin{pmatrix} y_1 \\ \vdots \\ y_i \end{pmatrix}$$

Setting up a balance, we know that the total output of the nodes, less the amounts consumed by the nodes themselves, should equal the final demand y.

$$\overbrace{x}^{total\ output} - \overbrace{Ax}^{consumed\ by\ nodes} = \overbrace{y}^{final\ demand}$$

The total output x from each node needed to fulfill the demand of the household, in addition to all the intermediate demand from other nodes, can then be calculated by

$$x = \overbrace{(I - A)^{-1}}^{Leontief\ inverse}\ y$$

The Leontief inverse, L, is a matrix describing multipliers for all nodes in the system, so that a column j in L gives the total direct *and indirect* outputs in all other nodes in order to deliver a unit of final demand from j.

Similarly, emissions can be treated in the same way, where the matrix S is total emissions and where an element skj contains the emissions of substance k from node j.

$$S = \begin{pmatrix} s_{11} & \cdots & s_{1j} \\ \vdots & \ddots & \vdots \\ s_{k1} & \cdots & s_{kj} \end{pmatrix} \rightarrow F = S\hat{x}^{-1}$$

Normalization by dividing by total node output gives a matrix F of emission intensities per unit output from each node.

The total emissions e occurring due to an arbitrary final demand from the nodes can now be calculated as

$$e = Fx = F(I - A)^{-1}y$$

Introducing characterization factors according to the description in section on *Life-cycle impact assessment (LCIA)* on page 106 gives the opportunity to translate the emissions data into more readily comprehensible environmental impact potentials d. The characterization factors are contained in the matrix C, where an element clk describes the contribution of emission type k to impact category l. The calculation of d then becomes

$$d = Ce = CFx = CF(I - A)^{-1}y$$

This is as far as we shall go in this presentation of computations in life cycle assessment, but the well-defined structure of the system enables easy calculation of the contribution of final demands, nodes and/or emissions to the different impacts. This is plain linear algebra and may easily be performed in any mathematical software or in specialized LCA software, such as Simapro[14].

Further breakdowns and analyses are also possible through structural path analysis, monte-carlo simulations and linear programming techniques. These require more advanced modeling and will not described here.

[14] www.simapro.no

3 The background input-output model

The underlying input-output model used in this study is similar to the one developed for Klimakost (www.klimakost.no). We have constructed a model for Norway following the procedure described in the section on Computational structure on page 108, using national accounts data for Norway from 2007 (Statistics Norway 2009b), and emissions data for the same year (Statistics Norway 2009a).

In addition to constructing a standard input-output model for the domestic economy, a few extra features need to be modified in order to achieve a useful input-output model for use in environmental calculations. These are indicated below.

Capital

The consumption of fixed capital is internalized in the model by assuming this to follow the average structure of the fixed capital formation in the given year. The practical meaning of this is that we allocate the depreciation of fixed capital in 2007 to the production that year, and the difference between fixed capital formation and depreciation is treated as a final demand and may be allocated to the production of tomorrow. This way of internalizing capital is somewhere in between the flow-matrix method and the augmentation method described in Lenzen and Treolar (2005).

Imports

Ideally, a true multiregional input-output model[15] (Peters and Hertwich 2006a, 2008; Hertwich and Peters 2009) should be used in combination with trade statistics in order to estimate the economic activity and emissions occurring abroad in order to supply domestic demand. These models are hard to come by, and usually suffer from issues regarding lack of data and time lags.

We have chosen, so far, to treat imports as if they were produced with domestic technology, i.e. what is often referred to as the domestic technology assumption (DTA). This differs from the standard version of Klimakost, in which we use the German economy as a proxy for the production of Norwegian imports. As shown in e.g. Peters and Solli (2010) the error introduced by this can be significant, and future versions of the model should seek to reduce this error by treating imports in a more detailed manner. For the demonstration purposes in this study use of the DTA is deemed sufficient. Later development could incorporate coming multiregional models such as the EXIOPOL database[16].

Electricity

There is some dispute over whether the mix of sources for electricity consumed in Norway should be considered in a marginal perspective or a national average perspective. The argument is that electricity not consumed in Norway may fully or partially replace electricity in Europe. However, this argument is not flawless as this can be argued for virtually any traded commodity. The EU ETS also complicates the picture by locking emissions levels in the short term.

[15] For a simplified presentation of results from a multiregional model, see www.carbonfootprintofnations.com

[16] www.exiobase.eu

Nevertheless, we have chosen to build in the option of modifying the electricity sectors. The two basic options are to use Norwegian mix including imports (~33 g CO_2-eq./kwh) or a Nordic mix (~186 g CO_2-eq/kwh) (MiSA internal calculations).

Included emissions and impact categories

The following pollutants are currently available in the model:
CO_2, CH_4, N_2O, CO, HFCs, PFCs, SF6, NO_x, SO_x, NH_3, NMVOC, PM_{10}

We are currently working on including more types of emissions and - perhaps more importantly - primary energy use (and land use), in the model. For imports, data are often less available, so when more regional detail is added in subsequent versions of the model, the scope of emissions that are covered may be narrowed down to a few global warming pollutants. This may be overcome if new data sources emerge (as the EXIOPOL data, which has significantly more pollution data and better sector detail).

4 A model of Norwegian agriculture

This section describes how additional information on the Norwegian agricultural sectors can be used to disaggregate the national statistics and expand a national model of emissions with extra detail on the agricultural sector.

During the course of work on this project, potential improvements for future studies have been identified,. These are discussed as the issues appear in the following sections.

The core information used to derive more detailed economic information on Norwegian agriculture comes from two main sources: the ”totalkalkylen for landbruket” NILF (2009) (TOTKALK) and “driftsgranskingene” NILF, (2007a) (DG). Together they form the main basis for the disaggregation of the national agricultural sector into smaller sectors with more detail on products. DG produce a set of “standard farms” (called “reference farms” NILF), 2007b) (RF) which we use as the basis for this study. Better data could be obtained by using the raw data in DG (NILF, personal communication), but time restrictions led us to use the more readily available figures from the reference farms.

We use data for 2007 (as for the background input-output system) to minimize the necessary price corrections etc. Table 4.1 shows the raw data for the 10 different farms (average/typical farms) that we assume represent the majority of Norwegian agriculture. The first 26 rows show outputs of various products from the different reference farms. We clearly see that these are multi-output processes, although most products are produced almost entirely by one or two farm types. Rows 27→ represent the inputs for the same farms. All numbers are in 2007 NOK/farm-yr. Note that we have pruned the data in the original datasets to include only material inputs (no salaries, taxes or subsidies), and that we (for now) ignore the data on land, capital and animal stock which are also included in the datasets. Capital depreciation is, however, included.

We want to use these reference farms as a starting-point for disaggregating the agricultural sector in the national input-output table. We then need information on the total output from each farm type. TOTKALK has national totals for production of the 26 various products (see table 4.3). A few types of agricultural production are poorly covered by the reference farms, including vegetable production and greenhouses, while for other commodities the accounting differs somewhat between the RF data and TOTKALK (e.g. internal purchases between the farms, such as live animals etc.). We have therefore made some adjustments to the numbers in TOTKALK to arrive at the national totals that we want to represented by RF 1-10. These are shown in column 3 of table 4.3.

We scale the RFs so that the difference to the national total output of agricultural commodities is minimized (least squares method). The scaled RF data is shown in table 4.2 where the calculated scaling factor is shown as the first row. We now assume that these scaled RF farms represent the Norwegian agricultural sector, except for a residual sector that accounts for the products not covered by the RFs.

Table 4.1 Overview of the input and output structure of the reference farms (NOK)

	01 Melk og storfeslakt, 17 årskyr. Landet 414 bruk	02 Korn, 305 dekar korn. Landet 89 bruk	03 Sau, 127 vinterfôra sauer. Landet 86 bruk	04 Melkeproduksjon geit, 79 årsgeiter. Landet 22 bruk	05 Svin og korn, 39 avlssvin + 310 dekar korn. Landet 41 bruk	06 Egg og planteprodukter, 4117 høner + 155 dekar korn. Landet 13 bruk	07 Poteter og korn, 80 dekar poteter + 248 dekar korn. Landet 17 bruk	08 Storfeslakt/ammeku, 18 ammekyr. Landet 29 bruk	09 Frukt/bær og sau, 40 dekar frukt + 13 vinterfôra sauer. Landet 19 bruk	10 Fjørfekjøtt og planteprodukter, 66077 fjørfeslakt. Landet 13 bruk
Barley	5 914	81 614	327	-	98 216	27 790	149 867	11 648	2 063	98 093
Oats	800	57 918	-	-	30 678	27 353	7 453	875	4	6 915
Wheat	1 198	107 757	-	-	140 043	42 038	103 457	1 667	-	76 258
Other grains	-57	6 577	-	-	7 718	2 785	5 440	-	-	4 429
Oilseeds	3	9 894	-	-	12 157	-	-	-	-	1 471
Potatoes	948	1 609	37	78	160	712	343 656	4 365	52	133 256
Grass fodder	12 261	3 881	6 985	8 611	8 311	2 862	3 131	16 623	868	1 895
Grass seeds	-	4 242	-	-	4 849	-	-	-	-	12 202
Fruit and berries	721	-	247	-	724	-	-	-	319 665	7 823
Other plant products	467	1 922	74	-	8 118	404	618	-	17 289	36 530
Cow milk	458 444	-	-	-	-	50	-	-	-	9 272
Cows, live animals	18 721	-	383	-	1 143	-	-	32 512	-	-
Cows, for slaughter	69 419	-	881	858	182	-979	70	55 980	-	1 260
Other cattle	108 810	-	1 277	3 890	-59	872	-	175 697	-	1 591
Goat milk	-	-	-	227 079	-	-	-	-	-	-
Goats, live and slaughter	1	-	110	14 116	-	-	-	-	-	-
Pigs, slaughter	718	-	154	-	1 140 240	-	-	-	-	-
Sows and boars, live and for slaughter	16	-	1	-	120 644	-	-	-	-	-
Piglets	-13	-	-	-	396 141	-	-	-	-	-
Sheep, live and slaughter	2 175	965	156 639	1 476	59	-	-	186	10 271	-
Wool	323	175	21 124	122	11	-	-	44	2 053	-
Eggs	36	-	55	-	3	1 028 108	-	33	-	21 414
Chickens	3	-	27	-24	-13	-55 506	-	-19	-	1 888 915
Other income from animals	303	-	10 503	-714	80	-	-	265	71	1 970
Rental income (machinery)	13 366	15 220	9 285	8 979	17 535	4 257	13 203	12 808	8 350	28 796
Other income (including land rent)	1 676	7 706	1 105	-	7 400	14 590	2 812	2 271	17 710	1 737
Sum	**696 253**	**299 480**	**209 214**	**264 471**	**1 994 340**	**1 095 336**	**629 707**	**314 955**	**378 396**	**2 333 827**
Expenses except salaries + taxes/subsidies										
Seed grains	1 807	24 028	129	228	22 426	12 683	26 525	2 241	1 128	24 235
Seed potatoes	112	226	32	61	-	-	25 136	695	25	4 175
Other seeds and plants	4 360	339	2 462	827	630	4	20	5 099	3 436	2 854
Fertilizer	31 754	45 535	19 509	13 684	40 930	17 407	69 652	31 150	8 629	42 458

Lime	1 872	3 878	957	363	3 128	2 399	560	2 172	95	1 434
Pesticides	1 656	17 860	632	233	25 714	5 652	45 930	1 664	23 039	26 154
Preservatives	7 533	12	2 708	4 185	46	103	-	5 681	324	116
Feed concentrates	166 769	115	35 283	96 273	646 247	486 159	-	41 494	3 090	868 305
Other fodder	22 066	452	19 017	12 215	24 147	2 552	-	24 530	384	-2 483
Misc for keeping animals	30 425	91	12 891	6 629	54 979	4 140	-	7 327	1 142	6 892
Purchase of animals	18 621	62	8 684	1 922	173 347	120 945	-	31 233	257	398 155
Other consumables	25 831	4 277	11 757	16 768	34 853	13 127	8 216	18 279	24 230	69 028
Maintenance of machinery and equipment	29 838	11 859	16 811	16 386	26 472	19 514	25 231	23 348	20 046	30 116
Maintenance of tractors	16 074	12 174	9 478	9 125	14 841	14 214	21 613	11 459	6 719	14 456
Maintenance of combine harvester s	57	4 968	-	-	3 863	466	1 028	15	-	830
Maintenance of cars	1 278	2 647	1 261	3 119	3 624	457	1 600	1 904	1 831	6 053
Maintenance of buildings	31 156	9 863	15 093	26 261	29 455	33 058	15 628	20 248	12 015	47 389
Maintenance of water supply, ditches etc	4 658	6 089	3 352	3 568	12 082	3 056	8 400	7 589	10 698	12 058
Fuel	20 796	19 395	11 883	13 074	29 649	17 597	27 327	18 935	7 657	27 710
Machine rental	42 975	28 308	19 512	43 756	102 968	16 517	59 930	45 448	7 505	87 309
Land rental	10 893	24 219	4 828	850	40 979	14 613	42 757	12 340	2 474	20 773
Insurance	20 593	15 661	11 401	12 030	33 156	20 238	20 561	16 913	8 445	38 322
Electricity	20 498	7 613	9 287	14 213	38 082	23 250	14 571	11 670	6 441	33 876
Adm and other fixed costs	43 520	27 285	29 842	35 162	56 768	34 708	35 964	33 137	30 463	57 381
Depr machinery and equip	20 600	13 634	10 466	10 027	17 921	12 447	37 043	17 110	8 317	25 750
Depr tractor s	21 703	19 342	14 770	10 485	19 409	11 347	20 201	15 478	13 565	22 023
Depr combine harv.	157	7 806	-	-	9 290	2 905	6 004	720	-	5 406
Depr cars	804	1 037	879	1 408	3 167	38	309	743	1 182	3 517
Depr buildings	49 013	22 529	21 633	23 090	108 086	87 949	31 264	32 108	14 152	98 595
Depr land, roads, ditches etc.	1 929	1 761	840	1 033	1 943	928	4 708	428	9 631	1 567
Depr office machinery	116	12	113	113	249	-	-	75	-	163
Sum	**649 464**	**333 077**	**295 510**	**377 088**	**1 578 451**	**978 473**	**550 178**	**441 233**	**226 920**	**1 974 617**

Table 4.2 Inputs and outputs of the reference farms scaled to minimize the error when comparing to total outputs in TOTKALK using the least squares method. MNOK

	01 Melk og storfeslakt, 17 årskyr. Landet 414 bruk	02 Korn, 305 dekar korn. Landet 89 bruk	03 Sau, 127 vinterfôra sauer. Landet 86 bruk	04 Melkeproduksjon geit, 79 årsgeiter. Landet 22 bruk	05 Svin og korn, 39 avlssvin + 310 dekar korn. Landet 41 bruk	06 Egg og planteprodukter, 4117 høner + 155 dekar korn. Landet 13 bruk	07 Poteter og korn, 80 dekar poteter + 248 dekar korn. Landet 17 bruk	08 Storfeslakt/ammeku, 18 ammekyr. Landet 29 bruk	09 Frukt/bær og sau, 40 dekar frukt + 13 vinterfôra sauer. Landet 19 bruk	10 Fjørfekjøtt og planteprodukter, 66077 fjørfeslakt. Landet 13 bruk
Scaling factor [p]	12 940	3 409	4 758	337	2 250	628	1 246	2 316	1 050	589
Barley	77	278	2	-	221	17	187	27	2	58
Oats	10	197	-	-	69	17	9	2	0	4
Wheat	16	367	-	-	315	26	129	4	-	45
Other grains	-1	22	-	-	17	2	7	-	-	3
Oilseeds	0	34	-	-	27	-	-	-	-	1
Potatoes	12	5	0	0	0	0	428	10	0	78
Grass fodder	159	13	33	3	19	2	4	39	1	1
Grass seeds	-	14	-	-	11	-	-	-	-	7
Fruit and berries	9	-	1	-	2	-	-	-	336	5
Other plant products	6	7	0	-	18	0	1	-	18	22
Cow milk	5 932	-	-	-	-	0	-	-	-	5
Cows, live animals	242	-	2	-	3	-	-	75	-	-
Cows, for slaughter	898	-	4	0	0	-1	0	130	-	1
Other cattle	1 408	-	6	1	-0	1	-	407	-	1
Goat milk	-	-	-	76	-	-	-	-	-	-
Goats, live and slaughter	0	-	1	5	-	-	-	-	-	-
Pigs, slaughter	9	-	1	-	2 565	-	-	-	-	-
Sows and boars, live and for slaughter	0	-	0	-	271	-	-	-	-	-
Piglets	-0	-	-	-	891	-	-	-	-	-
Sheep, live and slaughter	28	3	745	0	0	-	-	0	11	-
Wool	4	1	101	0	0	-	-	0	2	-
Eggs	0	-	0	-	0	645	-	0	-	13
Chickens	0	-	0	-0	-0	-35	-	-0	-	1 112
Other income from animals	4	-	50	-0	0	-	-	1	0	1
Rental income (machinery)	173	52	44	3	39	3	16	30	9	17
Other income (including land rent)	22	26	5	-	17	9	4	5	19	1
Sum	9 010	1 021	995	89	4 487	687	784	730	397	1 374
Seed grains	23	82	1	0	50	8	33	5	1	14

Seed potatoes	1	1	0	0	-	-	31	2	0	2
Other seeds and plants	56	1	12	0	1	0	0	12	4	2
Fertilizer	411	155	93	5	92	11	87	72	9	25
Lime	24	13	5	0	7	2	1	5	0	1
Pesticides	21	61	3	0	58	4	57	4	24	15
Preservatives	97	0	13	1	0	0	-	13	0	0
Feed concentrates	2 158	0	168	32	1 454	305	-	96	3	511
Other fodder	286	2	90	4	54	2	-	57	0	-1
Misc for keeping animals	394	0	61	2	124	3	-	17	1	4
Purchase of animals	241	0	41	1	390	76	-	72	0	234
Other consumables	334	15	56	6	78	8	10	42	25	41
Maintenance of machinery and equipment	386	40	80	6	60	12	31	54	21	18
Maintenance of tractors	208	42	45	3	33	9	27	27	7	9
Maintenance of combine harvesters	1	17	-	-	9	0	1	0	-	0
Maintenance of cars	17	9	6	1	8	0	2	4	2	4
Maintenance of buildings	403	34	72	9	66	21	19	47	13	28
Maintenance of water supply, ditches etc	60	21	16	1	27	2	10	18	11	7
Fuel	269	66	57	4	67	11	34	44	8	16
Machine rental	556	97	93	15	232	10	75	105	8	51
Land rental	141	83	23	0	92	9	53	29	3	12
Insurance	266	53	54	4	75	13	26	39	9	23
Electricity	265	26	44	5	86	15	18	27	7	20
Adm and other fixed costs	563	93	142	12	128	22	45	77	32	34
Depr machinery and equip	267	46	50	3	40	8	46	40	9	15
Depr tractor	281	66	70	4	44	7	25	36	14	13
Depr combine harvesters	2	27	-	-	21	2	7	2	-	3
Depr cars	10	4	4	0	7	0	0	2	1	2
Depr buildings	634	77	103	8	243	55	39	74	15	58
Depr land, roads, ditches etc.	25	6	4	0	4	1	6	1	10	1
Depr office machinery	2	0	1	0	1	-	-	0	-	0
Sum	8 404	1 136	1 406	127	3 551	614	685	1 022	238	1 162

Table 4.3 Comparisons between the totals in TOTKALK, the totals obtained by scaling of the reference farms, and finally an indication of which outputs have been modified from TOTKALK to comply with the scope of the reference farms (personal communication, NILF). MNOK

	Total estimated output	Totals from totalkalkylen	Adjusted numbers
Barley	868	854	
Oats	309	444	
Wheat	902	821	
Other grains	50	96	
Oilseeds	62	41	
Potatoes	535	558	
Grass fodder	273	138	
Grass seeds	33	23	
Fruit and berries	353	345	
Other plant products	72	1 400	73
Cow milk	5 938	5 939	
Cows, live animals	322	-40	243
Cows, for slaughter	1 033	2 852	1000
Other cattle	1 824	77	1 852
Goat milk	76	78	
Goats, live and slaughter	5	-0	
Pigs, slaughter	2 575	2 580	
Sows and boars, live and for slaughter	272	-6	273
Piglets	891	-	896
Sheep, live and slaughter	789	767	
Wool	108	137	
Eggs	659	657	
Chickens	1 077	1 077	
Other income from animals	56	292	
Rental income (machinery)	386	508	
Other income (including land rent)	107	200	
Sum	**19 575**	**19 837**	
Seed grains	218	219	
Seed potato	38	32	
Other seeds and plants	88	504	
Fertilizer	960	1 101	
Lime	57	93	
Pesticides	247	316	
Preservatives	126	134	

Feed concentrates	4 728	4 733	
Other fodder	493	255	
Misc for keeping animals	606	1 034	
Purchase of animals	1 056	-	
Other consumables	616	1 343	
Maintenance of machinery and equipment	708	351	
Maintenance of tractors	409	-	
Maintenance of combine harvesters	28	-	
Maintenance of cars	53	103	
Maintenance of buildings	711	1 124	
Maintenance of water supply, ditches etc	174	-	
Fuel	576	1 027	
Machine rental	1 241	743	
Land rental	445	-	
Insurance	562	237	
Electricity	512	883	
Adm and other fixed costs	1 147	895	
Depr machinery and equip	524	2 875	*
Depr tractor	560	-	*
Depr combine harvesters	64	-	*
Depr cars	31	178	*
Depr buildings	1 306	1 995	*
Depr land, roads, ditches etc.	58	206	*
Depr office machinery	3	-	*
Sum	**18 346**	**20 381**	

**Depriciation of capital goods is treated different in the DG accounting and the national accounts. The capital depreciation is therefore scaled to the official national account data (NILF, personal communication) in the matching stage of model construction.*

Table 4.4 Animal stocks (or turnover for short-lived animals) (thousands) used to determine direct emissions from the reference farms

	01 Melk og storfeslakt, 17 årskyr. Landet 414 bruk	02 Korn, 305 dekar korn. Landet 89 bruk	03 Sau, 127 vinterfôra sauer. Landet 86 bruk	04 Melkeproduksjon geit, 79 årsgeiter. Landet 22 bruk	05 Svin og korn, 39 avlssvin + 310 dekar korn. Landet 41 bruk	06 Egg og planteprodukter, 4117 høner + 155 dekar korn. Landet 13 bruk	07 Poteter og korn, 80 dekar poteter + 248 dekar korn. Landet 17 bruk	08 Storfeslakt/ammeku, 18 ammekyr. Landet 29 bruk	09 Frukt/bær og sau, 40 dekar frukt + 13 vinterfôra sauer. Landet 19 bruk	10 Fjørfekjøtt og planteprodukter, 66077 fjørfeslakt. Landet 13 bruk
Milk cattle	248	-	-	-	-	-	-	-	-	0,2
Other cattle	437	-	1	0,3	0,7	0,5	-	78	-	0,2
Sheep	27	5	632	0	-	-	-	1	13	-
Goats	-	-	1	28	-	-	-	-	-	-
Pigs	5	-	0	-	1 416	-	-	-	-	-
Egg chickens	3	-	5	0,1	0,2	2 611	-	2	-	3 883
Slaughter chickens	-	-	-	-	-	1 656	-	-	-	50 515

Direct emissions estimates

Please beware that the focus of this early stage modeling is not to produce as accurate as possible emissions estimates, but more to demonstrate the ability to create a holistic, aggregated input-output based model of Norwegian agriculture by using the information contained in DG/RF. In order to demonstrate the model we need some rough emission estimates for all types of farm activities. The direct emissions from manure management, enteric emissions from ruminants, and emissions from application of fertilizer were calculated by using the animal stock numbers from table 4.4 in combination with the emissions factors derived in Pettersen (2010). Since data for sheep and goats are missing in the report, some crude simplifications were made in the model, to save time. For simplicity sheep and goats were counted as fractions of a (milk) cow (0,1 and 0,05). Later versions will of course address this issue by complementing the figures of Pettersen with figures for sheep and goats. For fuel and fertilizer we converted the amounts (in NOK) to physical units by assuming a flat price of 10 NOK/kg N in fertilizer (amount of N is used to derive N_2O emission). For fuel (assumed to be 100% diesel) we assumed an average price of 3,30 NOK/liter diesel (taken from totalkalkylen).

Trade and transport margins (TTM)

The inputs to RF include taxes trade and transport margins. We need to correct for this by subtracting taxes (if any) and reallocating the trade and transport margins to the trade and transport sectors in the background economy. The scope of this project does not allow for a detailed investigation of the trade and transport margin. We have made some crude assumptions in table 4.5.

Table 4.5 Crude assumptions for taxes, trade- and transport margins. The transport margins on animal feed and fertilizer were taken from TOTALKALK.

	taxes	trade	trsp
Seed grains	3 %	5 %	3 %
Seed potato	3 %	5 %	3 %
Other seeds and plants	3 %	5 %	3 %
Fertilizer	3 %	5 %	7 %
Lime	3 %	5 %	5 %
Pesticides	3 %	5 %	5 %
Preservatives	3 %	5 %	5 %
Feed concentrates	3 %	5 %	6 %
Other fodder	3 %	5 %	5 %
Misc for keeping animals	3 %	5 %	5 %
Purchase of animals	3 %	5 %	3 %
Other consumables	3 %	5 %	0 %
Maintenance of machinery and equipment	3 %	5 %	0 %
Maintenance of tractors	3 %	5 %	0 %
Maintenance of combined harvesters	3 %	5 %	0 %
Maintenance of cars	3 %	5 %	0 %
Maintenance of buildings	3 %	5 %	0 %
Maintenance of water supply, ditches etc	3 %	5 %	0 %
Fuel	3 %	10 %	0 %
Machine rental	3 %	10 %	0 %
Land rental	3 %	0 %	0 %
Insurance	3 %	0 %	0 %
Electricity	0 %	0 %	0 %
Adm and other fixed costs	3 %	10 %	0 %
Depr machinery and equip	0 %	10 %	0 %
Depr tractor	0 %	10 %	0 %
Depr combined harvesters	0 %	10 %	0 %
Depr cars	0 %	10 %	0 %
Depr buildings	0 %	10 %	0 %
Depr land, roads, ditches etc.	0 %	10 %	0 %
Depr office machinery	0 %	10 %	0 %

As we are usually interested in products in the context of food and the environment, the *reference farms* as such, are not that interesting. Their construction and the breakdown of farm types is based on other motives than providing environmental information (NILF, personal communication). We therefore use the *industry technology assumption* to allocate emissions between products, and construct input-output tables for agriculture based on the weighted average production of the commodity. This means that all commodities produced by a reference farm are assigned the same input and emissions intensities per NOK. By applying this to all the inputs and emissions, we can split the inputs to and emissions from the agricultural sector commodity by commodity.

Matching

One central element in the model construction is the actual connection points between the inputs (and outputs for downstream matching) in RF and the sectors defined in the SNA

(NACE classification). In this study we have performed a crude mapping of all the inputs in RF to the NACE sectors in the economy, for the purpose of assigning upstream inputs to NACE categories. We used the agricultural sector totals as a *tuning target* for the matching matrix. In addition we had to create a "rest of agriculture sector" which makes up the balance between the SNA totals and the RF totals. This is also used as a "collection sector" that has all the other agricultural commodities as inputs in subsequent calculations. This sector then serves as the source of agricultural commodities for downstream use (i.e. downstream sectors/consumers purchase average of all agricultural commodities). This is done since this project did not allow for detailing the downstream linkages for the disaggregated agricultural commodities.

Table 4.6 Matching between agricultural inputs and commodities in agriculture and the SNA system. Note that capital depreciation is scaled by a factor of 2.5 to account for differences in the treatment of this in the SNA and RF

	'Såkorn'	'Settepoteter'	'Såvarer og planter'	'Handelsgjødsel'	'Kalk'	'Plantevernmidler'	'Konserveringsmidler'	'Kraftfôr'	'Annet innkjøpt fôr'	'Diverse til husdyrholdet'	'Innkjøp av dyr'	'Andre forbruksartikler'	'Maskiner og redskaper, vedlike'	'Traktorer, vedlikehold'	'Skurtresker, vedlikehold'	'Yrkesbil, vedlikehold'	'Driftsbygninger, vedlikehold'	'Jord, grøfter & vannanlegg, ve'	'Drivstoff'	'Maskinleie'	'Jordleie'	'Forsikring'	'Elektrisk kraft'	'Adm. og andre faste kostn.'	'Avskr. maskiner & redsk.'	'Avskr. traktor'	'Avskr. skurtresker'	'Avskr. yrkesbil'	'Avskr. driftsbygn.'	'Avskr. jord, veger, grøfter og'	'Avskr. kontormaskiner'
'Bygg'	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Havre'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Hvete'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Annet korn'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Oljefrø'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Poteter'	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Grovfôr'	0	0	0	0	0	0	0	0	0,1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Grasfrø'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Frukt og bær'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Andre planteprodukter'	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Storfe, melk'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Storfe, livdyr'	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Kuslakt'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Annet storfeslakt'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Geit, melk'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Geit, livdyr og slakt'	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

'Slaktegris'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Purker og råner, livdyr og sla'	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Smågris'	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Sau, livdyr og slakt'	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Sau, ull'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Fjørfe, egg'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Fjørfekjøtt'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Andre inntekter husdyrhold'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Leieinntekter (maskindel)'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
'Andre inntekter (inkl. jordlei'	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0
Products of agriculture, hunting and related services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0
Products of forestry, logging and related services	0	0	0	0	0	0	0	0	0,1	0	0	0	0	0	0	0	0	0	0	0	0,6	0	0	0	0	0	0	0	0	0	0
Fish and other fishing products; services incidental of fishing	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coal and lignite; peat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crude petroleum and natural gas; services incidental to oil and gas extraction excluding surveying	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Uranium and thorium ores	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Metal ores	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Other mining and quarrying products	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Food products and beverages	0	0	0	0	0	0	0	1	0,8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tobacco products	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Textiles	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wearing apparel; furs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leather and leather products	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Wood and products of wood and cork (except furniture); articles of straw and plaiting materials	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pulp, paper and paper products	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Printed matter and recorded media	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coke, refined petroleum products and nuclear fuels	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chemicals, chemical products and man-made fibres	0	0	0	1	0	1	1	0	0	0,3	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Rubber and plastic products	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Other non-metallic mineral products	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Basic metals	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fabricated metal products, except machinery and equipment	0	0	0	0	0	0	0	0	0	0,1	0	0,2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Machinery and equipment n.e.c.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0,8	0,8	0,8	0,8	0	0	0
Office machinery and computers	0	0	0	0	0	0	0	0	0	0	0	0,4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2,5
Electrical machinery and apparatus n.e.c.	0	0	0	0	0	0	0	0	0	0,1	0	0,4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Radio, television and communication equipment and apparatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Medical, precision and optical instruments, watches and clocks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Motor vehicles, trailers and semi-trailers	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0,5	0,5	0,5	0	0	0
Other transport equipment	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0,5	1	1	0	0	0	0	0	0	0	0	1,2	1,2	1,2	1,2	0	0	0
Furniture; other manufactured goods n.e.c.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Secondary raw materials	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Electrical energy, gas, steam and hot water	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Collected and purified water, distribution services of water	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Construction work	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	2	2	0
Trade, maintenance and repair services of motor vehicles and motorcycles; retail sale of automotive fuel	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0,5	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wholesale trade and commission trade services, except of motor vehicles and motorcycles	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Retail trade services, except of motor vehicles and motorcycles; repair services of personal and household goods	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hotel and restaurant services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Land transport; transport via pipeline services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Water transport services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Air transport services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Supporting and auxiliary transport services; travel agency services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,2	0	0	0	0	0	0	0	0	0	0	0
Post and telecommunication services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,1	0	0	0	0	0	0	0
Financial intermediation services, except insurance and pension funding services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,1	0	0	0	0	0	0	0
Insurance and pension funding services, except compulsory social security services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Services auxiliary to financial intermediation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Real estate services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0,5	0
Renting services of machinery and equipment without operator and of personal and household goods	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,1	0	0	0	0	0	0	0	0	0	0	0
Computer and related services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Research and development services	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Other business services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0	0	0	0,3	0	0	0	0	0	0	0
Public administration and defence services; compulsory social security services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Education services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Health and social work services	0	0	0	0	0	0	0	0	0	0,5	0	0	0	0	0	0	0	0	0	0	0	0	0	0,5	0	0	0	0	0	0	0
Sewage and refuse disposal services, sanitation and similar services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Membership organisation services n.e.c.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Recreational, cultural and sporting services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Other services	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Internal trade

The analysis of trade within the agricultural sector could be improved significantly. The sample results show that the connections and interdependencies between various agricultural subsectors are vital in the understanding of the value chains connected to agriculture. So far we have only included some direct connections between the sub-sectors, but several indirect connections (via e.g. fodder production) should be detailed further in a later model.

Equations

The following equations are used in the process of converting the reference farm data to sectoral input-output data on a commodity basis. Note that this is after the scaling of the different farm types to achieve the lowest least squares error in total commodity outputs.

i= commodities from agriculture
j= reference farm types

S(i,j)= Matrix with total output per farm type
P(i,1)= Vector of total volume of each commodity

A matrix with the market share of each commodity per farm type M, is then

$$M = S \times \hat{P}^{-1}$$

U(k,j)= Matrix with total inputs per farm type
Q=Vector of total volume of each farm type

The normalized use matrix per farm type, Un, is then

$$U_n = U \times \hat{Q}^{-1}$$

The market share weighted inputs are then

$$U_{nc} = U_n \times M'$$

Scaling to national totals for commodities Ut gives

$$U_t = U_{nc} \times \hat{P}$$

Further, taxes, trade and transport margins are extracted from all the uses, resulting in a total use matrix U_t in basic prices. The taxes are simply subtracted, while the trade and transport margins are distributed to the trade and transport sectors in the same proportions as the total use of TTM in the agricultural sector in the national accounts.

The capital consumption indicated in the RF is different to that of the national accounts (NILF, personal communication). We want to be consistent with the SNA and tune the capital depreciation to correspond approximately to the total in the national accounts.

The resulting system is merged with the previously developed IO-model to give a consistent model of the Norwegian economy, with improved detail in agriculture. The merging is done via the previously mentioned matching matrix.

4.1.1 Improvements to the model

There are however a number of improvement that should be done to the model in the future. Aside from the issues already mentioned in this report, the following should be considered:

Downstream use

Due to the limited time available for this exploratory possibility study, downstream use of the agricultural commodities was sourced into a collection sector "rest of agriculture" which in turn distributes to the other sectors in the economy as in the national accounts. This makes the connection to final demand and important feed value chains less clear and an improved downstream linkage is required to answer questions connected to these issues.

Aside from assigning the output of the different agricultural sectors to other sectors or final uses, there is also a need for further disaggregation of other economic sectors of particular relevance to agriculture. This is especially evident in the food production and chemicals sectors, where the production of different types of animal feed, and production of fertilizer, could be separated out as own sectors.

Allocation of emissions and inputs

Instead of using the industry technology assumption, a more eclectic approach can be used, where some emissions (e.g. enteric CH_4) are completely allocated to the ruminants, and some (less intuitively separable) inputs and emission are allocated based on the industry technology assumption. This allows for a more precise distribution of emissions on various products from the multi-output processes that farms are.

Geography

Imports are, as mentioned, treated as if they were produced using domestic technology. This is obviously not correct, and may have a significant influence on results, especially for items that rely heavily on imported products that are not produced using similar technologies in Norway (like animal fodder). Subsequent models should try to incorporate this in a more comprehensive manner.

Direct emission estimates

In addition connections to other models for more precise direct emissions estimates could be done (like the HOLOS-model). Using more parametrized models for the calculation of direct emissions, in combination with more detailed farm-level data in DG, provides a promising potential for significantly improved inventories that are able to separate between climate zones, soil types, farm practices etc.

More impacts and land use

More types of environmental impact could and should be included, along with a more detailed treatment of both land use *per se* (data is available) and the global warming consequences of various types of land use and land transformation.

Additional activities

Some significant activities connected to producing food in Norway are left out since we focused on farms covered by DG. This applies in particularly to fish farming. This sector not only produces large amounts of food, it is also heavily entangled in the feed production chains of agriculture, both up- and downstream. Future models aiming at understanding the links between consumption and production of food should address this issue.

Why develop an IO-based model?

The overall advantage with an input—output based model seems to lie in the fact that all upstream impacts are covered, and that the economic framework enables seamless connections to models describing the effects of different economic policies affecting agriculture (like PE models). In addition the methodological framework is standardized and consistent, and makes it easy to also investigate downstream linkages. The emergence of multi-regional models makes treatment of imported goods easier. The standardized structure also makes such a model well suited for investigations on farm level data (i.e statistical analysis of the DG data).

Disadvantages compared to e.g. process based LCA inventories, is the aggregation level and allocation used in multi-output systems (industry technology assumption). In LCA the cause-effect relationships between inputs, emissions and outputs can be established at a more detailed level.

However, some of this can be improved in IO based models, if more “agri-knowledge” is used in the conversion from reference farms to commodity sectors as opposed to the simple industry technology assumption that is used in this possibility study.

To keep things simple in this pre-study phase, we have chosen to use a version of the IO-model using the domestic technology assumption on imports, i.e. all imports are produced with domestic technology. For electricity we have used the official domestic emissions.

5 Sample results from the model

Below are some sample results from the preliminary model. Please remember the shortcomings and simplifications that have been mentioned earlier. Do not use these results in presentations or other contexts outside the further developments of this tool.

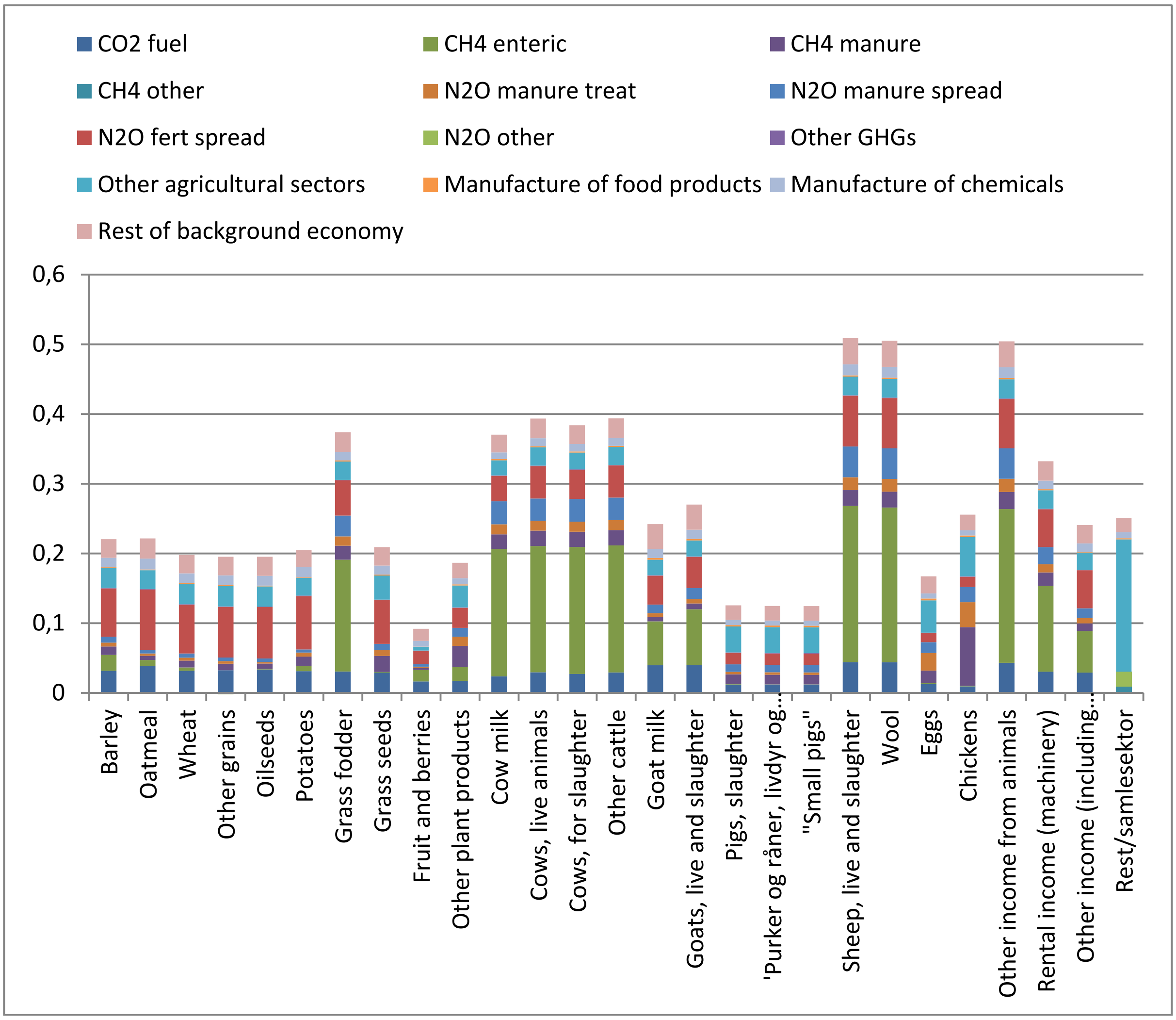

Figure 5.1 Life cycle GHG emissions of the products in the model. kg CO_2-eq./NOK out of farm. Emissions are allocated to the sectors where they occur. Direct emissions from the sector are split into different component

From figure 5.1 we clearly see that the emissions per NOK produced vary significantly across commodities, but perhaps not so much as one intuitively would think. If enteric methane is taken out of the figure the emissions become even more similar. This is an interesting finding. Further we observe that direct emissions in the agricultural sectors completely dominate emissions.

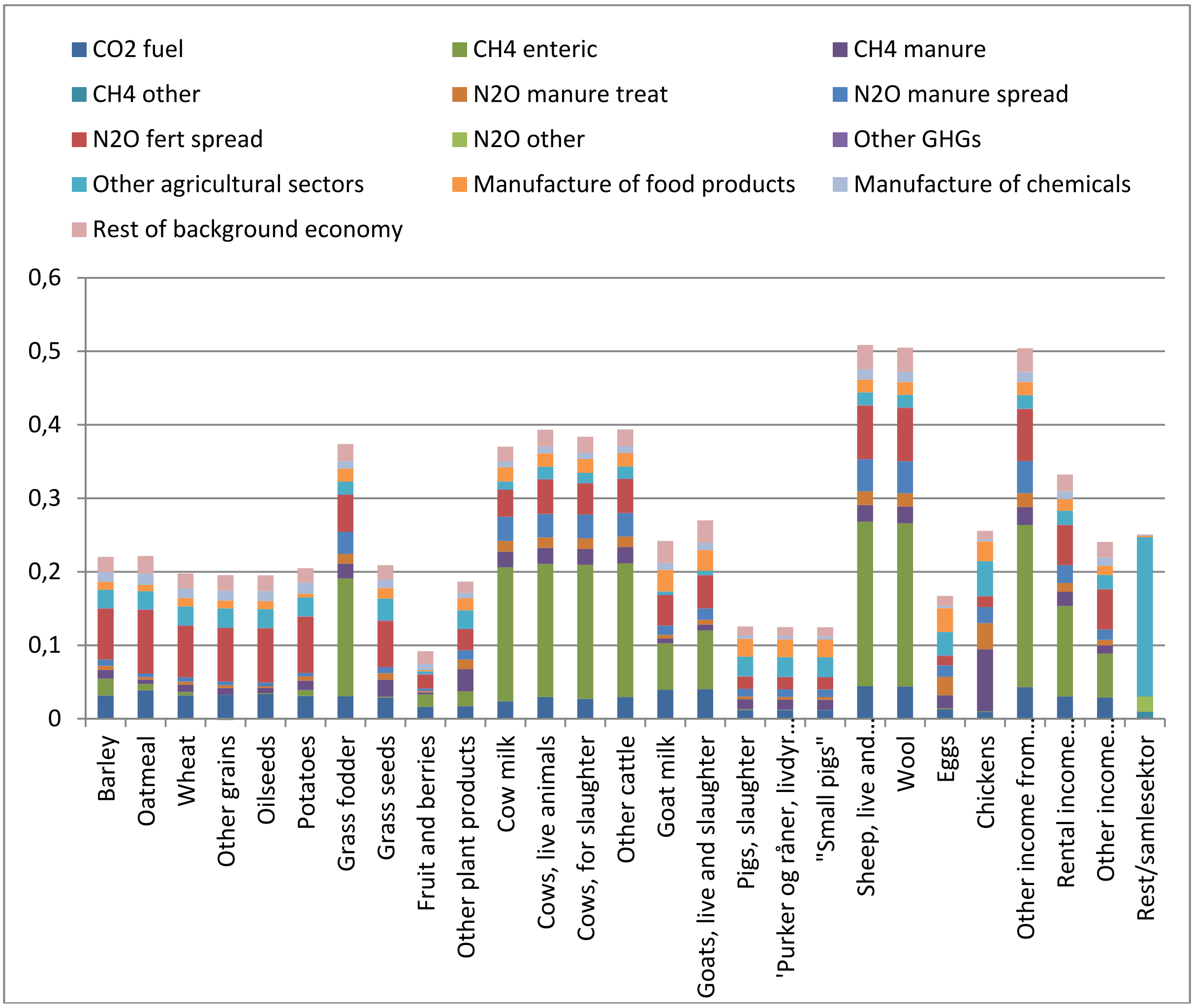

Figure 5.2 Life cycle GHG emissions of the products in the model. kg CO_2-eq./NOK out of farm. Emissions are allocated to the inputs to each sector. Direct emissions from the sector are split into different components

If we view the same results in the perspective of allocating emissions to the inputs to the various sectors, the picture look very much similar. The only difference is that manufacture of food products becomes more important since this now is an important linkage to the other agricultural sectors through fodder production. When downstream linkages are improves, the effects of fodder production may be explored in more detail.

If we instead move on to looking at emissions *per kg of product* (fig 5.3) we see much larger differences. Grains now stand out as low emissions commodities, while meat from ruminants move up to the top of the emissions list. These figures are in line with the figures found in the literature in part 1, although the methodological differences of approaches result in some discrepancies.

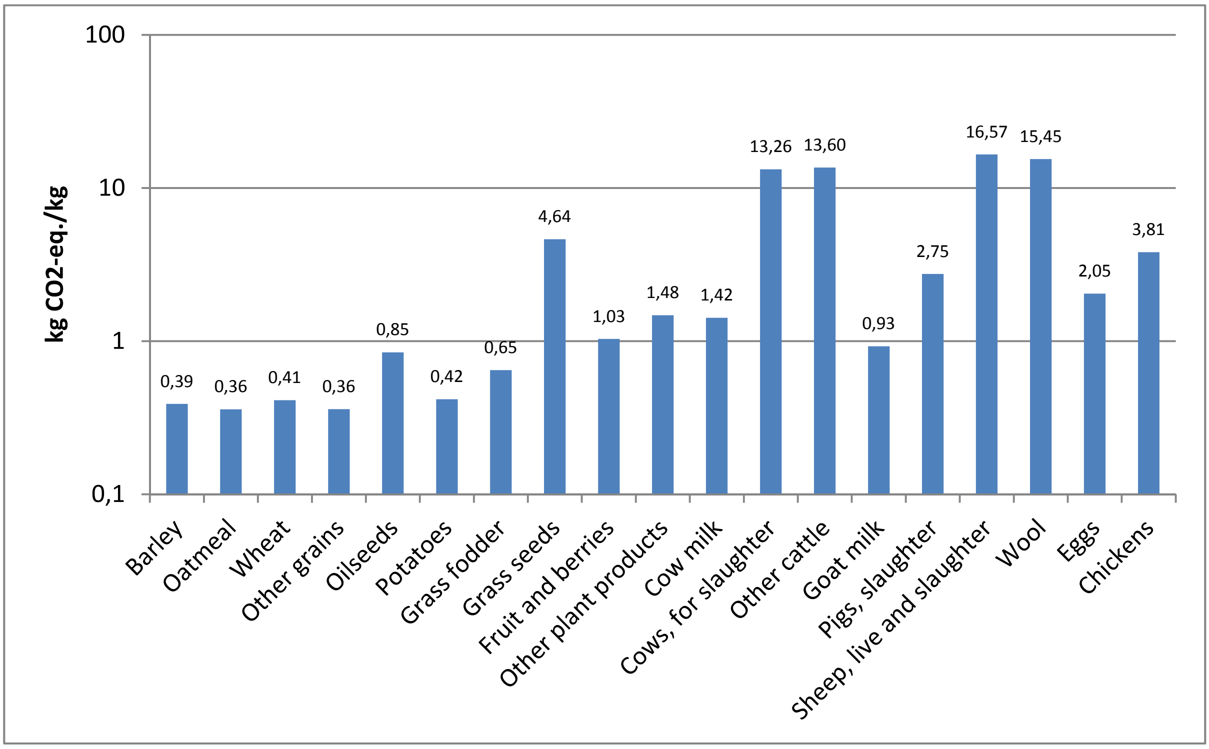

Figure 5.3 GHG emissions per kg @ farm gate. Prices of commodities are taken from totalkalkylen

One question that arises is how the results from such a model can be used and which questions can they answer. The potential of integrating the model with economic models gives the possibility of exploring the environmental consequences of economic policies or external shocks. This, together with the explicit linkage between consumption and production and emissions, may help guide the design of policies aimed at reducing the environmental footprint of our food consumption.

The overall picture, what about emissions downstream the farm?

So far we have focused mainly on the disaggregation of the agricultural sector and what happens directly and upstream the sub-sectors we have constructed. However, just as important as the production emissions, is the linkage to the consumers of food, most notably households. Therefore we also present results starting at the final demand for food products in the households. The results presented in this section are calculated with a similar model to the one developed above, but keeping the agricultural sector as one single sector. This is because we still haven't solved the downstream linkages to consumers via the food sector. Agricultural- and food commodities are therefore treated at an aggregated average level. Still, the figures give an overview of the dominating industries connected to producing the food for household consumption.

In order to estimate the real emissions from food purchases in the households, we have to account for the fact that there is a trade and transport component to be accounted for in the value chain from agriculture and food production to the consumer. In addition there are emissions connected to transportation of the food from the shop to the consumer (private transport) and processing and waste emissions. These are not considered here; although allocation of the correct fractions of these types of activities to food consumption probably is doable using transport survey statistics, energy use statistics, and waste statistics. It is, however, outside the modeling scope in this study.

Figure 5.4 shows the overall emissions profile for Norwegian household food consumption in 2007. Please note that the figures do not include food produced for public and export demand (directly or indirectly). Part 1 already shows some results for these steps in the value chain calculated with process-LCA.

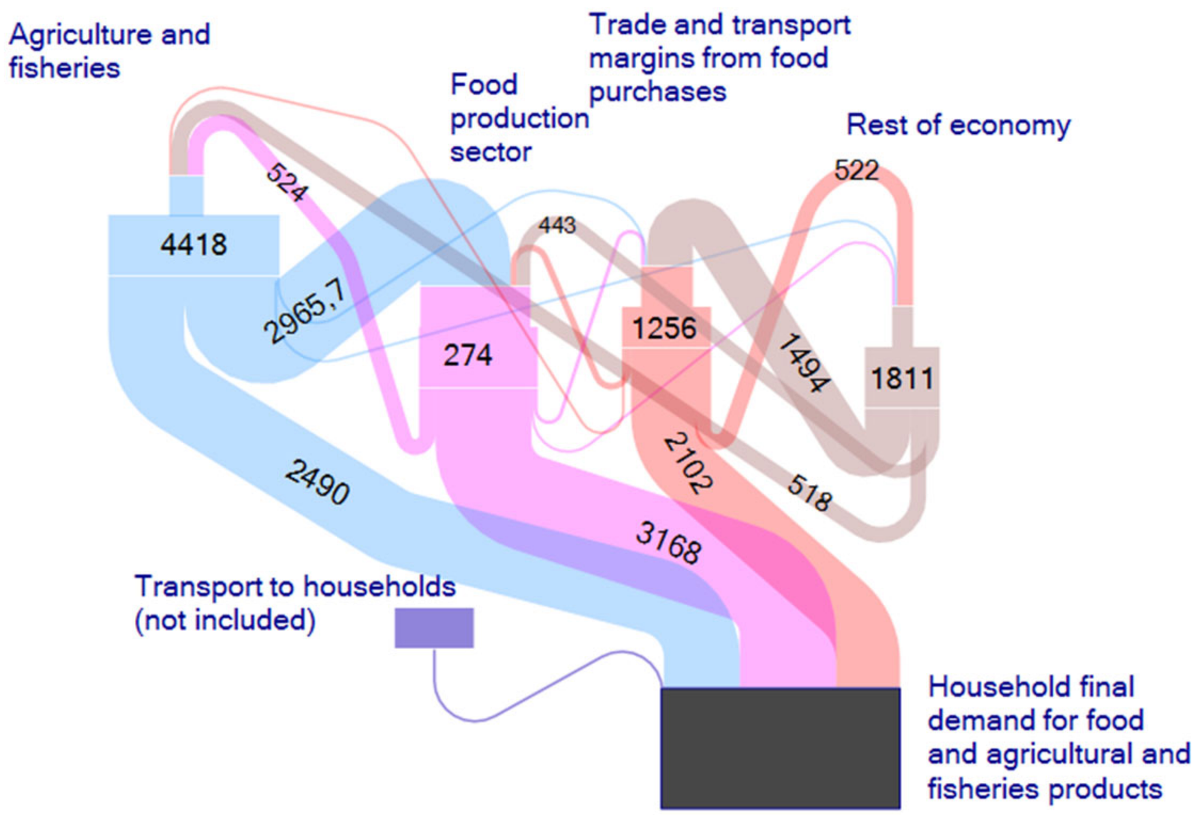

Figure 5.4 Embodied emissions from household consumption of food in 2007. [kT CO_2-eq.]. The numbers in the rectangular boxes are the direct emissions occurring in each sector while the numbers in the flows are embodied emissions

6 Where do we go from here?

The question on where to go from here really depends on what types of questions we want to answer. If the aim is to evaluate the consequences of various policy options, integration with economic models should be done. As mentioned earlier others have combined the capabilities of a partial equilibrium model (endogenize some variables) in combination with the functionality of an environmentally extended input-output model (Wolf et al. 2011).

Further there are several steps that are needed in order to achieve a model of the desired accuracy:

- Efforts to improve the agricultural sector disaggregation
- Improve the integration in existing IO framework (downstream, food sector, fertilizer production)
- Improve geographical coverage for international value chains
- Connect and adapt to ongoing initiatives and models including eg. jordmod, HOLOS, CAPRI for better direct emissions estimates
- Improve allocation procedures by applying more "agri-knowledge"
- Include a wider range of impacts, including land use and consequences of land use.

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PART 4: Environmental indicators for agriculture. Current status and possible directions

1 Introduction and background

Indicators are useful for describing current situations or tracking trends in many different fields. Environmental indicators can convey information about a present state or how it has evolved over time. They may be related to explicit targets or limits, which an enterprise may have set in its own or its customers' interests or to comply with regulations.

Agriculture can have a wide range of impacts on the environment. To reflect these impacts, many different indicators may therefore be considered relevant. Indicators may improve our understanding of cause and effect relationships in the agriculture-environment nexus. It is important to note that effects can be positive – for instance, some plants are dependent on grazing animals.

Environmental indicators for agriculture should contribute to "simplified description of complex reality, better communication with non-specialists, analysis of environmental trends in longer time series, building a common basis for discussion and identifying priorities in political decision-making" (European Environment Agency 2005).

The DPSIR model (Driving force, Pressure, State, Impact, Response) provides a way of structuring environmental indicators (Finansdepartementet 2005). These are some examples of how indicators may be categorized according to the DPSIR model:

- Driving forces – use of inputs, e.g. land, artificial fertilisers, pesticides
- Pressures – e.g. emissions of nitrogen, phosphorous, methane, nitrous oxide, ammonia
- States – e.g. levels of nitrogen and phosphorous in affected watercourses
- Impacts – e.g. numbers of red-listed species in pasture land, fish numbers in watercourses affected by agricultural runoff
- Responses - e.g. percentage of land subject to environmentally benign tillage or fertilization practices.

In other words, indicators may be based on data about responses (actions) or about effects (results). The effects on the environment are the matter of real interest, but in cases where they are difficult to measure, indicators of response can be useful. If, for instance, a clear connection has been established between tillage practices and rates of erosion on a particular type of land, then measurements of response (tillage) will provide a fair estimate of the effect (erosion rate). The downside of response-based indicators is that they provide no incentive to develop new practices that might contribute to reducing environmental impacts.

Agriculture is characterized by a diversity of products and operations, and is carried out under varying climatic conditions and on varying types of land. In order to estimate impacts at the national level, it is necessary to develop models which can represent the whole agricultural sector in a satisfactory manner. This can be done by using data from a representative sample of "model farms", as has been done for example in Denmark (Dalgaard 2006).

Indicators may show the environmental impacts of products per unit of energy or of weight, or they may be related to units of land. Results may be presented at various levels – for a farm, a catchment, a municipality, a district, a county or the whole country. Transparency concerning what indicators cover is important. In life cycle analysis, for instance, system boundaries – which define what inputs and processes are included in a product's impacts – can influence the results substantially. Cederberg (2004) found that in conventional production of milk, purchased feed concentrates were responsible for most of the fossil energy consumed. If environmental impacts due to processes outside the farm are left out of the analysis, then operations based largely on purchased feedstuffs will appear to perform better than those which are not. Life cycle analyses (LCAs) aim in principle to cover all major impacts that products have throughout their life cycle, but may stop at some point, such as the farm or factory gate (Daugstad et al. 2009). Carbon footprinting and environmental footprinting are simpler methods for analysing climate or environmental impacts.

2 Current status of indicator development and environmental reporting

Over the past 20-30 years, the UN, the EU and its statistical agency Eurostat, the World Bank and the OECD have all conducted work on indicators for agriculture and environment (Bye et al. 2010). Eurostat provides the OECD with data on European countries. The EU Commission has adopted 28 Agricultural Environmental Indicators (AEI), but only six of these are clearly defined and supported by satisfactory data. (Bye et al. 2010). The indicators are listed in Section 4.

The Nordic Council of Ministers has commissioned work on indicators for sustainable development (Norsk institutt for jord- og skogkartlegging 2004).

Through a project entitled DireDate, Eurostat has explored ways of collecting the necessary data for the 28 AEI indicators (European Commission 2011). Three different proposals, involving differing degrees of accuracy and therefore demands on suppliers of data, were advanced.

Statistics Norway publishes and annual report on agriculture and the environment (*"Jordbruk og miljø – tilstand og utvikling"*), which describes the current situation and trends in environmental impacts. The following topics are covered (Bye et al. 2010):

- The structure of agriculture
- Land use (additions to and losses of farmland, drainage)
- Organic agriculture
- Biodiversity
- Agricultural landscape
- Fertilisers
- Pesticides
- Energy
- Nutrient emissions to waterways
- Atmospheric emissions
- Wastes and recycling.

This report does not focus mainly on indicators, but the section on pesticides does for example include indicators of human toxicity and eco-toxicity.

The Budget Committee for Agriculture (Budsjettnemnda for jordbruket) issues an annual publication *("Resultatkontrollen for gjennomføring av landbrukspolitikken")* which relates trends in agriculture to the goals set and the guidelines laid down in national environmental and agricultural policies. The chapter on protection of resources and the environment in this report is based on the Statistics Norway publication mentioned above. This chapter deals with issues such as the agricultural landscape, tillage practices and erosion, pollution, fertilisers, pesticides and recycling of plastic sheeting.

- Greenhouse gases
- Runoff

- Tillage, erosion and intensity
- Consumption of fertilisers and lime
- Pesticides
- Recycling of plastic sheeting.

The report presents statistics on the various subjects, but fewer indicators. In certain cases, for instance with reference to the 3Q programme (monitoring of the agricultural landscape and fulfilment of targets in that area) indicators are presented. These include indicators of trends in agricultural landscapes and in biodiversity (populations of birds and vascular plants on a countrywide sample of 1,400 plots of 1 m2 each) (Norsk institutt for jord- og skogkartlegging 2004).

As part of a national soil monitoring programme (JOVÅ), later replaced by *"Jord- og vannovervåking i landbruket"* (Soil and water monitoring in agriculture) (JOVA), the state and evolution of water quality in selected rivers has been continuously monitored. The implementation of the EU Water Framework Directive has led to a stronger focus on monitoring of water quality, and will continue to do so. Indicators have been developed that show the percentages of freshwater bodies and of coastal waters that are in good or very good ecological state.

Environmental indicators and reporting systems were among the topics dealt with in a project called *"Landbrukspolitikk fra 2002 – forenkling og effektivisering"* (Agricultural policy beyond 2002 - streamlined, targeted and more effective) (Kallbekken 2002). The project's acronym was FOLA. Indicators were proposed for nitrogen and phosphorous balances, use and toxic risks of pesticides, soil erosion, water quality, gross GHG emissions, biodiversity and landscapes. However, little use has been made of these indicators.

Norwegian farms are required to have an environment plan and to fulfil certain more specific requirements in order to be eligible for full production subsidies. The latter requirements include a plan for fertiliser applications, a diary of pesticide treatments and documentation of any habitats or features of particular environmental importance on the farm, including plans for their maintenance and proof of implementation. A so-called Level 2 environment plan may make the farmer eligible for extra subsidies in addition to the production subsidy. This requires targets for environmental improvements, a plan setting out measures to attain them and later proof of implementation. None of these requirements are directly linked to environmental indicators, but the requirement that farmers assess the need for fertiliser applications does imply certain requirements concerning nitrogen, phosphorous and potassium balances. The recommendations from the FOLA project on this point have thus been partly implemented, but only partly.

The agreement between the government and the Norwegian Farmers' Union following their annual negotiations in 2011[17] states that there is a need to develop indicators and reporting systems that are better able to measure and to demonstrate the effects of actions and policy measures. The contracting parties agreed that a commission should be appointed to propose a consistent reporting system for environmental goals, policy measures and indicators of goal fulfilment. (Landbruks- og matdepartementet og Norges Bondelag 2011). It will presumably be regarded as relevant to harmonise this work with international developments in the field of environmental indicators.

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[17] Norway has a unique system whereby the government and the two national farmers' organisations conduct annual negotiations on the level and structure of agricultural subsidies as well as other agricultural policy measures. (Formally, all expenses must be granted by Parliament, and Parliament decides the issue if the negotiations fail to produce an agreement.) In 2011 the Norwegian Farmers' and Smallholders' Union withdrew from the negotiations, but the Norwegian Farmers' Union (the larger organisation) reached an agreement with the Ministry of Agriculture and Food.

3 Which indicators should be included in environmental and climate analyses?

The choice of relevant indicators depends largely on the purpose of an analysis. The Environmental Strategy 2008-2015 of the Ministry of Agriculture and Food sets out the following goals for agriculture, forestry and food[18] (Ministry of Agriculture and Food 2008):

- Strong and long-term land protection to safeguard the most valuable farmland resources
- Maintain the farming landscape throughout the country by sustaining active agriculture
- Sustainable forestry as a basis for increased value creation and increased commitment to the use of bioenergy, timber and non-cultivated land resources
- Maintain a good plant and animal health status
- Maintain and develop high animal welfare standards
- Contribute to diversity of outdoor recreation experiences and activities
- A viable reindeer industry that sustainably uses grazing resources and contributes to maintaining the distinctive character of the Sami culture
- Maintain food security and sustainable agriculture through conservation and use of agriculture's genetic resources
- Avoid the introduction and limit the spread of invasive alien species
- Prevent GMO contamination of conventional and organic crops
- Manage the diversity of cultural monuments and environments as a basis for knowledge, recreation and value creation
- Contribute to securing a good ecological status of aquatic ecosystems
- Maintain Norway's high health and environmental standards related to the use of pesticides
- Contribute to reducing the amount of food waste and utilising valuable resources in organic waste materials
- 15 per cent of food production and consumption shall be organic by 2020
- Enhance and make visible forests' positive role as a climate mitigation measure
- Limit emissions to the atmosphere from the production, processing and consumption of food.

It would hardly be possible to carry out a coherent environmental and climate analysis of Norwegian agriculture or of the Norwegian food chain that included all of the topics covered by the Ministry's goals. Reducing air and water pollution will be central goals of environmental and agricultural policy in coming years. Agricultural impacts on biodiversity will also be a central issue. Biodiversity and agricultural landscapes are so closely interlinked that it would hardly be reasonable to include only one of the two in an analysis.

[18] The Ministry is also responsible for forestry. Its Norwegian name is *Landbruks- og matdepartementet*, "landbruk" being a Norwegian term which covers both agriculture and forestry.

The choice of indicators must also depend on how comprehensive an environmental and climate analysis one aims at. Possible directions and levels of ambition could be:

- Environmental accounts (all environmental impacts)
- Emissions accounts (impacts on air and water bodies)
- GHG accounts (carbon dioxide, methane, nitrous oxide, chlorinated gases)
- Facilitating life cycle analyses (LCAs) of individual products or production systems.

Environmental indicators should be relevant, comprehensible and reliable, the latter meaning that the metrics should have reasonably low margins of error. How stringently the reliability criterion should be applied must nevertheless be judged in the light of complexity, environmental risk and other issues. In the cases of many potentially relevant indicators, it is not practicable to base them on direct measurements. Estimates of emissions for instance may have to be based on modelling procedures, using input data on consumption of concentrates and other feedstuffs, artificial fertilisers, management and application of manure, or other factors.

Many indicators can be expressed in relation to units of land, or of production by weight, content of dietary energy or content of particular nutrients. By using such units as denominators, data on consumption of materials or energy can be converted to quantities of material or energy input per unit of land or of output (weight, dietary energy or nutrient content). This leads to indicators of efficiency. If data on inputs and outputs are commensurable (quantities of energy or of particular elements on both sides) then balances can also be calculated (e.g. carbon, nitrogen or phosphorous balances). Energy balances should distinguish on the input side between different energy carriers (e.g. fossil fuels, electricity and bioenergy).

Greenhouse gas emissions from food production can be expressed in terms of CO_2 equivalents per decare or hectare or per unit of dietary energy (kcal/kJ). Emissions are sometimes presented per kg of product, but since the content of dry matter, energy and protein varies widely between foods, this measure can be misleading.

Regarding emissions, the following indicators appear most relevant:

- Methane (to air)
- Nitrous oxide (to air)
- Carbon dioxide (to air)
- Nitrogen (to air as ammonia and to water)
- Phosphorous (to water)
- Pesticides (to water, air and soils).

Regarding other environmental impacts, indicators of the following may be relevant:

- Trends in biodiversity
- Trends in cultural landscapes.

Concerning the two last mentioned topics, further development of indicators could be based on the indicators and methods that have been applied in the 3Q programme.

4 The EU Agricultural Environmental Indicators

Vedlegg: EU sine miljøindikatorar for jordbruk

No	Indikator (norsk)	Indikator (engelsk)	Hovudansvar for nasjonal oppfølging	Hovudansvar for europeisk oppfølging	Nivå for utvikling per 2006
1	**Miljøforpliktingar i jordbruket**	Agri-Environmental commitments	**SLF/SSB**	DG AGRI	**B**
2	**Jordbruksareal under Natura 2000**	Agricultural areas under Natura 2000	**?**	EEA	**A**
3	**Bøndene sitt utdanningsnivå og bruk av miljøfagleg rådgjeving**	Use of environmental farm advisory services and farmers` training level	**SSB**	Eurostat	**A/B**
4	**Økologisk jordbruksareal**	Area under organic farming	**SSB/Debio**	Eurostat	**A**
5	**Bruk av mineralgjødsel**	Mineral fertiliser consumption	**SSB**	Eurostat	**B**
6	**Bruk av plantevernmiddel**	Consumption of pesticides	**SSB**	Eurostat	**C**
7	**Vatning av jordbruksareal**	Irrigation	**SSB**	Eurostat	**A**
8	**Bruk av energi**	Energy Use	**SSB**	Eurostat	**B**
9	**Endring i arealbruk**	Land use change	**SSB/SoL**	EEA	**B**
10.1	**Dyrkingsmønster**	Cropping patterns	**SSB**	Eurostat	**B**
10.2	**Husdyr**	Livestock patterns	**SSB**	Eurostat	**B**
11.1	**Jorddekke**	Soil cover	**SSB**	Eurostat	**B**
11.2	**Jordarbeidingspraksis**	Tillage practices	**SSB/SLF**	Eurostat	**B**
11.3	**Lagring av husdyrgjødsel**	Manure storage	**SSB**	Eurostat	**B**
12	**Intensivering/ekstensivering**	Intensification/extensification	**NILF/SSB**	DG AGRI	**A**
13	**Spesialisering**	Specialisation	**SSB**	Eurostat	**A**
14	**Risiko for at jordbruksareal går ut av drift**	Risk of land abandonment	**NILF/SSB**	DG AGRI	**C**
15	**Brutto nitrogenbalanse**	Gross nitrogen balance	**SSB/Bioforsk**	Eurostat	**B**
16	**Risiko for fosforureining**	Risk of pollution by phosphorus	**Bioforsk/SSB**	DG ENV	**B**
17	**Risiko ved bruk av plantevernmiddel**	Pesticide risk	**Mattilsynet/SSB**	DG ENV	**B**
18	**Utslepp av ammoniakk til luft**	Ammonia emissions	**SSB**	EEA	**B**
19	**Utslepp av klimagassar**	Greenhouse gas emissions	**SSB**	EEA	**A**
20	**Uttak av vatn**	Water abstraction	**SSB**	EEA	**C**
21	**Jorderosjon**	Soil erosion	**Bioforsk**	JRC	**B**
22	**Genetisk mangfald**	Genetic diversity	**Norsk genressurssenter**	EEA	**C**
23	**Jordbruksareal av høg naturverdi**	High nature value farmland	**SLF/DN**	DG AGRI	**C**
24	**Produksjon av fornybar energi**	Production of renewable energy	**SSB**	DG AGRI	**B**
25	**Fuglar knytte til jordbrukslandskapet**	Population trends of farmland birds	**SoL/DN**	EEA	**B**
26	**Jordkvalitet**	Soil quality	**SoL**	JRC	**C**
27.1	**Vasskvalitet - nitratureining**	Water quality – Nitrate pollution	**Bioforsk**	EEA	**B**
27.2	**Vasskvalitet – pesticidureining**	Water quality – Pesticide pollution	**Bioforsk**	EEA	**B**
28	**Landskap – status og mangfald**	Landscape – State and diversity	**SoL/SSB**	JRC	**C**

Nivå for utviklingsnivå:
a) indikatoren er klart definert og datagrunnlaget er rimeleg godt
b) indikatoren er veldefinert, men kan ikkje nyttast i forhold til sitt fulle informasjonspotensial pga. mangel på regional fordeling, eller at han vanskeleg kan samanliknast mellom land, eller pga. veikskap i modellen han er basert på
c) indikatoren treng betydelege forbetringar for å bli operasjonell

Forkortelser:
SSB = Statistisk sentralbyrå
SLF = Statens landbruksforvaltning
SoL = Norsk institutt for skog og landskap
NILF = Norsk institutt for landbruksøkonomisk forskning
DN = Direktoratet for naturforvaltning

DG AGRI = EU-kommisjonen sitt generaldirektorat for jordbruk
DG ENV = EU-kommisjonen sitt generaldirektorat for miljø
JRC = EU-kommisjonen sitt felles forskingssenter

Indicator status 2006:

A=The indicator is clearly defined and the data are reasonably well available
B= The indicator is well defined but can not be used in relation to its full information potential because of lack of regional distribution, or that it can hardly be compared between countries, or because of weaknesses in the model it is based on.
C= The indicator needs substantial improvements to be operational.

Abbreviations:

Responsible Norwegian agencies:
SSB: Statistics Norway
SLF: Norwegian Agricultural Authority
SoL: Norwegian Forest and Landscape Institute
NILF: Norwegian Agricultural Economics Research Institute
DN: Norwegian Directorate for Nature Management

Responsible European agencies

DG AGRI: EU Directorate-General for Agriculture
DG ENV: EU Directorate-General for Environment
JRC: Joint Research Centre of the EU Commission

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PART 5: Summary, Discussion, and Conclusion

1 Summary: The main findings

In this report we distinguish between environmental and in particular the climate impacts from agriculture and food. Focusing on agriculture implies to look at the agricultural sector's production of food, while focusing on food also implies to consider processing, distribution, and consumption of food. In part 1 we reviewed the carbon foot prints and other measures for environmental impacts caused by food production – based on results from other studies around the world. In part 2 we discuss the current state of knowledge regarding climate change impacts from food and agricultural production in Norway with particular emphasis on results from some recent studies. In part 3 the environmental impacts from agriculture was calculated based on input-output models and we discussed the potential of using such a model. In part 4 we have looked at different indicators for environmental impacts from agriculture and their potential also in policy. Neither part 3 nor part 4 considers the choices made by consumers. In part 2 an analysis by NILF of the climate gas emission from food and agriculture in Norway is presented which uses both a consumer and a production perspective. In part 4 we mainly dealt with indicators measured from an agricultural viewpoint at farm level. Due to limited resources we did not, in part 3, test the importance of uncertainties in the model (although this issue might partly be dealt with by a toolbox that uses sensitivity testing). In the literature survey in part 1 we found that various estimates of GHG emissions in the same geographical area may differ substantially. These differences may partly be blamed uncertainties in measurements of input data, but they may also be caused by the use of different assessment methodologies (model uncertainty).

Two previous Norwegian studies (Hille et al 2008; Hertwich and Peters 2009) indicate that food consumption in Norway produces GHG emissions of approximately 2.5 t/person/year. This is within the range of results from studies in other Western countries, and assumes that food contributes to at least one sixth, or maybe more, of the total emissions from consumption. Studies referred to in part 2 and part 3 present results that are in agreement with this. Refsgaard et al. (2011) consider a few important food products that account for about 50% of food consumption on a weight and energy basis, and show that these products are responsible for emissions of 5.7 Mt/year, or 1.2 t/person. In part 3 is presented a basket of food where the results show that nationally produced items contribute to emissions of approximately 7.8 Mt/year, or 1.6 t/person. Emissions also originate from imported foods. We can thus state that food consumption is a significant contributor to the GHG emissions in Norway.

Food production is also a dominant contributor to land use and to the consumption of energy and material resources. Further food production is also a source for acidification and other environmental impacts such as eutrophication. More recent Norwegian studies, presented in part 2 and part 3, only consider GHG and land use.

Different types of food provide very different GHG emissions, whether measured per kilogram or per dietary energy content. Measured per kg all international studies show that meat from ruminants are among the largest contributors to GHG emissions of the food items. Meat from chicken and pigs, eggs, and milk have lower emissions than meat from ruminants. In general, most plant food provide much lower emissions than animal food, but vegetables from heated greenhouse and/or highly processed food provide equal or greater emissions per kilogram than milk. Locally produced cereals, potatoes, and fresh vegetables in general give lower emissions. Looking at emissions per dietary energy content, vegetable food *usually* show lower emissions than animal food, but the difference decreases, e.g. between milk and meat from ruminants or between animal food and vegetables and fruit when measured per dietary Kcal instead of per kg because of the higher content of energy in meat compared to milk and vegetables. Using dietary energy as measure, implies that the most energy-dense vegetable products (grain, sugar, and vegetable oils) have lower GHG emissions than for instance fruit and vegetables.

The recent Norwegian studies presented in Parts 2 and 3 show that meat from ruminants give the largest emissions per kilogram in this country. Both studies presented show that milk provides far fewer emissions, and potatoes and grain products contribute an even smaller amount. The study in Part 3 also shows that chicken and pork, followed by eggs, are between beef and milk in the ranking from largest to smallest GHG emissions per kilogram.

GHG emissions from food occur in agricultural production or in the manufacturing of inputs to agriculture, such as fertilizers. They can also occur further downstream through the processing, distribution, and trading of goods. International studies show that emissions from agricultural production and the manufacturing of intermediate goods to agriculture are clearly the largest contributing source when animal products are manufactured. At least 80-90% of the emissions from these products takes place before the products leave the farm. For plant products, however, processing, distribution, and trade are responsible for a much larger relative contribution to total emission, in some cases above 50%. The model developed in Part 3 ignore emissions originating from processing, distribution, and trade with food. Refsgaard et al. (2011) analysed these factors in the context of potatoes, bread, milk, and beef, and showed that emissions from both manufacturing and trade may be very small in Norway for these food products, which is probably due to the climate-friendly Norwegian energy mix.

International studies indicate that organic production often provide lower emission than conventional production when it comes to cereals. However, for potatoes, vegetables, fruits, and animal products, the literature shows different results. Some studies show that conventional goods show lower emissions while others indicate that organic products perform better. The study by Refsgaard et al. (2011) shows that organic production performs better in Norway for bread, milk, and beef for specific types of production systems. Several factors like feeding strategy, cropping system and types of land used may effect these results significantly. Both the study by Refsgaard et al. (2011) and other European studies show that organic production requires more land area than conventional production. The study from Norway further shows that the production of beef requires far more land area per dietary energy content than the production of milk, which in turn requires more than grain or potatoes.

2 Discussion: Possible actions to reduce GHG emissions?

There is political pressure on actors in all sectors of the economy to find ways to reduce GHG emissions. This also applies to those involved in food production. If we aim to realise as much of the potential for reducing the carbon footprint from food consumption as possible, an understanding of how different foods contribute to emissions is clearly useful, as is an understanding of the contributions from different stages in the food life cycle and the interactions within the chains. We have in this report tried to compile recent and precent literature and analyses internationally and nationally showing impacts mainly in the form of GHGemissions from food and agriculture. Based on this we try to point out a number of possible strategies for reducing GHG emissions.

Production-oriented strategies

In a production-oriented strategy, there are different ways to reduce GHG emissions from a sectorial level. There are opportunities to take action in different parts of the production system at the farm level. In their white papers, Klif (2010) and LMD (2009) discuss GHG reducing actions at the farm level, such as biogas, biocoal, and the management of animal manure. However, as shown in Part 2 (Chapter 2), it is important to consider both the upstream and downstream sectors when attempting to find strategies to reduce GHG emissions from food production, which is also emphasised by LMD in their white paper. It is also important to choose the better ranked strategies first. We can summarize the possible actions in the following way:

1. The relationship between product volume and climate gasses within the agriculture sector:
    a. Number of lactations per cow
    b. Changed feeding regime
    c. Meat produced in combination with milk
    d. Import versus domestic production
    e. The intensity of capital and other inputs
2. Change in the production system (considering land use changes)
3. Here an important issue is the use of land and the relationships with carbon fixation in soil varying between different types of soil
4. Technological changes outside the agricultural sector
    a. Agriculture/fish farming
    b. Manufacturing of inputs
    c. Processing, distribution, and trade.

Consumer-oriented strategies

Alternatively, one can focus on the issue from a consumer perspective and make a comparison of how different foods and food mixes can be prepared in meals. Thus, the analysis will show how changes in diet and food choices affect GHG emissions. For instance, we have the following possibilities:

1. changing the composition of foods (red versus white meat, plant versus animal consumption, reducing meat consumption);

2. reducing the disposal of food (at retail stores, in the home, increasing the use of raw materials);
3. examining the importance of different transport types, and local versus global choice; and
4. choosing particular foods from specific systems (production, transport, processing, or distribution), such as organic versus conventional, fair trade, local versus imported, and so on.

The land issue

When considering possible courses of action to reduce GHG emissions, it is also necessary to consider the possible consequences of these actions. A fair comparison of alternative agricultural systems must include an assessment of how much land will be required to produce a given amount of food. Land can substitute for fossil energy and organic agriculture (at least under European conditions) and tends to make up for the avoidance of artificial fertilisers and pesticides by requiring more land per unit product as compared with conventional agriculture. A preliminary estimate on the consequences of converting to 100% organic production of bread, milk, and beef in Norway would, according to the estimates of Refsgaard et al. (2011), reduce emissions by some 0.9 Mt CO_2-eq, but this is before the change in land use is considered. Such a transition would also require 2.1 million daa of agricultural land, land, including both grassland and arable land. Assuming only domestic production, the necessary composition would be an increase of 2.1 million daa of domestic cropland, in addition to 1.6 million daa increase in domestic grassland to replace 1.6 million daa decrease in 'imported' land. About 12.2 million daa of land is available in Norway, but because of the change in the composition of cropland, the change in the carbon sequestration of soil must be considered. A large amount of the potential land is currently either moor or forested area, and is primarily located in the middle and northern parts of Norway. Therefore, further analyses and calculations of whether total emissions would decrease or increase as a result of a total change to the organic agricultural system could be conducted.

Consumption patterns

The analyses by Refsgaard et al. (2011) show that consumers and the agricultural sector can contribute to reducing the carbon footprint from food consumption and production. Many recent analyses and proposals for climate strategies have focused on measures within the agricultural sector, including the improved management of manure, optimization of fertiliser applications, better drainage, production of biogas, and mixing of biochar into soils. However, both downstream and upstream processes within the food chain — including processing and distribution — need to be further explored in view of identifying the GHG reduction potential. An alternative (or complementary) approach is to start from the consumer's end of the food chain and ask how changes in diet, represented by alternative «baskets» of food, may contribute to reducing emissions and changes in land use.

Changes in consumption patterns could allow GHG emissions from the food chain to be reduced without any increase in the demand for farmland. Specifically, two changes could make a major difference: lowering the amount of food waste and increasing the share of plant versus animal foods in our diet. At present, some 30% of food ends up as waste in Norway, of which at least 50% is edible and estimated that emissions could be reduced by some 0.9 Mt CO_2-eq if this waste were eliminated in the case of the four products analysed (assuming conventional production). (Refsgaard et al., 2011). As illustrated in Chapter 2, plant foods *not only* tend to generate much smaller GHG emissions per unit of dietary energy than animal foods, but also require much less land.

The importance of local and global transport

Transport within Norway contributes to a much higher degree to GHG emissions than transport to Norway. This is likely to apply to many countries because primary production generates major GHG emissions that are mostly unrelated to the use of electric energy or fossil fuels, whereas downstream processes usually do not. Further trailer transport on land consumes much more energy and therefore contributes to a much higher degree of GHG emissions (measured per tonnes-km) as compared with transport by ship or train. This creates a challenge in analyses dealing with local versus global food. Empirical analyses of fuel consumption and emissions from food distribution are lacking in Norway. This is partly due to the fact that road transport is the dominant mode for most foodstuffs, although ship and rail transport are much more energy efficient.

3 Conclusion: Future need for studies

In general, there is a large gap in the literature regarding the impact of the Norwegian food sector on the environment. Part 1 points out that there were few LCA studies on food in Norway before the studies described in Parts 2 and 3 were conducted. We found only a few earlier studies including a study on Norwegian seafood by SINTEF (Winther et al. 2009) and a study on consumer milk (Høgås Eide 2002), in addition to individual studies on greenhouse production. However, many LCA studies are currently being carried out.

When it comes to more holistic studies, or those focused on total food consumption or larger portions of food production, a few studies examining a limited set of issues were found:

- Hertwich and Peters (2009) provide an estimate for GHG emissions from total food consumption (emissions cannot be broken down to individual food groups or processes). They do not consider other environmental influences.
- Hille et al. (2008) provide estimates on energy and land use related to total food consumption, but only the estimates for areas in use can be broken down into the various food groups. Hille and Germiso (2011) only consider land use.
- Before these studies, one must go all the way back to Breirem et al. (1980) to find a study addressing energy use and land use in regards to foods produced in Norway. On the other hand, Breirem et al. (1980) said nothing about the effects of imported food or seafood on the energy use downstream from the farm gate or about emissions.

The studies presented in part 2 and part 3 of this report cover a limited number of food products. Using four foods, Refsgaard et al. (2011) provide figures for GHG emissions and land area. The study in part 3 included more foods, but only up to the farm gate, and only figures for GHG emissions are provided. Vegetables and imported food are not included.

However the study by Refsgaard et al. (2011) uses basic economic farm models for the basic input and output for farm production, Referansebruk. As there exist a wide number of Referansebruk representing Norwegian agricultural production for different production systems, regions and sizes these can be combined with the framework and design for calculating GHG emissions. Such a development could provide future estimates for calculations of emissions from different production systems and/or types of food. In addition to such statistics it could provide more knowledge than what is currently available about the effects upstream and downstream in the agriculture chain. The results from the present study show that there is a potential for reducing GHG emissions from food and agriculture as pointed out above

Specific needs that are pointed out in Parts 2-4

One cannot neglect the fact that the global environmental effects of changes in the Norwegian agriculture and food sector cannot be accounted for before we investigate the effects of changes in the composition of meals as well as the effect of the sum of changes in Norwegian agriculture and diets on food and input factor imports. In Part 2, we call for more studies based on a consumer perspective, including studies that focus on imported foods and what happens downstream of the farm gate or import port, especially in terms of distribution (transport). We also call for more focus on the relationships between use of land, feeding strategy for ruminants and the associated carbon fixation.

The analysis in Part 3 is a sample analysis meant to test suggested practices, and partly to assess the usefulness of existing data sources for use in a model. To make the model reliable the following issues must be considered:

- better resolution of the actual agricultural sector in order to get better estimates of the impact of cross-deliveries (particularly regarding fodder);
- better integration of processes both upstream and downstream;
- better geographical resolution for the production of imported intermediate goods in order to get more realistic emission estimates rather than assuming that everything is produced with Norwegian (or German) technology; and
- ongoing research on the development of national/international models, such as HOLOS, CAPRI, or Jordmod, to obtain more precise estimates of the direct emissions from agriculture
- sensitivity testing.

As stated in Part 3, emissions of GHG are only one type of environmental impacts that can be handled with the model tool employed by MiSA. This is related to Part 4, where we discuss the need for further development of indicators for impacts from agricultural production on the environment.

Benefits of this report

The environmental assessment of food and the agricultural sector is difficult. Moreover, the methods used, and the indicators applied, will be of great importance for the result. In this report we have established a base material for further studies and projects being increasingly demanded by public bodies, the agricultural organisations, NGOs on environmental issues and the food sector.

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