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The environmental performance of Swedish food production
– An analysis of agri-environmental indicators

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The environmental performance of Swedish food production – An analysis of agri-environmental indicators

Miljöprestanda för svensk livsmedelsproduktion
– En analys av miljöindikatorer för jordbruket

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Abstract

In the European Union 43.5% of the land area is used for agricultural purposes, providing food, feed and fiber for the population, but also causing detrimental environmental impact. Agri-environmental indicators (AEIs) are used to measure, and communicate environmental performance of agriculture, and serve as important tools to develop and evaluate progress of agri-environmental policy and measures.

In this paper, commonly used AEIs for describing some of the biggest environmental issues coupled to agriculture, are analyzed, and applied to Sweden, to describe the present environmental performance of Swedish food production. An international comparison, including countries with similar climatic and agricultural conditions to Sweden, is used as support for discussing the difference in performance and the underlying causes. Novel indicators are also presented: *Nutrient leaching per kg protein*, and *Ammonia emission per kg milk/meat*, to visualize the link between food production and environmental impact. The paper is mainly based on a literature review, and the analysis and processing of data from Eurostat, OECD, FAO and EEA.

In comparison to the countries presented in this paper, Sweden has a high environmental performance in *Greenhouse gas emissions (GHG)*, *Gross nutrient balance*, *Nutrient leaching*, *Ammonia emissions (NH_3)* and *Pesticide use*, when expressed in kg/ha. The performance for the indicators *Nitrate pollution of groundwater* and *Agricultural water use* was also high (above average). The performance was low (below average) when NH_3 emission was expressed in mass units (kg/ton), for beef meat. The performance was average for the indicator *Soil erosion*. The performance was also average when *Nutrient leaching, GHG and NH_3 emissions* (pig, poultry and milk) were expressed in mass units (kg/kg). The overall high performance of Swedish food production is mainly explained by low nutrient inputs and low livestock densities in Sweden, compared to the other countries in this study.

Driving force and pressure indicators do not always manage to predict environmental performance, i.e. a high use of mineral fertilizers or a high gross nutrient balance does not necessarily mean that nutrient leaching is correspondently high. Natural conditions (soil type and precipitation) are likely a part of the explanation. The functional unit had great impact on the results, when the functional unit was mass based (kg), instead of land based (ha), countries with intensive production (high Animal- or Plant-Protein production per ha) were favored. When using mass based indicators (kg), the total environmental impact might still be high, even if the indicator result indicates low impact, thus mass based indicators should not be used alone if the aim is to describe total environmental impact.

Keywords: Nutrient leaching, nitrogen, phosphorus, greenhouse gases, ammonia, pesticides

Sammanfattning

Av den totala landytan i EU används 43.5% för jordbruksändamål, och förser samhället med livsmedel, foder och fiber, men bidrar också till negativ miljöpåverkan. Miljöindikatorer för jordbruket (AEIs) används för att mäta och kommunicera miljöprestanda inom jordbruket och tjänar som viktiga verktyg för att utveckla och utvärdera framsteg inom miljöpolitiken och åtgärdsarbetet som följer av denna.

I denna uppsats, analyseras och appliceras ett urval av de miljöindikatorer som vanligen används för att beskriva några vanliga jordbruksrelaterade miljöproblem, för att beskriva nuvarande miljöprestanda för svensk livsmedelsproduktion. En internationell jämförelse, som inkluderar länder med liknande klimat- och jordbruksförhållanden som Sverige, används som underlag för att diskutera skillnaden i miljöprestanda och de bakomliggande orsakerna. Nya indikatorer presenteras även: *Läckage av näringsämnen per kg protein* och *Ammoniakutsläpp per kg mjölk/kött*, för att synliggöra sambandet mellan livsmedelsproduktion och miljöpåverkan. Uppsatsen är huvudsakligen baserad på en litteraturstudie, samt analys och bearbetning av data från Eurostat, OECD, FAO och EEA.

Jämförelsevis, med länderna i denna uppsats, har Sverige en hög miljöprestanda för indikatorerna *Växthusgasutsläpp*, *Brutto näringsbalans*, *Näringsläckage*, *Ammoniakutsläpp* (NH_3) och *Användning av bekämpningsmedel*, uttryckt i kg/ha. Miljöprestandan för *Jordbrukets vattenanvändning* och *Nitratförorening av grundvatten* var även den hög. För indikatorn *Jorderosion*, var prestandan genomsnittlig. Prestandan var också genomsnittlig när näringsläckage, utsläpp av växthusgaser och ammoniakemissioner (för gris, fjäderfä och mjölk) uttrycktes i massenheter (kg/kg). Miljöprestandan var låg för indikatorn *Ammoniakutsläpp* (för nötkött), när denna uttrycktes i massenheter (kg/ton).

Så kallade ”Driving force” och ”Pressure” indikatorer misslyckas i vissa fall att förutsäga miljöprestanda, det vill säga en hög användning av mineralgödselmedel eller en hög brutto näringsbalans betyder inte nödvändigtvis att utlakning av näringsämnen är motsvarande hög. Naturgivna förhållanden (jordart och nederbörd) är sannolikt en del av förklaringen. Val av funktionell enhet inverkar på resultaten. En massbaserad (kg), istället för landbaserad (ha) funktionell enhet, är till fördel för länder med intensiv produktion (hög näringstillförsel och hög djurtäthet). Ett lågt beräknat värde för en massbaserad indikator (kg) kan dölja en stor miljöpåverkan, varför massbaserade indikatorer inte ska användas ensamma om syftet är att beskriva den totala miljöpåverkan.

Nyckelord: Näringsläckage, kväve, fosfor, växthusgaser, ammoniak, växtskyddsmedel

Preface

This master thesis was written during spring 2018 at the Swedish University of Agricultural Sciences (SLU), at the department of Soil and Environment, as an independent project in soil science, within the agriculture programme plant/soil, on advanced level (30 hp). The subject of the thesis was initiated by the Swedish Farmers Association (LRF), and written in a collaboration between SLU and LRF. The work was supervised by Ass. prof. Helena Aronsson at SLU and dr. Markus Hoffman at LRF. The thesis was examinated by professor Lars Bergström, at SLU.

The thesis analyzes the present environmental performance of Swedish food production, by analyzing and presenting the results of currently available agri-environmental indicators. This is an important and relevant subject in times with a rapidly growing population, and changeable climate, altering the conditions for modern food production. Not the least is it important to analyze the tools we use to evaluate our food production systems, i.e. agri-environmental indicators, which serve as support for decision makers and consequently for developing agri-environmental policy. Hopefully this thesis will contribute to a deepened knowledge of the environmental impact of Swedish food production, while simultaneously shedding light on advantages and disadvantages of some selected agri-environmental indicators.

Danira Behaderovic
Uppsala, May 2018

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Abbreviations

AEI	Agri-Environmental Indicators
CH_4	Methane
CLRTAP	Convention on Long-Range Transport of Air Pollutants
CO_2	Carbon oxide
Denitrification	Microbial process referring to the reduction of NO_3 to N_2
EEA	European Environment Agency
EF	Emission Factor
GHG	Greenhouse gases
GWP	Global warming potential (reference value is one CO_2-molecule)
Ha	Hectare, an area of 10 000m^2
IPCC	Intergovernmental Panel on Climate Change
LSU	Livestock units
N	Nitrogen
N_2O	Nitrous oxide
NH_3	Ammonia
Nitrification	Microbial process referring to the oxidation of NH_4 to NO_3
NVZ	Nitrate Vulnerable Zones
OECD	Organization of Economic Co-oporation and Development
P	Phosphorus
SBA	Swedish Board of Agriculture
SEPA	Swedish Environmental Protection Agency
SOC	Soil Organic Content
UAA	Utilized Agricultural Area
UNECE	United Nations Economic Commission for Europe
UNFCCC	United Nations Framework Convention on Climate Change

1 Introduction

Agriculture provides food, fiber and fuel to a growing population, expected to reach 9.8 billions by 2050, accompanied by an estimated 100-110% increase in global crop demand between 2005 and 2050 (Spiertz & Ewert, 2009; Tilman *et al.*, 2011; United Nations, 2017). The use of 43.5% of the land area in the European Union for agricultural purposes, including arable land, permanent crops and permanent pasture (World Bank, 2018), has a significant environmental impact. Application of fertilizers and pesticides, tillage and livestock farming are some of the agricultural practices coupled to negative environmental impacts such as emissions of greenhouse gases (GHG), leaching of nutrients causing eutrophication, dispersion of pesticides to ground and surface waters and soil degradation (Skinner *et al.*, 1997). Simultaneously, agriculture can contribute to positive environmental impact. Agricultural soils can act as carbon sinks (Poeplau *et al.*, 2015), agricultural landscapes can play an important role in enhancing biodiversity and preserving cultural landscapes such as some of the ecologically important grasslands maintained by grazing cattle (Watkinson & Ormerod, 2001; SCB *et al.*, 2012). Further, agriculture provides biocrops for a society in transition from fossil fuels to renewable energy and has an important role in nutrient cycling from urban to rural areas (SCB *et al.*, 2012).

Increased environmental concern and a growing demand for food has led to a call for sustainable agriculture which can provide food for a growing population by maintaining high yields while concurrently having an acceptable environmental impact (Tilman *et al.*, 2002). Sustainable development is defined in the Brundtland Report as follows:

> "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (United Nations, 1987)

The increased importance of measuring sustainability of agriculture and the results of policy measures, has led to the development of agri-environmental indicators

(AEIs). AEIs are used to evaluate progress of agricultural policy measures (Parris, 1997), and could be considered as tools used to describe a complex reality by simplification; "*Indicator is a variable which supplies information on other variables which are difficult to access*" (Gras et al., 1989, see Bockstaller *et al.*, 2008, p 139). OECD was among the first organizations to develop AEIs, and today there is an array of AEIs which can be used for measuring and comparing environmental performance of agriculture (FAO, 2016a; EEA, 2018a; Eurostat, 2018g; OECD, 2018). AEIs should be simple to use and obtain data for, while giving high quality of prediction. (Bockstaller *et al.*, 2008).

1.1 Aim and scope

The main research aim is to *describe the present performance of Swedish food production,* by using and analysing agri-environmental indicators. As it is not always apparent that AEIs relate to the environmental impact of food production, one aim of this thesis is to develop new AEIs, where environmental performance of agriculture is directly linked to food production. The thesis is divided into three main parts:

1. Produce an overview of various measures used for calculating environmental impact of primary food production by identification and compilation of current agri-environmental indicators used for a limited number of environmental issues.
2. Deepen the knowledge of environmental impact of food production by describing the methodology, scale of use and input data of a few selected agri-environmental indicators.
3. Describe the performance of Swedish food production by making an international comparison for the selected agri-environmental indicators, as well as for new AEIs where agricultural environmental performance is directly linked to food production.

The system boundary of this study is the farm level. Hence the environmental impact of food production after leaving the farm gate, such as processing and transport will not be regarded, simply as this is beyond the scope of the AEIs. Environmental factors impacted by food production covered in this thesis are those with a link to soil processes, such as geophysical fluxes and nutrient cycles including:

- Emission of greenhouse gases (GHG)
- Emission of ammonia (NH_3)
- Eutrophication and nutrient imbalances
- Water – quality and usage
- Soil quality and soil erosion

Biodiversity and energy use will not be regarded in the study. In the comparing part of the thesis, countries for comparison are chosen based on similarities in climatic and agricultural conditions to Sweden. Scandinavian neighbouring countries Finland, Denmark and Norway will be included as well as Germany, the Netherlands, United Kingdom and Ireland.

1.2 Disposition and method

The thesis consists of four main parts:

1. A background including a review of global and regional environmental impacts by food production, relevant environmental policy and a general description of agri-environmental indicators (chap. 2).
2. A review of common AEIs – in this part selected indicators are described by analysing what environmental issues they relate to, what scale they are appropriate for, and what input data they consist of (chap. 3).
3. Presentation of the results of the reviewed AEIs and new AEIs, by an international comparison, with a simultaneous discussion of the results (chap. 4).
4. Discussion and conclusions (chap. 5).

Chapter 2 and 3 of the thesis have been conducted through a literature review mainly consisting of scientific journals and reports from authorities and organizations. International organizations have been the source of agri-environmental indicators, mainly Eurostat, Organization for Economic Co-operation and Development (OECD), European Environment Agency (EEA) and United Nations Food and Agricultural Organization (FAO). A list of the AEIs which were encountered during this thesis is found in Appendix 2. The selection of AEIs included in the review part of the thesis was based on AEI relevancy for the environmental issues which were the focus of this thesis, and the result of discussions with supervisors, as well as data availability of the AEIs. Chapter 4 mainly consists of officially available data found at the mentioned organization's databases (production figures, land use data and AEI values). These figures were also used in the experimental part, when calculating a new AEI "Nutrient leaching per kg animal and plant protein (g N or P/ kg plant or animal protein)". See Appendix 1 for input data and demonstration of calculations. Personal communication with research institutes and authorities were used as sources where data was lacking for the countries included in the international comparison.

2 Background

2.1 Food demand and future projections

Global food demand has continuously been increasing for the last 50 years, in correspondence to a growth in gross domestic product (GDP) and household income increase. The global food demand is predicted to increase further by 2050, under several projected scenarios. In year 2000, the average global food demand was estimated to 2736 kcal/capita/day. By 2050 this figure might increase to 3177 kcal/capita/day (in a scenario with sustainable economy and high environmental awareness), and the share of animal calories might reach 1235 kcal/capita/day for OECD countries (Bodirsky *et al.*, 2015). Average global human food intake, measured in energy consumption, dominates by cereal products (53%), followed by non-processed vegetables (16%), sweeteners (8.2%) vegetable oils (6%) and meat and dairy products (13.3%). Still, 68% of all energy obtained in biomass is appropriated for producing meat and dairy, as livestock requires large amounts of feed, as energy is lost as heat and excrete in the process of converting animal feed to milk and meat (Wirsenius, 2000).

Food intake is closely related to land use, as land is used to produce biomass used as either animal feed or plants used for human consumption. 98% of global food supply comes from terrestrial ecosystems (Duarte *et al.*, 2009) and globally 37% of terrestrial land is used for food production (World Bank, 2018). Most of this land (70%) is used as permanent grasslands, and 30% is used as cropland (Wirsenius, 2000). Some of the cropland is used for producing non-food products such as cotton, fibre and biofuels, however, land used for food and feed production dominates (Wirsenius, 2000).

2.2 Negative environmental impact driven by food demand

The environmental impacts coupled to food demand and agricultural activities can be divided into regional and global impacts. Emission of greenhouse gases (GHG) can be considered global, while many impacts are regional, such as pollution of waterbodies, nutrient leaching and eutrophication. In this chapter, the environmental issues coupled to agricultural practices are only mentioned briefly. Detailed reviews are found in chapter 3.

2.2.1 Global environmental impact

Agriculture is a major source of non-CO_2 greenhouse gases (GHGs) and contributes to 10-12% of global anthropogenic GHG emissions, driving global warming (Smith *et al.*, 2007). The implications of global warming such as warming of oceans and atmosphere are world-wide, altering human and natural systems and decreasing food safety (IPCC, 2014). Global GHG emissions from agriculture are estimated to 5.0-6.1 Gt CO_2 –eq/year, consisting of nitrous oxide (N_2O) and methane (CH_4). CO_2 makes up only a small share of the total emissions. CH_4 is 28 times stronger GHG than CO_2, and N_2O is 265 times stronger, measured as global warming potential (GWP) over a 100-year period (Myhre *et al.,* 2013). The main source of CH_4 is livestock, and the enteric fermentation by ruminants. N_2O is mainly emitted from soils where nitrogen containing compounds i.e. mineral fertilizer, manure, green manure and crop residues are converted to N_2O by microbial processes. CO_2 is mainly emitted from soils, when organic matter is decomposed. However, agriculture is the sector with the highest potential for cost effectively mitigating climate change, through improved management practices (Smith *et al.*, 2007).

2.2.2 Regional environmental impact

Most of the environmental impacts coupled to food production at the farm scale, have a regional environmental impact. Dispersion of pesticides to water bodies and soil might impact aquatic and terrestrial ecosystems, and pollute drinking water. There is high variance between the risks of pesticides, depending on the characteristics of the pesticide and agricultural management practices (Skinner *et al.*, 1997).

Leaching and runoff of nitrogen (N) and phosphorus (P), from agricultural soils might cause eutrophication in waterbodies and contaminate drinking water with nitrates, with possibly detrimental effects on human health (Skinner *et al.*, 1997).

Agriculture is the sector contributing to the largest emissions of ammonia (NH_3), mainly derived from livestock excreta. NH_3 contributes to acidification of aquatic and terrestrial ecosystems, as well as eutrophication, reducing the quality and productivity of these systems. NH_3 can be deposited close to its emitting source but also transported several hundred kilometres before deposition (Skinner *et al.*, 1997).

On soils prone to erosion, agricultural practices, such as tillage and bare soil between crops, can augment soil erosion, contributing to land degradation and dispersion of nutrients attached to soil particles, enhancing eutrophication (Pimentel, 2006).

The agricultural sector is a major user of water, primarily used for irrigation of intensive crops and drinking for livestock. With climate change, altered temperatures and altered hydrological patterns the concern for sustainable water use is increasing. In countries in southern Europe, experiencing water scarcity during some periods of the year, agriculture can account for 80% of total water use (EEA, 2012).

2.3 Global and EU environmental policy

Several policies and international treaties at global, European and national level concern sustainable agricultural practices. At EU level directives force into law agricultural practices with environmental consideration of which the most relevant are described briefly in this chapter. On national level, Sweden has developed a set of 16 environmental objectives based on the United Nations Sustainable Development Goals (SDGs), supposed to be met by 2020. Several goals are strongly linked to agricultural practices; *Zero Eutrophication, A non-toxic Environment, Natural Acidification Only and Reduced Climate Impact* (SEPA, 2017b).

2.3.1 Global treaties

- *The Paris Agreement*, which entered force in 2016, is a part of the United Nations Framework Convention on Climate Change (UNFCCC) and regulates GHG emissions. It binds all parties to make Nationally Determined Contributions (NDC) on how to reduce national GHG emissions to keep the rise of global mean temperature well below 2°C, compared to pre-industrial levels, during this century. Parties are required to continuously report GHG emissions and GHG reducing efforts.
- *The Convention on Long-range Transboundary Air Pollution (CLRTAP)* under the United Nations Economic Commission for Europe (UNECE), is an international treaty to combat air pollution, including regulation of NH_3 emissions. The treaty went into force in 1979 and obligates its parties to report on air pollutants, including NH_3. The Gothenburg Protocol (1999) added national emission ceilings, which the parties must meet by 2020 (UNECE, n.d.).

2.3.2 EU directives

- *The Nitrates Directive (91/676/EEC)* protects ground and surface waters from nutrients originating from agricultural sources. The directive requires the designation of Nitrate Vulnerable Zones (NVZ), for land where risk of nutrient loss is high. The directive requires Member States to legislate and regulate application and storage of fertilizers (organic and mineral) on NVZ, which will minimize the nutrient loss from agricultural land (SoCo Project Team, 2009; Aronsson & Johnsson, 2017).
- *The Water Framework Directive (2000/60/EC)* regulates sustainable use of water resources and the maintaining of good ecological status of water bodies. The directive indirectly affects pesticide pollution and nutrient loss from agricultural land (SoCo Project Team, 2009).
- *Sustainable Use of Pesticides Directive (2009/128/EC)* regulates pesticide use, with a main cornerstone being the implementation of Integrated Pest Management (IPM). IPM promotes the use of preventive methods (crop rotations, cultivation techniques etc.) and follow up, to reduce the reliance of chemical plant protection agents, and protection of sensitive areas, such as watercourses (European Commission, 2018)
- *Good Agricultural and Environmental Condition (GAEC*) is a part of the cross-compliance rules, and directly supports soil protection measurements by requiring e.g. minimum land cover. Under the EU Common Agricultural Policy, cross compliance rules are included, with the purpose to support sustainable agricultural practices, which farmers must obey to receive direct payments.
- Currently, no political agreement has been reached on a Soil Framework Directive proposed by EU in 2006. Thus, there are no directives directly concerning soil degradation and erosion (SoCo Project Team, 2009)

2.4 Agri-environmental indicators

An indicator in environmental context, consists of aggregated data, which can be easily interpreted and used to communicate environmental trends. Conversely data alone, which is the foundation of indicators, cannot be directly used to understand driving pressures, change of environmental status or impact. When two or more indicators are aggregated, it is called an index. An index is easily communicated, but can become too weak for being the base of decision making (Segnestam & Persson, 2002), due to the loss of information (Bockstaller et. al, 2008). Generally, it can be expressed that the communicability increases with increased aggregation of data, while accuracy and usefulness for decision making decreases (Segnestam & Persson, 2002).

2.4.1 Development and criteria of AEI

The development of Agri-environmental indicators (AEIs) in Europe began with the establishment of the European Union in 1993, which demanded environmental concern when implementing EU policies (Eurostat, 2018h). AEIs where thus developed to give information to support policy making and progress under the Common Agricultural Policy, which addresses sustainable management of natural resources. AEIs give information on the state and trends of the environment under agricultural practices and in extension information on the environmental impact of food production. They consist of aggregated data, and can be used to give information on variables which are hard to measure, especially on national scale. There are many different definitions of indicators, Heink & Kowarik (2010) suggests the following definition;

> "An indicator in ecology and environmental planning is a component or a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals. Environmentally relevant phenomena are pressures, states, and responses as defined by the OECD (2003)." (Heink & Kowarik, 2010)

OECD was among the first trans-national organizations to develop AEIs, built on questionnaires filled by OECD countries. This resulted in variance in data, as countries tend to not gather data for issues not relevant for the countries, i.e. a country which does not use irrigation will not have this type of data. The identification of agri-environmental issues is based on issues considered to be under professional, political and public debate (Wascher, 2002). Input data for agri-environmental indicators used at national scale usually consist of data on farmer practices obtained by censuses, farm structure surveys and questionnaires (OECD, 2001; Eurostat, 2016a). This can be explained by the difficulty of obtaining measured data or data estimated by models, as the first can be expensive and the second requires both model development and gathering of input variables. Preferably both model based and measured data should be used when evaluating cropping systems, as it can be difficult to measure all variables of interest (Bockstaller *et al.*, 2008).

To be useful, AEIs should fulfil the following criteria (OECD, 2001):

- Policy-relevant – the indicators should address environmental issues of relevance for policy makers and stakeholders
- Analytically sound – they should build on scientific knowledge
- Measurable – the indicators should be cost-effective in terms of collecting or measuring data
- Easy to interpret – the indicator should be easy to communicate to policymakers as well as to the general public (OECD, 2001 p 22)

2.4.2 The DPSIR framework

AEIs are classed as Driving force, Pressure, State, Impact or Response indicators. This is called the DPSIR framework and underlies the work and methodology of AEIs (table 1). The state of the environment is the main object for the framework, which can be an undesirable state (nitrate concentrations in groundwater), or a desirable state (an ecologically important habitat). The state can have a negative impact such as reduced drinking water quality, caused by a pressure. The pressure is s result of farming practices, causing the change of state (the pesticide risk which the environment has been a subject to). The pressure is driven by a driving force, such as the actual pesticide use. The driving forces are the targets of agri-environmental policy, as these are driven by market forces which can be altered by agricultural policy. Lastly, responses to the issues are measured. To clarify the concept underlying this framework examples from an agricultural case are used in table 1(Eurostat, 2018h).

Table 1. *DPSIR framework.* Conceptual example of the DPSIR framework in an agri-environmental context (Eurostat, 2018h)

Driving force	**Pressure**	**State**	**Impact**	**Response**
Farmers activities driven by market demands, such as pesticide use	Pesticide risk	Measured levels of pesticides in waterbodies	Reduced drinking water quality, detrimental effects on human health/ecosystems	Organic farming, farmers training level, protected areas

All the parameters mentioned in table 1 could be used as indicators for describing an agri-environmental phenomenon. However the most common indicators belong to the group *Driving Force* and *Pressure indicators*, they are often easier and cheaper to measure than *State indicators* (Wascher, 2002). A *State indicator* (or “results-oriented indicators” as referred to by Bockstaller et. al, 2008) is usually associated with high costs, as it is expensive to make actual measurements of e.g. pesticide levels in waterbodies on national level, or develop mechanistic models which could estimate pesticide levels. *Pressure indicators* or *Driving force indicators,* such as pesticide use, which can rely on data on national pesticide sales or farm statistics obtained by surveys, are easier and cheaper to produce but can many times fail in providing a clear link between pressure and state (Bockstaller *et al.*, 2008).

3 Review of some common agri-environmental indicators

In this chapter, some commonly used agri-environmental indicators by OECD, Eurostat, EEA and FAO will be reviewed. The following aspects will be discussed for each indicator: I) how the indicator relates to the environmental impact of food production; II) which environmental impact or pressures the indicator is coupled to (note; several indicators could be linked to numerous environmental impacts/pressures, only the 2-3 most significant impacts from a West-European perspective will be included); III) how well the indicator describes respective environmental impact when used at a national scale and; IV) the source of the indata.

Table 2. *Overlook of the indicators reviewed in this chapter* (Eurostat, 2018h)

Indicator	**Type of indicator**	**Unit**	**Used by**
Livestock density	Driving force	LSU/ha of UAA	Eurostat, OECD
Pesticide use	Driving force	kg active ingredient/ha/year	Eurostat, OECD, FAO
Mineral fertilizers use	Driving force	kg N and P/ha /year	Eurostat, OECD, FAO
Gross N and P balance on agricultural land	Pressure	kg/ha/year	Eurostat, OECD
Leaching of N and P	State	kg N and P/ha/year	NA*
Nitrate pollution of groundwater	State	mg NO_3^-/l	EEA
Ammonia emissions	Pressure	kilotonnes NH_3/year	Eurostat, OECD, FAO
GHG emissions from agriculture	Pressure	kilotonnes of CO_2-eq/year	Eurostat, OECD, FAO
Soil erosion by water	Pressure	t/ha/year	Eurostat
Agricultural water withdrawal	Pressure	% of total water withdrawal	Eurostat, OECD

*Leaching of N and P is not used as an indicator by any international organization encountered in this report.

3.1 Livestock density (LSU/ha of UAA)

65% of agricultural land, including grazing and production of feed crop, is dedicated to livestock production, corresponding to 28% of land surface in the European Union (Leip *et al.*, 2015). *Livestock density* (LSU/ha of UAA), is a driving force indicator, expressed as total livestock units/ha of utilized agricultural area (UAA). The indicator is calculated by dividing the sum of livestock units for a country by the area of arable land:

- Livestock unit; a reference unit, calculated by counting the number of heads/birds for each country, including cattle, goats, sheep, pigs, poultry, equidae (horses) and rabbits. The number of heads/birds is converted using species specific coefficients (see table 2), based on nutritional requirements, enabling aggregation of livestock of different species and age. 1 LSU is equivalent to one adult grazing dairy cow producing 3000 kg of milk annually (Eurostat, 2013).
- Utilized agricultural area; total area used as arable land, permanent/temporary grassland, permanent crops, kitchen gardens and fallow land (Eurostat, 2017h).

3.1.1 Relevance of indicator in assessing environmental impact of food production

The food demand of livestock products is steadily increasing, although somewhat stagnated in developed countries. Since 1980 the annual per capita meat consumption in developed countries increased from 73 kg to 78 kg, and is predicted to reach 89 kg by 2030. The corresponding numbers for annual per capita milk consumption in developed countries was 195 kg in 1980, 202 kg in 2002 and predicted to reach 209 kg by 2030 (Steinfeld, 2006). Within EU animal density is indirectly regulated through the Nitrates Directive where a maximum 170 kg N/ha of manure is allowed for application per year in nitrate vulnerable zones (NVZ), although EU Member States might have individual regulations. In Sweden, there is an additional maximum allowance of phosphorus of 22 kg P/ha year (Aronsson & Johnsson, 2017).

The indicator livestock density is coupled to several pressures on the environment;

- *Greenhouse gas emissions*; on a global scale the livestock sector emits 18% of the anthropogenic GHGs measured in CO_2 equivalents, and the main contributor is land-use-change (deforestation). The livestock sector also emits 37% of anthropogenic CH_4, originating from enteric fermentation by ruminants, and 65% of anthropogenic N_2O, derived from enteric fermentation by ruminants, manure management and N_2O-leaching from soils (Steinfeld, 2006).
- *Ammonia and NO_x emissions*; The excreta produced by livestock contributes to 75% of anthropogenic NH_3 emissions in Europe (Webb *et al.*, 2005) and can cause acidification of sensitive habitats and soil acidification after deposition to

land. Both NH_3 and NO_x contribute to the formation of secondary particulate matter (PM), decreasing air quality (Leip *et al.*, 2015)

- *Nutrient imbalances and eutrophication*; in Europe the livestock sector contributes to 23-47% of N river load to coastal waters, and 17-26% of the P river loads, where the upper limit includes mineral fertilizers used for feed production (Leip *et al.*, 2015)

3.1.2 Scale of use

The indicator *livestock density* is most often presented on national level, the relevance of the national scale is discussed for each pressure described in the previous section:

- *Greenhouse gas emissions*; Since GHGs are relevant on global scale, the indicator can be useful in indicating the total GHG pressure from livestock production on national scale, when used with supplementary indicators (GHGs emissions from agriculture (Eurostat, 2018i)).
- *Ammonia and NOx emissions*; Evaluating emission of NH_3 and NO_x on national scale of livestock density both fails to regard factors on farm level influencing emission of NH_3 such as kind of manure (solid/liquid), temperature, wind speed, pH of manure and soil, method for manure application etc. (Buijsman *et al.*, 1987). It also fails to describe the pressure on the environment as much of the locally emitted NH_3 is deposited in the close surroundings of the source (5 km^2) (8-50%). Further habitat variation in sensitivity to acidification, i.e. critical load exceedance, is not accounted for at national level (when limit of critical load is exceeded the ecosystem suffers from the nitrogen load) (Dragosits *et al.*, 2002).
- *Nutrient imbalances and eutrophication*; When evaluating N- and P nutrient loads, the national scale fails to indicate significant regional variations, shown in table 1, as nutrient leaching and eutrophication affect watercourses in connection to arable land. Further, several of the main factors decisive to the N and P nutrient loss from soil are not considered in this indicator, such as climate, precipitation, soil type, soil P-content, topography, hydrology, tillage practices, manure management and application (Magdoff *et al.*, 1997; HELCOM, 2011). The country scale does hence fail to pin-point areas under elevated risk for nutrient leaching.

Table 3. *Livestock density. Comparison of results for the indicator "Livestock density" as estimated by FAO, Eurostat and SCB for some water district areas in Sweden* (Eurostat; Widell & Hedevind, 2003; FAO, 2018a)

Source (year)	**Livestock density Sweden (LSU/ha)**		
		Average	**Variation***
FAO (2014)		0.62	
Eurostat (2013)		0.56	
Water district area Sweden (SCB, 2000)	Kalmar	0.44	0.28–0.95
	Blekinge	0.44	0.28–0.97
	Skåne	0.32	0.00–0.60
	Halland	0.48	0.00–0.68

*The variation indicates highest respectively lowest LSU/ha found for basins within respective water district.

3.1.3 Source of indata

Data underpinning this indicator is obtained from the Farm Structure Survey (FFS) in a harmonizing procedure, which is done by all EU Member States every 3-4 years as sample survey, and every 10 years as census. A census, covering all members of the population gives more accurate data than a survey, including a sample of the population Data is also attained from crop and livestock statistics gathered each year within the European Union. The FFS is conducted by all agricultural holdings within the EU Member States (Eurostat, 2017d). Until 2007, it included all agricultural holdings with a UAA of at least one hectare. Since 2008, several extra thresholds have been incorporated into the definition 'agricultural holding' (Eurostat, 2017e). The coefficient used for aggregating livestock units is an important part of the indicator. However, coefficients used vary between FAO, Eurostat and Swedish Board of Agriculture as shown in table 2. Thus, the result of *livestock density* will vary depending on the source.

Table 4. *Livestock coefficients. Comparison of some livestock coefficients used to calculate livestock units* (Chilonda & Otte, 2006; Eurostat, 2013; SBA, 2017a)

Livestock		**FAO**	**Eurostat**	**Swedish Board of Agriculture**
Bovine animals	Under 1 year old	n.d..	0.4	0.17*
	Heifers, ≥ 2 years	n.d.	0.8	0.33
	Dairy cows	0.9	1	1.0
Sheep and goats		0.1	0.1	0.1
Equidae		0.65	0.8	1.0
Pigs	Breeding sows ≥ 50 kg	0.25	0.5	0.33
Poultry	Broiler	0.01	0.007	0.005
	Laying hens	0.01	0.014	0.01

*The coefficient 0.17 is used for animals younger than six months

3.2 Mineral fertilizer use (kg N and P/ha)

Fertilizer consumption per hectare of fertilized utilized agricultural area (kg N or P/ ha of fertilized UAA) is a driving force indicator, calculated by dividing the sum of estimated country consumptions of mineral nitrogen (N) and phosphorus (P) fertilizer, by hectares of UAA. Rough grazing and fallow land are excluded since they do not receive mineral fertilizer (Eurostat, 2017b).

3.2.1 Relevance of indicator in assessing environmental impact of food production

Use of easily available forms of plant nutrients N and P in modern agriculture have been key components in providing increasing yields with adequate nutritional quality for a growing population, meeting its dietary preferences (Galloway & Cowling, 2002). On average 64 kg N/ha of UAA and 6.3 kg P/ha of UAA where applied in 2015 within EU. The average trend is that N applications are increasing and P applications decreasing in EU (Eurostat, 2017b). Use of mineral fertilizers have altered global nutrient flows of N and P, and the surpluses from crop production have several environmental impacts. Further mineral fertilizers have enabled the specialization of previously mixed farms to crop or livestock farms. This has led to concentration of livestock farms in some areas, importing feed crops from areas with favorable crop production conditions. Regional nutrient imbalances arise as nutrients from crop production areas are accumulated in areas with high livestock production, lacking enough land area for spreading excess manure (Carpenter *et al.*, 1998; Bouwman *et al.*, 2013). Again, insufficient nutrient inputs on harvested land leads to depletion of nutrients, organic matter and reduced soil fertility (Vitousek *et al.*, 2009). Environmental impacts/pressures coupled to mineral fertilizer use include:

- *Nutrient imbalances and eutrophication*; N and P surpluses contribute to eutrophication of water ecosystems and pollution of surface and ground water. N is primarily lost by leaching from the root zone and P loss occurs by both soil erosion and leaching (Djodjic *et al.*, 2004; Eurostat, 2017b).
- *Greenhouse gas emissions*; Main on-farm GHG emission coupled to mineral fertilizer application is the soil release of N_2O due to microbial redox-processes such as denitrification and nitrification boosted by application of N-fertilizers. Mineral fertilizer have an estimated emission factor (EF) of 1%, meaning that about 1% of applied N mineral fertilizer is transformed to N_2O. EF has in recent studies been shown to accelerate when application rates exceed plant requirements (Shcherbak *et al.*, 2014).

The EU Nitrates Directive, indirectly regulates application of mineral fertilizers (legally binding for farmers within NVZ). Examples of measures regarding fertilization are: regulation of fertilizer application to snow covered or frozen ground, sloping ground, ground close to water courses. The Directive requires nutrient management plans for N, based on crop demand etc. (Aronsson & Johnsson, 2017).

3.2.2 Scale of use

The indicator is used on national level, the relevance of the national scale is discussed for each pressure described in the previous section:

- *Nutrient imbalances and eutrophication*; This indicator should be used cautiously when estimating nutrient losses at national scale, since it does not contain information on which actual area has been fertilized or what crop has been grown, as crops have different nutrient requirements (SBA, 2017b). Furthermore, soil type and soil characteristics are important factors when estimating nutrient loss. Clay soils tend to be sensitive to P run-off, and sandy soils are coupled to N leaching (Sogbedji *et al.*, 2000)
- *Greenhouse gas emissions*; The indicator can roughly say something on a national scale of N_2O emissions due to mineral fertilization, using EF. However recent studies show that EF is not linear, but exponential and increases as N-application exceeds plant requirements. Furthermore, EF was higher for nitrogen-fixing crops, pH < 7, and soils with soil organic carbon (SOC) > 1.5%. To reveal hot-spots, farm level conditions should be considered such as fertilizer regime and what crops are cultivated (Shcherbak *et al.*, 2014).

3.2.3 Source of indata

Data on N and P used in the indicator come from Member State estimations. The methodology is not harmonized across Member States, since there is no legal requirement on reporting use of fertilizers. Countries use data based on sales, production and trade statistics which might include fertilizers not used for agricultural purposes. Some countries use farmer surveys and their reliability depend of sampling size and design. There is also no harmonization in whether to use crop year or calendar year for collection and reporting of data (Eurostat, 2017c). When accounting for fertilized UAA, land use rough grazing and fallow land are not included, since these land types are not fertilized (Eurostat, 2017b). There are likely more land use types included in UAA which do not receive fertilizer, such as arable land under organic management. Since these fields receive less mineral fertilizer than conventional fields it could have a diluting effect as the fertilizer consumption can be divided on a larger area, resulting in too low estimations of fertilizer consumption/ha UAA.

3.3 Ammonia emissions (kilotonnes NH_3/year)

Ammonia (NH_3) emissions is a pressure indicator, presented in kilotonnes NH_3/year, used by Eurostat. The collection of data on NH_3 emissions is done by EU Member States for the reporting to United Nations Economic Commission for Europe (UNECE) under the Convention on Long-Range Transboundary Air Pollution (CLRTAP) and the EU National Emission Ceilings Directive fort certain pollutants (NEC) (Eurostat, 2018a).

Livestock production is the main source of NH_3 emissions in Europe. The indicator is therefore to a large extent based on livestock patterns. NH_3 emissions are calculated by using statistics for livestock patterns, animal manure and fertilizer use. The statistics are then converted to estimated NH_3 emissions by multiplication with emission factors (EF). There are numerous EF depending on type of manure, manure storage- and spreading system, livestock species, N-efficiency etc. (SEPA, 2017c).

3.3.1 Indicator relevance for assessing environmental impact of food production

Agriculture represents 94% of total NH_3 emissions in EU (Eurostat, 2018a), of which livestock excreta contributes to 80-90% of the total agricultural emissions (Webb *et al.*, 2005). Other minor contributors are mineral N-fertilizers, which in Sweden represented 4% of NH_3 emissions from agriculture in 2009 (Staaf & Bergström, 2009). Once NH_3 is emitted it can be deposited close to its source, but also transported up to 2500 km by atmospheric transport, causing environmental impact far away from the original source. NH_3 emissions are primarily linked to the following environmental impacts:

- *Surface water and soil acidification;* NH_3 deposition has an acidifying effect on surface waters and soils. Soils can buffer acid deposition to some extent through cation exchange, however, once equilibrium has been reached acidifying substances will start to appear in run off and contribute to the acidification of surface waters. In general, acidification alters species composition, reduces decomposition rates and productivity of both terrestrial and aquatic ecosystems (Fangmeierfl *et al.*, 1994; Hildrew & Steve, 1995) .

- *Eutrophication;* The deposition of NH_3 emissions can cause eutrophication in water courses as nitrogen levels are increased, which can alter the flora and fauna composition and reduce water quality (Fangmeierfl *et al.*, 1994).

3.3.2 Scale of use

This indicator is used on national level, the relevance of the national scale is discussed mutually for surface water and soil acidification and eutrophication since similar reasoning can be applied to both impacts:

- *Acidification of surface water and soil, and eutrophication;* NH_3 emissions as indicator on national scale is useful for setting national emission goals but three difficulties have been identified for the national scale in this report: I) There is large local variation of NH_3 emissions, depending on livestock housing and manure storage losses, landspreading of manure and grazing (Dragosits *et al.*, 2002). II) NH_3 can be deposited in close surrounding of the emission source, as well as up to 2500 km from the source. Even within a 5km grid area the deposition can have high variation. Hence both local and transnational scales are important. III) Habitats susceptibility to NH_3 deposition differs greatly, why the concept of critical loads exceedance (the amount of loading an ecosystem can tolerate before negative impacts occur) has been developed for both nitrogen loading and acid loading. The exceedance of critical load varies between ecosystems as they have different buffering capacities (Fangmeierfl *et al.*, 1994; Dragosits *et al.*, 2002).

3.3.3 Source of indata

EU Member States are responsible of reporting national NH_3 emissions, to UNECE, for the CLRTAP, and to EEA under NEC Directive (EEA, 2017b). Eurostat reuses the data published by EEA for their indicator on ammonia emissions (Eurostat, 2018a). There is a standard model developed by the EEA for calculating NH_3 emissions with several EF, however countries are encouraged to adapt the model to country specific conditions. Sweden has partly deviated from the default model, by adjusting EFs, as the Swedish climate is considerably colder than European average (Andersson *et al.*, 2017). Furthermore, the Swedish model has incorporated several more variables with specific EFs comparing to the EEA guidelines such as timespan between spreading and mulching manure, spreading strategy, season for spreading and type of manure storage (EEA, 2016; SEPA, 2017c).

Simplified, the Swedish NH_3 emissions are calculated by using data on animal manure from surveys performed by Statistics Sweden. Data on livestock populations are obtained by the farm register produced by the Swedish Board of Agriculture, as well as data on nitrogen levels in animal manure and excrete, based on nutrient balances. The country specific EFs are then applied to the statistics giving an estimation of national NH_3 emissions (SEPA, 2017c).

3.4 Gross N and P balance on agricultural land (kg/ha)

The pressure indicator *gross nutrient balance* is used by Eurostat and OECD for measuring nutrient efficiency of nitrogen (N) and phosphorous (P) in agriculture as well as to estimate environmental risk to air, water and soil coupled to nutrient surpluses. Simplified, nutrient budgets are calculated by subtracting nutrient outputs from nutrient inputs (nutrient inputs – nutrient outputs = nutrient surplus/deficit), where the sum zero indicates a balance between nutrient inputs and outputs. The sum is divided by the total utilized agricultural area (including arable land, permanent grassland and permanent crop, preferably excluding land under extensive management).

When calculating nutrient budgets different methods can be used. A *farm budget*, compares import and export of nutrients for a system. The system boundaries can be the farm, a region or the whole country. This method does not consider nutrient cycling within the system. A *soil budget* is more detailed and accounts for all flows entering and leaving an area of arable soil. N lost by volatilization as (N_2O), ammonia (NH_3) and nitrogen gas (N_2) is subtracted. The surplus therefore provides an estimation of net nutrient loss by leaching and runoff. The *land budget*, which is the recommended method by Eurostat and OECD and will therefore be the method referred to in this chapter, estimates the actual usage of fertilizers and manure on the farmland, as well as atmospheric deposition of N and biological N fixation (BNF). This scale considers nutrient recycling on the farm by accounting for grazing and harvesting of ley, the re-deposition of manure and crop residue inputs etc. The input – output represents the gross N and P which can be lost by leaching, runoff and volatilization (OECD & Eurostat, 2013).

The gross N budget is important for measuring progress of the Nitrates Directive, which aims at protecting European waters from agricultural nitrate pollution (European Commission, 2016c). Both the N and P budgets are important for following up the Water Framework Directive, which aims at ensuring good quality of European inland waterbodies and marine waters (European Commission, 2016a)

3.4.1 Indicator relevance for assessing environmental impact of food production

The productivity of arable land used for crop production depends on the returning of nutrients to the soil, to avoid nutrient depletion, as nutrients are continuously removed by crop harvest. A yield of 6 t/ha of wheat will approximately remove 110 kg N/ha and 19 kg P/ha (SBA, 2017b). Other nutrients are also applied such as potassium (K) and micronutrients, but do not pose an environmental concern and are therefore not included in the indicator. Nutrients can be applied in various forms such as mineral fertilizers, animal manure, crop residues, nitrogen fixation by crops, sewage sludge, compost and other organic fertilizers (OECD & Eurostat, 2013). If nutrients in harvested crops exceed nutrient inputs continuously (indicated as a deficit in the nutrient budget) depletion of nutrients stored in soil stocks can occur. This can lead to soil degradation, with reduced yields and increased risk of erosion as consequences (Tan *et al.*, 2005). Nutrients which are not taken up by the plant can either be stored in soil nutrient stocks, or be lost by the means of leaching, runoff (applies to both P and N) or emitted in various gas forms (applies only to N) (OECD & Eurostat, 2013).

P and N behave differently in soil due to differences in chemical characteristics. N has a complex cycle and can go through various biochemical processes in soil, shifting between plant available (NO_3^- and NH_4^+) form and plant unavailable form (incorporated into organic material, also referred to as soil N stocks). It can be exchanged to air as the inert N_2, as NH_3 or as the greenhouse gas N_2O. Further, it can be fixed from the atmosphere to the soil by biological nitrogen fixation or anthropogenically by the Haber-Bosch method (fixes atmospheric N_2 to NO_3^-) (OECD & Eurostat, 2013).

P primarily occurs in soil by weathering of P rich minerals naturally found in soils. P has no exchange with the atmosphere, but can just as N be converted by microbiological processes in the soil from plant available form (orthophosphates) to plant unavailable form (organically bound). P in soils is divided into three categories: I) Solution P pool, a very small fraction that partly consist of the plant available orthophosphates; II) Active P pool, which refers to P adsorbed to soil particles but can easily be released and taken up by plants; III) and fixed P pool, which is the largest pool and contains organic and inorganic P in insoluble forms meaning that they are rarely made plant available. An important difference between P and N is that excess P can be stored in the active and fixed P pool, while excess N cannot be stored in soil, unless incorporated in organic material (OECD & Eurostat, 2013). As P can be readily stored in the soil, it has a lower correlation between nutrient surplus and nutrient loss compared to N. However, high, occasional P losses often occur once soil is saturated by P (Djodjic *et al.*, 2004). Thereby, a surplus of P may have effects on the risk of losses in the long-term. Following environmental concerns coupled to the indicator:

- *Eutrophication*; P and N can contribute to eutrophication once they end up in waterbodies. P is often the limiting nutrient for eutrophication to occur (Djodjic *et al.*, 2004), and can possess a greater risk to the environment than N, even though it is lost in smaller quantities. Eutrophication is the phenomena of obsessive growth of algae and primary production in aquatic ecosystems, due to excess inputs of nutrients. Once algae degrade, microorganisms conducting the degradation, consume excessive quantities of oxygen, leading to oxygen deprived conditions, in the sediments. This state alters the species composition and is a major concern of the Baltic Sea (HELCOM, 2011).
- *Pollution of surface and groundwater;* Both P and N can act as contaminants in surface and ground waters (OECD & Eurostat, 2013). Nitrate is a common pollutant in ground water (see indicator chapter on nitrate in groundwater).
- *Climate change;* Excessive amounts of N in the system means increased risk of losses, as N_2O emissions during microbial denitrification processes occurring in soil and manure. N_2O is a potent GHG contributing to climate change (see indicator chapter on GHG emissions).
- *Soil and water acidification;* A share of the nitrogen surplus will be volatilized as NH_3, because of poor, or inadequate, manure storage and manure application. After deposition NH_3 can cause soil and water acidification (see indicator chapter on ammonia emissions).

3.4.2 Scale of use

Gross nutrient balance is used on national scale for international comparisons as well as to follow national progress. As the indicator does not explicitly demonstrate GHG loss, NH_3 loss or losses due to leaching and runoff, the indicators *GHG emissions from agriculture, ammonia emissions from agriculture and nutrient leaching* will give more accurate descriptions of these environmental pressures. The indicator should be used carefully when making country comparisons as data comparability is estimated to be < 75 % (see section Source of indata). The indicator on national scale is therefore best used as a benchmark for countries to follow their own progress. However, the regional differences can be large within a country due to differences in production, climate and other geographical conditions (see table 4). As an example, the Southern plain of Götaland has higher N surpluses (43 kg N/ha) than the national average (33 kg N/ha), due to intensive crop production with high inputs of mineral N fertilizers. The P balance is lower (-4kg P/ha) than average (0 kg P/ha) due to lower than average country inputs of manure. This can be compared to the woodlands of Götaland where livestock production is intensive and inputs of manure are high resulting in a P balance of 4 kg P/ha. This is line with the assumption that the largest nutrient surpluses are found in areas with high livestock densities (> 1LSU/ha) (SCB, 2013).

As demonstrated, the national scale can be too rough, whereas the regional scale cover differences which arise between regions. Further, the path of nutrients i.e. if they are lost to air, water or stored in soil stocks varies between regions. Highest leaching of N is found in the southern parts of Sweden (36 kg/ha), due to climate, soil type and production intensity, whereas NH_3 emission will be highest in areas with intensive livestock production as the woodlands of Götaland (SCB, 2013).

Table 5. *Nutrient balances. Swedish nutrient balances expressed as kg N or P/ha. Table shows both regional variations and variations due to the use of different methods for calculations* (SCB, 2013; Eurostat, 2016b). *Note that the 'Soil surface balance' used by SCB is corresponding to the 'Land budget' used by Eurostat/OECD, presenting the gross nutrient balance*

	SCB (2013)				**Eurostat/OECD (2013)**	
	Soil surface balance		**Farm gate**		**Land budget**	
	N	**P**	**N**	**P**	**N**	**P**
Country scale	33	0	51	1	35	-1
Southern plain of Götaland	43	-4				
Woodlands of Götaland	41	3				

3.4.3 Source of indata

Member states report data on inputs and outputs required to calculate nutrient balances to OECD and Eurostat. In case data is missing the organizations make their own estimates. The indicator is thus a collaboration between Eurostat/OECD and Member States. National reported figures might therefore differ from figures reported by Eurostat/OECD (table 4). Countries are expected to follow the Eurostat/OECD handbook for calculating their N and P budgets using the land budget (OECD & Eurostat, 2013).

Inputs to be included are the following; mineral fertilizers, organic fertilizers, livestock manure, BNF, atmospheric N deposition, seed and planting materials (OECD & Eurostat, 2013). In Sweden, most of this data comes from Farm Structure Surveys (FSS). Data on atmospheric deposition is obtained by the Swedish Meteorological Institute (SMHI). Biological nitrogen fixation is calculated using a Danish model adjusted for Swedish conditions (SCB, 2013). Issues coupled to input estimations are data on fertilizers, as some countries use sales statistics, which might include fertilizers not used in the agricultural sector. Crop statistics, which only contain data on main crops, often miss the use of nitrogen fixing crops as they are used as secondary crops for nitrogen fixation, outside the growing season. Furthermore, countries use different coefficients, reducing international comparability (OECD & Eurostat, 2013).

Outputs to be included are sold crops and fodder crops, fodder crops for own use, and grass (harvested and grazed). Sweden compiles output data based on farm structure surveys (FSS), normative values of yields and farmer interviews (SCB, 2013). It can be difficult to estimate nutrient cycling due to grazing and harvested grass as this is not always included in general farm statistics. Countries are therefore recommended to do calculations based on recommended animal feed requirements. Another possible bias is delimitation of what land is accounted for as utilized agricultural area (UAA), the most extensively used lands should not be included as it receives less fertilizer. There is however no harmonized methodology for how to make the delimitation between extensive and intensive land. There is also the issue of excluding certain important flows due to the difficulties in calculating them such as stock changes of N and P in soil which can be considered as an input or an output depending on if the stocks are increasing or decreasing. Crop residues are not regarded in the current budget, nor atmospheric deposition of P (OECD & Eurostat, 2013).

The altogether estimated comparability of the data is less than 75% due to different methodologies in calculations and the possible biases mentioned above (Eurostat, 2018k).

3.5 Leaching of N and P from agricultural land (kg N or P/ha/year)

Nitrogen (N) and phosphorous (P) leaching from agricultural land can be regarded as a state indicator, indicating the pollution of marine, surface and ground waters posed by nutrient losses from agricultural land. It is used by HELCOM (Baltic Marine Environment Protection Commission – Helsinki Commission) to assess nutrient loads to the Baltic Sea (Pollution Load Compilations, PLC), and for following up the Swedish Environmental goal of “Zero Eutrophication” (SMED, 2013). Furthermore, it can be used as a complement to the indicator *gross nutrient balances,* as this indicator calculates nutrient surplus but does not describe how much of the surplus is lost to water and air and stored in soil. Thus, *leaching of N and P from agricultural land* explicitly describes the nutrients lost to water bodies. The indicator is usually calculated by using system models (SMED, 2013).

3.5.1 Indicator relevance for assessing environmental impact of food production

The indicator *Leaching of N and P from agricultural land,* can act as a complement to the indicator *Gross N and P balances on agricultural land* (chapter 3.4), as it is a direct estimate of the nutrient leaching from agricultural land, provided that it is possible to make somewhat accurate calculations. The nutrient surpluses which can occur within agriculture can be lost to the environment by volatilization to the air, lost to water by leaching and runoff or stored in soil stocks. This chapter will focus on the leaching of N and P, by starting with a basic description of the general behavior of N and P in agricultural soils. Leaching, as referred to in this paper, accounts for the nutrients lost from the root zone by leaching (for P, surface runoff is included in leaching) i.e. nutrients which have passed the root zone and can no longer be taken up by plants. The nutrients will eventually reach ditches and watercourses, either by ground water transport or transport through drainage pipes (SMED, 2013).

N is often found in the soluble ion form as NO_3^- in soil, either added as such by mineral fertilizers, or formed by the processes of mineralization of organic material. Organic material such as soil organic matter, manure, green manure, crop residues, or N fixed by N-fixing crops, and other organic fertilizers can be transformed through mineralization (transformation of organic N to NH_4^+), and further oxidized through nitrification (NH_4^+ converted to NO_3^-). These processes are conducted by microorganisms and occur continuously in the soil, at decreased rates during periods with low temperatures. NO_3^- can be taken up by plants or move quickly through soil as it does not adsorb to negative clay particles or form any other soil particle surface complexes. Nitrogen is therefore prone to leaching, if not taken up by the crop, or incorporated in organic material through immobilization (NO_3^- converted to organic N). It is most easily leached from sand and silt soils, as they have high percolation capacity in areas with high precipitation. The leaching from soils with high clay content is generally lower as percolation is slower and NO_3^- is instead lost at higher rates by denitrification, under water saturated conditions, as NO, N_2O or N_2 (Eriksson *et al.*, 2011).

P has a different leaching pattern than N, as P added to soil, either in organic or inorganic form, can easily form complexes with soil surfaces, and thus become inaccessible to plants and build up soil P pools. P is therefore often leached as particulate-P, or lost by runoff. However, the largest loss in a context of leaching, is as dissolved reactive phosphorus (DRP), which is the readily plant available form. In contrast to N, which is relatively evenly lost in quantity, scale and time, large amounts of P can be lost from minor parts of the field under short time periods when the degree of P saturation in soil is high (Djodjic & Bergström, 2005). The sorption of P is highly correlated to the soil pH and recent studies show that the highest portion of soluble P usually occur at pH below 4.5 and above 7.5, as P containing precipitates are dissolved (Eriksson, 2016). Soils with high clay content are generally more prone to P leaching than sandy soils as they can form macropores through which particulate-P can be lost (Eriksson *et al.*, 2011). The following environmental concerns are coupled to P and N leaching:

- *Eutrophication*; P and N can contribute to eutrophication once they end up in waterbodies. N is usually the nutrient limiting the growth of green algae, whereas P is limiting for cyanobacteria, as they can fix their own nitrogen (Eriksson *et al.*, 2011). The ecological state of the Baltic Sea is a major concern. Anthropogenic diffuse (non-point) sources of N and P cause the greatest load to the Baltic Sea, after natural background losses, and agriculture is often the most important source. Agriculture contributes to 60-70% of the diffuse P and N loads to the Baltic Sea from Sweden. Important point-sources are municipal waste water treatment plants and industry (HELCOM, 2011).
- *Pollution of surface and groundwater;* Both P, and mainly N, can act as contaminants in surface and ground waters (OECD & Eurostat, 2013). Nitrate is a common pollutant in ground water, regarded for its possible detrimental health effects to humans. EU has therefore applied a threshold of 50 mg NO_3^-/l for drinking and ground water (see indicator chapter on *nitrate in groundwater* for a more detailed report).

3.5.2 Scale of use

Leaching of N and P is highly dependent of abiotic variables such as climate, soil type, atmospheric deposition and hydrology, biotic factors such as soil microfauna and vegetation, as well as human factors such as production orientation and intensity. For estimating P leaching, soil P content and topography are important factors as well. Thus, when used on national scale an indicator like this must consider all these small-scale variations. To highlight the variance, examples will be used from a report on N and P leaching from Swedish agricultural land produced by the Swedish Environmental Emission Data (SMED, 2013). Estimated N leaching from Swedish arable land was 19 kg/ha/year in 2013, but the regional variance was 6-46 kg/ha/year, depending on both natural prerequisites, types of crops, fertilization and tillage management. For P, the corresponding figures were 0.6 kg P/ha/year with a regional variance of 0.13-1.33 kg P/ha/year (SMED, 2013).

N leaching can vary greatly depending on what crop is grown, primarily due to differences in growing periods and root length. For example, potato, which in Sweden has a relatively short growing season and high N content in its residues, has high N leaching (> 55kg N/ha/year) compared to a 5-year ley (<10 kg N/ha/year).

N leaching depending on soil type differed between about 10 kg N/ ha when winter wheat was grown on a clay soil, compared to about 50 kg N/ha when grown on a sandy soil (SMED, 2013).

The regional differences for P leaching are not as dependent on cropping regimes as N, rather precipitation, runoff and soil type are decisive. However, an important crop factor is the soil cover, as soil cover reduces the risk of particulate-P surface runoff. An example of the soil factor is that P leaching could vary between 0.5 – 2.5 kg P/ha/year between a sandy soil and a clay soil in the same region and for the same crop. The highest P leaching was found in areas with the highest average annual runoff, the Swedish west coast (1.33 kg P/ ha), and the lowest P leaching was found at the islands of Öland and Gotland (0.13 kg P/ha), which also have the lowest annual average runoff (SMED, 2013).

3.5.3 Source of indata

Various tools can be used to estimate nutrient leaching; simulation models (empirical/mechanistic), estimations from monitoring, rough assumptions from nutrient budgets or estimated through both models and measurements. In Sweden, the meta-model SOILNDB, which is built on mechanistic models, respectively the simulation model ICECREAMDB are used for N and P (SMED, 2013), for calculations at national level, e.g. for the PLC calculations. A mechanistic model assumes that a system can be understood by examining the mechanics of the systems individual parts, such as the soil-plant system processes in this case, and their impact on nutrient leaching. A meta-model is a user-friendly version of the mechanistic model, as simplified inputs (readily available data) can be used, and is converted through algorithms to fit the mechanistic model (Johnsson *et al.*, 2002a).

The models include water and heat fluxes, and transformations and transport in soil for N and P, based on system research. Both models are built on an adjustable matrix, where the user can choose between 22 regions, based on climate (precipitation, runoff, solar radiation, daily average temperatures), 10 different soil classes and 13 different crops, to estimate region specific leaching. For P, 3 soil P classes and 3 gradients are included in the matrix (SMED, 2013). The model can thus result in 2869 and 25 821 specific N respectively P-leaching values. The leaching from a region can vary greatly between years, primarily due to runoff/precipitation variability. Average 30-year climate data is therefore used in the model. The same reasoning applies to yields, as they can vary greatly between years. Therefore, estimations of area-average yields are used in the models (SMED, 2013).

Input data on crop sequences and cultivation measures (fertilizer, tillage, catch crop etc.) come from Statistics Sweden (farm structure surveys) and Swedish Board of Agriculture (farm registers) (SMED, 2013). It should be emphasized that simulation models only make estimations, as they cannot mimic the system fully. This is especially true for P simulation models as we still lack knowledge about P behavior in soil due to its complexity (Larsson *et al.*, 2007). Further, the quality of output data is strongly dependent on the quality and resolution of the input data. The input data for the models relies on statistics based on farm structure surveys and assumptions which might contribute to margin of error. As an example, ICECREAMDB has been found to have an r^2-value of 0,75 when compared to actual field measures (Heckrath *et al.*, 2007)

3.6 Greenhouse gas emissions from agriculture (kilotonnes of CO_2-eq/year)

The indicator *greenhouse gas (GHG) emissions from agriculture* is a pressure indicator expressed as kilotonnes of CO_2-equvivalents. The main GHGs emitted from agriculture are methane (CH_4) and nitrous oxide (N_2O), which are converted to CO_2 equivalents based on their Global Warming Potential (GWP) where the baseline is one entity of CO_2. The GWP over a 100-year period is 265 for N_2O and 28 for CH_4 (Myhre *et al.*, 2013). The indicator does not cover the sequestration or emission of CO_2 from agricultural soils, caused by changed cropping patterns or land use change, such as the converting of cropland to grassland or vice versa. The GHG emissions originating from these activities are accounted for under the Land Use, Land Use Change and Forestry (LULUCF) inventories. CO_2-emissions derived from energy use or transport in the agricultural sector are not included, as they are accounted for in the energy sector according to Intergovernmental Panel on Climate Change (IPCC) framework (Eurostat, 2018b).

3.6.1 Indicator relevance for assessing environmental impact of food production

Population and economic growth are the main drivers behind anthropogenic GHG emissions, which have resulted in the highest atmospheric concentrations of GHGs for the preceding 800 000 years at least (IPCC, 2014). The agricultural sector emits roughly 10% of European GHGs. The main sources are N_2O emission from soils (3.7%) and CH_4 emissions from enteric fermentation by ruminants (4.3%) (Eurostat, 2018b). Since LULUCF is not included in this indicator, direct CO_2-emissions are a minor part of total agricultural emissions (2% in Sweden in 2015). The reason for this is that a large source of agricultural CO_2 emissions comes from the cultivation and drainage of organic soils, primarily peat soils, and this is accounted for under the LULUCF inventory. Peat soils contain large carbon stocks, built up by organic plant materials which cannot be fully degraded under the anoxic conditions typical for these types of soils. Once peat soils are drained and cultivated they start to act as carbon sources, as plant material is oxidized. It is estimated that 6-8% of total Swedish anthropogenic GHG emissions are coupled to the cultivation and drainage of peat soils. Berglund and Berglund estimated that Swedish agricultural peat soils emitted up to 5600 Gg CO_2-eqvivalents, although they only make up about 8 % of total agricultural land (Berglund & Berglund, 2010). CO_2 loss accounts for 85-95% of total peat emissions, N_2O for 5-15% and CH_4 for less than 1% (Norberg, 2017).

CH_4 is produced in microbial processes when organic material is decomposed under anaerobic conditions such as enteric fermentation by ruminants and manure storage. Other contributors are rice cultivation under flooded conditions, soil management and field burning of agricultural residues (Smith *et al.*, 2007). N_2O is produced under the microbial processes nitrification and denitrification, that are a part of the global nitrogen cycle. In agricultural soils, 50% of the N_2O emissions are coupled to soil applications of mineral fertilizers or animal manure (Shcherbak *et al.*, 2014). However, all nitrogen containing organic inputs to agricultural soils can contribute to N_2O emissions. When crop residues, organic matter, nitrogen-fixing crops or cover crops are incorporated into soils, mineral forms of nitrogen can be released through mineralization, and thereafter be lost as N_2O to the atmosphere through the same pathways as mineral fertilizers and manure (Paul, 2014).

Agriculture can also play a role in mitigating climate change, soil carbon sequestration is considered one of the options with biggest potential. This can be achieved by increasing crop yields or photosynthesis, and thereby increasing soil carbon sequestration. Some examples for how this can be done is by using adequate crop varieties, extending crop rotations, avoiding nutrient deficit, using cover crops etc. (Smith *et al.*, 2007). Emissions of GHGs contribute to the following environmental impacts:

- *Climate change and global warming;* GHGs readily absorb and emit light of the infrared spectra emitted by earth and the atmosphere (AMS, 2012b), thus changing the earths radiation balance i.e. energy flux. Climate change is the earths response to the change in net energy flux, as an attempt to reset the balance (Myhre *et al.*, 2013). Climate change is the long-term changes of climatic conditions such as wind, temperature and pressure (AMS, 2012a). Environmental impacts of climate change are numerous; distorted hydrological patterns due to altered snow and ice melt and precipitation, decreased food security, ocean acidification, rising sea levels, altering of species composition in aquatic and terrestrial ecosystems, increased frequency of extreme weather events such as heavy rainfalls, droughts, heat waves, floods, wildfires and cyclones (IPCC, 2014).

3.6.2 Scale of use

This indicator is used on national scale. However, it is total global GHG concentrations which determine the magnitude of climate change. National GHG emissions contribute to global impacts due to climate change, and the environmental impacts are unevenly distributed globally where some regions suffer more than others (IPCC, 2014). Thus, the national scale is relevant for reaching national reduction targets in compliance with the Paris Agreement which aims at reducing the global threat of climate change. The target set by the Paris Agreement, which entered into force in 2016, is to keep global warming well below 2 degrees Celsius above pre-industrial era (UNFCCC, 2014).

3.6.3 Source of indata

EU Member States generate their own data on GHG emissions from the agricultural sector by following IPCC framework. The data is reported to the United Nations Framework Convention on Climate Change (UNFCCC) under a common reporting format. Nations are encouraged to use country specific methodologies for higher accuracy (Eurostat, 2018b). Countries collect agricultural data and use emission factors (EF) to estimate GHG emissions (table 4). IPCC offers three methods (Tier 1-3) for calculation of GHGs emissions, where Tier 3 has the highest accuracy as it is based on advanced country specific methods. Tier 1 uses default methods and default EF values. Tier 2 uses country specific EFs. Choice of Tier will depend of the nation's ability to provide own data and methods. Sweden uses Tier 1 and Tier 2. Agricultural data is obtained by Statistics Sweden and Swedish Board of Agriculture, complemented by data from industry associations (SEPA, 2017a).

For the indicator, *Greenhouse gas emissions from agriculture,* GHGs are converted to CO_2-eqvivalents, to enable aggregation and to facilitate communication on GHG contribution to climate change, by using GWP which is based on how effectively the gas absorbs infrared radiance, which wave-lengths of the infra-red spectra it absorbs and its lifetime in the atmosphere. There are significant uncertainties coupled the use of this metric (Myhre *et al.*, 2013), however, it should not affect comparability between countries. Aggregation of GHG also disregards the factor that GHGs are important on different time scales due to their expected lifetime in the atmosphere. CH_4 emissions are important over a 20-year period, while CO_2 emission will contribute to the largest global warming over a 50 to 100-year period (Myhre *et al.*, 2013).

Table 6. *Emission factors. Example of some implied emission factors (IED) used by Sweden for the UNFCCC National Inventory Submission 2017. The emission factors are used to calculate direct and*

indirect N_2O emissions from agricultural soils after application of mineral fertilizer/manure as kg N_2O-N/kg N. For cultivation of organic soils IED is expressed as kg N_2O-N/ha (UNFCCC, 2017)

	Inorganic fertilizer	Animal manure	Sewage sludge	Urine and dung deposited by grazing animals	Crop residues	Cultivation of organic soils
Sweden	0.01	0.01	0.01	0.02	0.01	13.00

3.7 Pesticide use (kg active ingredient/ha/year)

Pesticide use is a driving force indicator expressed as kg active ingredient/ha/year. It is calculated by dividing total applications of pesticides by the sum of arable land and permanent crops, annually (FAO, 2016c). The active ingredient in a pesticide product is the chemical part that controls pest and makes up a portion of the total pesticide volume (NPIC, 2015). Kilogram active ingredient is calculated by multiplying the concentration of active ingredient with total pesticide volume.

3.7.1 Indicator relevance for assessing environmental impact of food production

Pesticides have been a key element contributing to an increase of food quality and a fourfold increase of food grain production between 1950-2000, by protecting crops from pest and weeds. Pesticides are divided in several categories. Insecticides, herbicides and fungicides are the most common, making up 44%, 30% and 21% of total global pesticide consumption (Aktar *et al.*, 2009). The actual global loss of wheat due to pest and weeds was 28.2% between the years 2001-2003, the same number for maize was 31.2% and 28.8% for cotton (Oerke, 2006). Pesticides can spread to soil, water and air by several pathways and pose an environmental risk. In context of spreading to waterbodies the pathways are divided in point and diffuse sources. Diffuse sources are sources derived from application of fertilizers in agricultural fields, which cannot be directly localized. Point sources are one or a couple of sources that can be pinpointed in connection to the waterbody (Reichenberger *et al.*, 2007). The environmental risk posed by pesticides depends on the chemical features such as volatility, water solubility and adsorbaility. Pesticides in the environment undergo degradation by biological chemical and photochemical pathways. Pesticide persistence depends of the degradation rate. Substances with high persistency and low adsorbaility have the highest spreading risk (SLU, 2016). The following environmental impacts are coupled to pesticide use:

- *Surface water contamination;* Tile drain discharge, leaching, runoff, soil erosion, spray drift during pesticide application and deposition after volatilization are some diffuse sources of pesticides to surface water (Reichenberger *et al.*, 2007). Chemical concentrations exceeding species tolerance have a toxic effect, and a negative impact on aquatic life (Aktar *et al.*, 2009)

- *Ground water contamination;* Diffuse sources of pesticides in ground water are leaching and infiltration through river beds and banks. Point sources can occur in combination with filling equipment as accidental spills and sewer overflows (Reichenberger *et al.*, 2007). Pesticides in ground water can reduce quality of drinking water (Aktar *et al.*, 2009). The drinking water limit for individual pesticide compounds is 0.1 μg/l in EU Member States (Swedish Water, 2016).
- *Soil contamination;* Some pesticides are strongly bound to soil, depending on chemical properties such as hydrophobicity and persistency. Organic content of soils increases the adsorption of positively charged pesticide residues. Pesticides in soil can cause decline of microfauna i.e. bacteria and fungi, important for several soil processes, causing soil degradation (Aktar *et al.*, 2009).

3.7.2 Scale of use

The same reasoning regarding scale of use will be applied to the environmental impacts identified in the previous section. Three difficulties have been identified with the use of the indicator *Pesticide use* on national scale for drawing conclusions on the environmental impact associated with pesticide application: I) How and to what extent a pesticide will spread to the environment depends on the pesticide specific chemical characteristics as discussed in the previous section. Pesticides can cause environmental impact close to the source but also miles away, after volatilization, by atmospheric long range transport. The numerous pathways of a pesticide after field application means that farm field perspective needs to be applied for each pesticide to discover the potential risks it might pose to the environment (Reichenberger *et al.*, 2007).

II) The indicator aggregates all crops and all types of pesticides. However, there is large variation in pesticide treatments between crops both in how many applications they receive per season, dosage and what type of pesticide they receive, depending on which pest possesses the largest risk for a crop in a certain region (table 7). Sugar beets are poor competitors against weeds in their early establishment stages, why they receive large amounts of herbicides. Swedish food potato is prone to some severe fungal diseases such as potato blight why its fungicide usage number is high (SCB, 2010).

III) Chemicals used in pesticides are attributed Predicted No-Effect Concentrations (PNEC), which is the highest concentration below which no adverse effect is measured in an ecosystem (Baun *et al.*). As chemicals have different PNEC values a high kg a.i./ha might not necessarily mean a high environmental impact.

Due to the complexity of pesticides and their associated environmental risks mathematical models are used as tools to estimate local risk. One such model is HAIR2014 found at the pesticide models EU webpage. This model accounts for climate, soil type, crop, type of pesticide, application rate and several other parameters decisive for the estimations (Kruijne *et al.*, 2014).

Table 7. *Pesticide use. Application of pesticides (kg active ingredient/ha) for crop year 09/10 of some common Swedish crops. Figures within parenthesis represent the % of total area treated* (SCB, 2010)

		kg a.i./ha			
	Crop area (ha)	**Insecticide**	**Fungicide**	**Herbicide**	**Total**
Winter wheat	331 805	0.02 (33)	0.26 (70)	0.41(94)	0.61 (94)
Table potato	15 807	0.05 (20)	2.44 (90)	1.05 (85)	3.41 (91)
Sugar beet	37 950	..	0.13 (38)	3.74 (98)	3.79 (98)

3.7.3 Source of indata

The indicator *pesticide use* expressed as kg active ingredient/ha and year is used by FAOSTAT. FAO obtains data on pesticide use from their annual questionnaire on pesticides. Data is usually based on national statistics on pesticide sales, imports and productions. Some countries specifically report pesticides used for crops and seeds, while some countries report aggregate data for agriculture, forestry and garden use. It is not considered that sold quantities within a year are not always used during the corresponding period, but stored on the farm (FAO, 2016c). FAO data on land use is gathered by annual questionnaires on land use. The sum of arable land and permanent crop presented by FAO in 2015 was equal to the figure presented by Statistics Sweden in 2015 (SCB, 2015b). Data reported by countries to FAO can have varying comparability due to inconsistency in classification, reasoning or sources of data (FAO, 2016b).

As shown in table 8, data on land use and sold quantities of pesticides from organizations from same years can differ. Possible explanations could be disparities due to type of pesticides included in their calculations or how the definitions “arable land” and “permanent crop” are defined, and how they are derived. FAO and Eurostat use different definitions of arable land e.g. (Eurostat, 2017f; FAO, 2017).

The differences of kg a.i./ha presented in table 8 is a consequence of inconsistencies in data but also of different methods for calculating the metric. Statistics Sweden is responsible for compiling Swedish pesticide statistics in compliance with the national environmental goal “a toxic-free environment”. They make calculations by using sales statistics and retail information on recommended doses for pesticide application. Thus, they calculate how many hectares the sold volumes of each pesticide cover, and do not use actual statistics on land use (SCB, 2015b).

Table 8. *Examples of pesticide data from different sources. All data applies to Sweden for year 2015 if not stated as other* (SCB, 2015b; FAO, 2016c; Eurostat, 2017a)

	FAOSTAT	EUROSTAT	SCB
Sold quantities of a.i. (tonnes)	1826	2307	1698
Pesticides included in sold quantities	I, H, F, PGR, R	F, B, H, I, A, PGR, HD, M, MK, OP	H, I, F, PGR
Arable land + permanent crop (1000ha)	2590	2614*	2590
kg a.i./ha	0.71	0.88**	0.8

Abbreviations used in the table; F=Fungicides, H=Herbicides, I=Insecticides, PGR=Plant Growth Regulator, R=Rodenticides, HD=Haulm Destructors, M=Molluscicides, MK=Moss Killers, OP= Other Pesticides

*Data from 2010

**Writers own calculation based on Eurostat data presented in this table (tonnes a.i./1000 ha)

3.8 Nitrate pollution of groundwater (mg NO_3^-/l)

Nitrate is considered as a major pollutant of both ground and surface water, hence the indicator *Nitrate pollution of groundwater* used by several agencies such as Eurostat, OECD and EEA. It is a state indicator expressed as mg NO_3^-/l and usually presented as percentage of monitoring sites exceeding threshold values for groundwater and drinking water limits set by EU. Several directives within EU regard nitrate pollution of water. According to the Nitrates Directive (91/676/EEC) and Drinking Water Directive (98/83/EC), groundwater, surface water and drinking water should not exceed concentrations of 50 mg NO_3^-/l. Concentrations above 25 mg NO_3^-/l indicate a level of concern (European Commission, 2016c; Eurostat, 2018e). It should be noted that nitrate concentration can be expressed as either the nitrate ion (NO_3^-) with the molecular weight 64g/mole, or as the N weight of NO_3^-, expressed as NO_3^--N, with a molecular weight of 14g/mole. The threshold of 50 mg NO_3^-/l corresponds to 11.3mg NO_3^--N/l (WHO, 2011). This report will further refer to nitrate concentrations (NO_3^-/l).

3.8.1 Indicator relevance for assessing environmental impact of food production

Nitrate (NO_3^-) is a soluble and mobile form of nitrogen, easily taken up by plants and naturally occurring in most environments. For groundwater in connection to undisturbed ecosystems such as temperate grasslands the baseline concentration is

2 mg NO_3^-/l, or 0.5mg NO_3^--N/l (Wakida & Lerner, 2005). However elevated concentrations of nitrate are often measured in ground waters due to urban and agricultural activities. In urban areas leaking water mains, leaking sewers, waste water treatment plants and landfills are major nitrate sources to ground water (Wakida & Lerner, 2005). In agricultural areas nitrate leaching from the root zone on arable fields is the major contributor to elevated NO_3^- concentrations in ground water. Driving forces for nitrate leaching from the root zone have been identified as intensification of crop production and livestock production, with high livestock densities and excess production of manure (Strebel *et al.*, 1989). Application of mineral fertilizer and manure, as well as incorporation of green manure and crop residues, and atmospheric deposition to soil, are factors increasing soil nitrogen pools which can be lost by leaching. Mineral nitrogen fertilizers are applied in directly plant available forms of nitrogen (ammonia or nitrate). If applied in excess or if rain occurs before plants have taken up the nitrogen, leaching can occur as nitrate is a mobile compound. When manure or other organic fertilizer is applied, nitrogen bound in organic form needs to be converted to the plant available ammonia or nitrate. This occurs through mineralization, which is a slow process highly dependent of temperature and positively correlated to nitrogen content of the organic substrate. Risk for leaching is high if mineralization occurs after harvest when there is no crop to take up the nitrogen. A high risk period for leaching in Sweden is during autumn, winter and early spring due to precipitation and snow melt, if there is no crop cover and especially on sandy soils due to their high hydraulic conductivity (Jarvis *et al.*, 1996; Johnsson *et al.*, 2002b). The following environmental impacts are coupled to nitrate pollution of groundwater:

- *Reduced drinking water quality*; Groundwater is considered as a better source for drinking water than surface water as it has an even, low temperature and is relatively free from pollutants and therefore requires less treatment. Nitrate is a common pollutant of groundwater (SEPA, 2002). Nitrate pollution of drinking water is known for causing infant methemoglobinemia, an unusual, acute condition where the oxygen transport ability in blood is decreased, of which infants fed with well water are group at highest risk (Greer & Shannon, 2005). The various cancer risks coupled to elevated nitrate levels in drinking water are debated and under research (WHO, 2011). In the body nitrate can convert to nitrosamines, known for their risk of causing several gastric and intestinal cancers, and the drinking water limit of 50 mg NO_3^-/l has recently been questioned (Hansen *et al.*, 2018). However, nitrate can be transformed to NO, by the nitrate-nitrite-NO pathway in the human body, which has been linked to the prevention of several diseases (Lundberg *et al.*, 2008). Further, epidemiological studies show that a dietary nitrate intake, primarily from vegetables, can play a role in fortifying the immune system of mammals and reduce the risk of gastric cancers. If this is due to the positive effects of a high vegetable intake, or the nitrate itself, has not been clarified (Duncan *et al.*, 1997). Elevated nitrate concentration of groundwater is generally not a risk for households receiving municipal water, as nitrates are removed by methods with varying efficiency and cost (Pretty *et al.*, 2000). Households receiving water from private wells (close to farmland or municipal water treatment disposal) are a risk group as they carry their own responsibility for testing water quality as well as installing nitrate removing facilities which can be expensive (Greer & Shannon, 2005).

3.8.2 Scale of use

The indicator *nitrate pollution of groundwater* is best used at regional scale, as several factors, both site specific and coupled to farm practices, are decisive for the nitrate leaching from agricultural land. Thus, when only regarding the country scale regions with high nitrate concentrations in ground water are neglected. Site specific factors are those coupled to climate and soil, such as precipitation, topography, soil texture and soil organic matter. Factors influenced by agricultural practices are tillage practices, crop rotations, which impact soil cover during autumn and winter, root depth and water and nitrogen uptake. Furthermore, type of fertilizer, doses and time of application are important factors (Strebel *et al.*, 1989).

On national scale, 2% of the monitoring sites exceeded 50 mg NO_3^-/l in Sweden which could be regarded as a low figure (EEA, 2015). As the indicator is based on the analysis of a smaller number of monitoring sites (110), it fails to show regional differences or the status of about 250 000 private wells, supplying drinking water for 1 140 000 Swedish inhabitants and livestock (SCB, 2015a; Enköpings-Posten, 2017). In 2002, the Swedish Environmental Protection Agency (SEPA) published a report on the state of nitrates in groundwater in Sweden. The report was based on municipal surveys, data on 30 000 private wells from the Swedish Geological Institute (SGU), and calculations by the Swedish University of Agricultural Science (SLU) (SEPA, 2002). The report concluded that regions with elevated nitrate concentrations can be found in the south and south west parts of Sweden. In these regions, the number of private wells with elevated nitrate concentrations can be high, as well as the cost for municipal nitrate water treatment. For the Kristianstad plain it was estimated that 30-40% of the shallow private wells had nitrate concentrations exceeding 44 mg NO_3^-/l (recommended nitrate threshold by the Swedish National Food Agency prior to year 2001). The suggested reason for this was the high portion of sandy soils, root-crop cultivation and intensive pig production in the area (SEPA, 2002).

3.8.3 Source of indata

The European Environmental Agency (EEA) is responsible for presenting European data on surface and ground water quality as comprehensive indicators. The indicator *nitrate pollution of groundwater* consists of aggregated annual average concentrations from groundwater monitoring stations (EEA, 2015). Data used by EEA is reported by Member States under the Water Information System (WISE-SoE). Member States are responsible for monitoring national ground water quality, following harmonized guidelines for data reporting, and encouraged to use already implemented monitoring stations. Data for nitrate should be reported disaggregated, as individual samples. Each sample must be coupled to a monitoring site, accompanied by the spatial characteristics of the site (Eionet, 2018).

The Swedish Geological (SGU) institute is responsible for compiling Swedish water quality data, while the monitoring is performed by County Administrations. 110 trend stations are monitored annually, and another 528 stations are monitored during a six-year period. The stations which are monitored are located in areas unaffected by human activity and should serve as reference when following up the results of environmental actions (SGU, n.d.). There are several factors which can influence the certainty and thus the comparability of data both on national and regional scale: I) the human factor; there can be variations in how the personnel handles sampling, use the equipment correctly and store samples correctly; II) site of sampling; the most common sampling spots for Swedish water quality monitoring are springs. The groundwater quality of springs is sensitive to shifting groundwater levels. This is a limitation from a sampling perspective as the concentration of pollutants can shift quickly; III) weather conditions; weather conditions prior to sampling, such as heavy precipitation can impact the sampling outcome. Thus the sampler is responsible for giving accurate description of the sampling site, weather conditions, land use, water conductivity, air temperature, sampling depth in relation to soil surface etc. which can be used for correction when data compiling is done by SGU (Tunemar, 2016).

It is probable that monitor designs, sampling methods and number of monitoring sites varies between countries because of country size, hydrologic patterns and available resources (Loaiciga, 1989).

3.9 Soil erosion by water (tonnes/ha/year)

Soil erosion by water is a pressure indicator, used to describe soil degradation and is expressed as lost soil in tonnes/ha/year. Soil is considered a non-renewable natural resource due to its slow formation. Loss rates exceeding 1 ton/ha/year are considered as irreversible. The need for soil protection has been stressed on EU level through The Soil Thematic Strategy, including soil protecting measures such as legislation, recommendations, guidelines and voluntary agreements. Soil erosion is one the soil degrading factors identified under the Soil Thematic Strategy (Van-Camp *et al.*, 2004). The erosion indicator focuses on soil loss by water as this is the erosion form with the highest impact on soil degradation of arable land in Europe (Panagos *et al.*, 2015).

3.9.1 Indicator relevance for assessing environmental impact of food production

Soil is defined as the top layer of earth´s crust consisting of mineral particles, organic matter, water, air and living soil organisms. Soil fills numerous functions essential for human life. It provides a medium and nutrients for growing crops and thus producing food, it is the habitat of numerous species and a vital part of the hydrological cycle (European Commission, 2016b). Furthermore, the global carbon stored in soil is 3.3 times bigger than the carbon pool stored in the atmosphere (2500 Gt versus 760 Gt) (Lal, 2004).

Erosion by water is one of the major threats of soil quality and it is estimated that 12.7 % of the arable land in EU suffers from moderate to high erosion, equivalent to a soil loss of 5-20 tonnes/ha/year (Eurostat, 2018f). Erosion is the physical process where finer soil particles are moved by water or wind and transported to other locations. Three major processes are known to cause soil erosion by water; *heavy rainfall*, when rainfall is powerful enough it can detach soil particles and move them short distances. *Overland flow,* occurs when saturated soil is overflowed by excess water from rain or snow melt. The last process is known as *rills*, developed when overland flow forms preferential pathways where water can move rapidly, and further erode the passageway (Eurostat, 2018f). Soil erosion is a natural process forming our landscapes. However human activities have exaggerated the process and agriculture is a major driving force for human induced erosion (Van-Camp *et al.*, 2004). Agricultural practices identified to have a negative correlation to soil erosion are tillage practices with high disturbance rate such as deep soil ploughing, intensive irrigation, soil left bare after harvest, and high grazing intensity. Reversely good agricultural practices can reduce erosion risk through enhancing soil structure (e.g. increasing organic matter content) and soil cover. Generally, this can be achieved by reduced tillage, continuous land cover by crops, customized crop rotations, moderate grazing, converting arable land to pasture and various other farm specific measures such as grass strips (SoCo Project Team, 2009). Soil erosion influences the following environmental aspects:

- *Soil degradation*; Topsoil is the upper part of the soil which contains the most nutrients, adsorbed to mineral soil particles or incorporated in organic matter, and has the highest rate of microbiological activity. The topsoil is the part of the soil exposed to erosion. Organic matter, which can compose 4% of total soil weight, for a fertile soil (Poeplau et al. 2015 estimated that Swedish agricultural soils contain 2.65% C on average), contains about 95% of soil N and up to 50% of soil P. Organic matter is important for enhancing soil structure by contributing to the formation of soil aggregates thus increasing soil porosity and water infiltration. The loss of organic matter and impaired soil structure reduces the water holding capacity of soil and increases the runoff from land. Altogether, these impacts have directly negative impact on crop productivity (Pimentel, 2006). Further, loss of mineral soil, such as erosion of clay and silt soils, is an important path for P loss (Djodjic & Bergström, 2005).

- *Eutrophication and water pollution;* Globally about two-thirds of all soil eroded from crop land by water is deposited in rivers, lakes and streams. The soil particles can contain both nutrients and pesticides causing contamination and eutrophication of aquatic ecosystems (Pimentel, 2006; Ekholm & Lehtoranta, 2012) .
- *Climate change:* The soil organic matter lost by erosion, is primarily built up by carbon. Once eroded the carbon will be oxidized and released as the greenhouse gas CO_2, thus contributing to climate change. Erosion has contributed to an estimated 26 Gt carbon loss since 1850, compared to the burning of fossil fuels, which has contributed to a 270 Gt carbon loss. One of the consequences of climate change is altered hydrological patterns with increased frequency of extreme rainfall thus causing a fortifying feedback loop (Lal, 2004).

3.9.2 Scale of use

The consequences of the national scale of use for the environmental impacts described in the previous section are discussed below:

- *Soil degradation*; From the perspective of global food security (it is estimated that 10 million ha of cropland is abandoned yearly due to reduced productivity caused by soil erosion) the national scale of soil erosion rates is useful. On the other hand, soil degradation is an issue for the individual farm holding, as land productivity decreases (Pimentel, 2006). From this perspective, the national scale is not useful, as the soil erosion rate can vary greatly within a country foremost depending on soil type, land management, topography and precipitation (Panagos *et al.*, 2015). For agricultural fields, the tillage practice can be very important. In Norway the annual soil erosion rate by water was 4.36 t/ha in an autumn ploughed field, compared to 0.46 t/ha for a field harrowed in spring (fields in the same location, same soil type and topography)(Oygarden *et al.*, 2006). Areas with relative high erosion risk in Sweden are the counties around lake Mälaren with high soil clay content, and the silty soils on the northwest coast and the southwest coast (Ulén, 2006).

- *Eutrophication and water pollution;* For estimating eutrophication and water pollution the country scale is to broad, as the erosion rate varies between regions in a country and even between sections of a field. The appropriate scale would be the land located within a basin. Additional factors which also need to be acknowledged are; the complexity in estimating to what extent the eroded material will leave the field and be deposited in water bodies (estimates of eroded material leaving the field varies between 5-80% in Swedish studies). It is also difficult to distinguish how much of the sediment in a watercourse origins from arable land and how much from eroded sides and bottoms of the watercourse. Studies show that phosphorous loss was strongly linked to soil type and less linked to soils with relative high erosion risk (Ulén, 2006).
- *Climate change:* Climate change is driven by total global emissions of GHGs. The nation scale for estimating GHG emissions coupled to erosion would be appropriate, as it is the total to soil carbon loss due to erosion which is interesting from this perspective. Some of the countries with the highest erosion rates in Europe are those with a combination of steep topography and high rainfall erosivity such as Italy, Austria and Slovenia that all have mean annual erosion rates exceeding 7 t/ha/year (including all lands) (Panagos *et al.*, 2015).

3.9.3 Source of indata

In EU soil erosion is calculated by using the soil erosion model Revised Universal Soil Loss Equation (RUSLE2015), as it is not feasible to measure erosion directly. The model calculates average soil loss in t/hectare/year (E) (by using the equation presented in table 7) at a 100-m resolution and only accounts for water erosion. Parameters included in the model are rainfall erosivity (R), soil erodibility (K), cover-management (C), topography (LS) and support practices (P) applied as layers in the model at different resolutions (table 7). Input data in the model consist of data from European databases, remote sensing tools and soil samples. Under the parameter ‘support practices’, the model can account for Good Agricultural and Environmental Condition (GAEC) which are required under the EU Common Agricultural Policy (CAP). Soil protection practices which can be accounted for in the model include reduced and no tillage, crop residues, cover crops, grass margins, contour plowing and preservation of stone walls (Panagos *et al.*, 2015; Eurostat, 2018f).

Table 9. *Summary of the layers and parameters included in the RUSLE2015 model, and sources of the input data* (Panagos *et al.*, 2015)

Layer	Parameters included	Resolution (m)	Source of data
E= Average soil loss (t/ha yr)	E= R x K x C x LS x P	100	..
R= Rainfall erosivity (MJ mm/ha h yr)	Rainfall erosivity estimates	500	Rainfall Erosivity Database
K= Soil erodibility factor (t ha h/MJ mm)	Sand, silt, clay, organic carbon, permeability, coarse fragments, structure, stone cover	500	LUCAS Soil, European Soil Database, LUCAS Earth Observation
C= Cover management factor	Arable, non-arable, vegetation density, crops, cover crops, tillage, plant residues	100	CORINE Land Cover, Copernicus Remote Sensing, Eurostat
LS= Slope length and slope steepness factor	LS factor	25	Digital Elevation Model
P= Support practices factors	Contour farming, stone walls, grass margins	1000	LUCAS Earth Observation, Good Agricultural Condition

The model is best used to predict long-term, annual soil erosion. However, the model has also received criticism for not being able to predict event soil losses at all geographic locations as it was developed for application in New South Wales, Australia and does not have an explicit runoff factor (Kinnell, 2010). To what extent this affects the European modelling of soil erosion using RUSLE2015 is not clear.

3.10 Agricultural water withdrawal (as % of total water withdrawal)

The indicator *agricultural water withdrawal* is a pressure indicator used by the Food and Agriculture Organization of the United Nations (FAO) and Eurostat. It implies a pressure on water resources and presents the total water use by the agricultural sector as the percentage of the whole country's water abstraction (ground and surface water). Agricultural water withdrawal includes water used for irrigation, livestock and aquaculture, and it can include the use of agricultural drainage water, direct use of treated wastewater and desalinated water. Total water withdrawal includes water abstracted for municipal, industrial and agricultural use. Industrial processing of meat, dairy and crops is included under industrial water use (FAO, 2016a). The indicator often builds on figures reported by national authorities or other organizations. It is useful for measuring the progress of achieving sustainable use of water resources, set by the EU Water Framework Directive, as well as the EU

‘Blueprint to safeguard Europe’s water resources’, which includes a strategy for water scarcity and droughts (EEA, 2012).

3.10.1 Indicator relevance for assessing environmental impact of food production

Human societies depend on water and its natural cycles which provide various ecosystem services, both provisional and regulating, e.g. water for producing food, groundwater recharge and flood control. However, human activities put pressure on water resources and the European Environment Agency (EEA) has described land use, water abstraction and climate change as the three, human induced, major threats to natural water availability in Europe. European agriculture accounts for 33% of total European water use. In some regions of southern Europe the figure can be as high as 80%, primarily due to irrigation of agricultural crops, during periods of the year when water availability is low (EEA, 2012). Irrigation requirements vary greatly within Europe due to climatic conditions, type of crops and soil type, with generally higher irrigation requirements in the Mediterranean region and lower requirements in the boreal regions of north Europe. Estimated net irrigation requirement for Sweden is 22 million m^3 per year. The corresponding number for Spain is 35 919 million m^3 per year. The actual amount of water required for irrigation is usually 1.3-2.5 x higher, than the absolute amounts needed, as the irrigation efficiency varies (Wriedt *et al.*, 2009).

The indicator “agricultural water withdrawal” can be more informative by examining the type of water usage, as use of the concept “water footprint” (WF) (Hoekstra & Mekonnen, 2012). WF accounts for water used for production and consumption of goods, and includes the use of freshwater, referring to groundwater and surface water (blue water), rain water (green water) and volumes of polluted water (grey water). Globally, agricultural production accounts for 92% of the WF. As an example, the consumption of cereals, meat and milk products account for 56% of the total WF for the average human, compared to the 9% derived from consumption of industrial products and domestic water use (Hoekstra & Mekonnen, 2012). Livestock, indirectly require large amounts of water, primarily through the water required to produce their feed. As an example, to produce 1 ton of cereals requires 1644 m^3 water, while 1 ton of produced beef requires 15 415 m^3 water. The water requirement of livestock highly depends on the feed conversion ratio (how effectively feed is converted into desired livestock outputs). Thus, intensive production generally has lower WF than extensive production forms, as the feed conversion ratio increases in intensive systems. However, the ratio of blue and grey WF increases in the intensive systems as livestock in intensive systems receives higher portions of concentrate feed. Producing feed crops requires larger amounts of blue and grey water, while roughage will require larger amounts of green water. Extensive systems primarily obtain nutrients from grazing and roughage, thus the WF consist of a larger ratio of green water (Mekonnen & Hoekstra, 2012). The WF for 1 ton of beef produced in an intensive system, requires 10 244 m^3 of water, of which 683 m^3 is blue water and 712 m^3 is grey water. The corresponding numbers for an extensive production form, the total WF is 21 827 m^3, of which 243 m^3 is grey water and 465 m^3 is blue water (Mekonnen & Hoekstra, 2012). The WF approach also suggests that countries can import significant amounts of water through food commodities and animal feed, thus exporting water usage, which is not accounted for in the national water abstraction figures. The following environmental impacts can be coupled to agricultural water withdrawals:

- *Water scarcity;* An unsustainable abstraction of freshwater can cause water scarcity of both groundwater and freshwater. Water scarcity is a human induced phenomenon, which occurs when water usage exceeds the renewal of water, or when water becomes scarce due to pollution (Schmidt & Benítez-Sanza, 2013). This is typically a more pronounced issue during summer months in south Europe, but concerns have also been raised in northern Europe during the last years, due to the impact of climate change on altered precipitation patterns and increased temperatures (Wriedt *et al.*, 2009). Improper management of water, and water scarcity, can exacerbate the implications during a drought (Schmidt & Benítez-Sanza, 2013). It is estimated that 86 million Europeans live in areas which experience water stress during the summer months (EEA, 2018b). Water scarcity has detrimental effects on human health, industry, agricultural productivity and the ecological status of water bodies. Aquatic species within a water body, are adjusted to certain water flows and variability, when this is altered due to abstraction, the species composition also alters (EEA, 2012).

3.10.2 Scale of use

Agricultural water withdrawal can act as an indicator on country scale on the sustainability of water use. However, when using the country scale, which is presented on annual basis, seasonal variations are not included. The seasonal variation can be of great concern, as water scarcity can occur during some periods of the year, and the magnitude of the impacts will vary depending on the volume abstracted and the season of the abstraction (EEA, 2012). An example is the dry spell in Sweden during the summer of 2016 and the very low groundwater levels during the spring and summer 2017, particularly in the south and southeast parts of the country, caused by low precipitation (SGU, 2017; SMHI, 2017).

Further, the indicator does not show regional variabilities. In countries, generally not considered as countries with low water availability, some river groundwater bodies might be under pressure. For example, both Denmark and United Kingdom, had groundwater bodies with low quantitative status within several river basin districts (RBD), where the percentage of groundwater bodies with low quantitative status within the RBD ranged between 10-70 % (EEA, 2012). Therefore, when using this indicator to predict water scarcity, and other aquatic implications coupled to excessive water abstraction, it is necessary to both regard the seasonal variations as well as regional hot spots. This is partly possible with the EEA water exploitation index (WEI), where water abstraction from individual RBDs can be explored. It expresses total freshwater abstraction of total renewable freshwater within a RBD. A WEI above 40% is regarded as highly unsustainable natural resource use (EEA, 2018b).

3.10.3 Source of indata

The data on this indicator, which is found at AQUASTAT, FAOs global water information system, is obtained by FAO voluntary questionnaires directed towards official governments, with no standardized methodology on how to estimate the data. Thus, the data can be based on information from national authorities, expert judgements, other published figures or external data obtained by other international agencies (FAO, 2012). Some of the AQUASTAT data for EU Member States is obtained by Eurostat, as they gather information for EU policy progress on water management. This data is also obtained by national questionnaires and national data collection (Eurostat, 2018l).

In Sweden, agricultural water withdrawal is calculated by estimating crop and livestock water requirements. Crop requirement is calculated by estimating the percentage of irrigated area per crop, using data from farm structure surveys. The water required for irrigating a certain area of respective crop is calculated by using coefficients for plant water requirement by region and crop, developed by the Swedish University of Agricultural Sciences. The water required for livestock is calculated by using livestock numbers from farm registers, and coefficients of annually water requirement per animal/day in m^3, developed from data on daily water requirements per animal/day (SCB, 2015c).

4 Linking agricultural environmental impact to food production

In this chapter, the results of the agri-environmental indicators (AEIs) reviewed in the previous chapter, are presented and discussed. For comparison, results for countries with similar farming and climate conditions as Sweden, will be presented. When possible, the result of the AEIs will be presented in a novel manner, to better link the connection between agricultural environmental impact and food production. Countries used for comparison, in addition to Sweden (SE), are Finland (FI), Norway (NO), Denmark (DK), Germany (DE), the Netherlands (NL), United Kingdom (UK) and Ireland (IE).

4.1 Anthropogenic vs. natural conditions

The result of the AEIs presented in this chapter, depend on both natural conditions found within a country, i.e. climate, soil type, and anthropogenic factors, primarily agricultural practices. Most of the AEIs depend on both, however, some AEIs depend more strongly on anthropogenic factors, for instance *NH_3 emissions*. NH_3 emissions highly depend on manure management, and there are methods which could abate up to 80% of the NH_3 emissions, for example by incorporating manure quickly after spreading to soil (Webb *et al.*, 2005). On the other hand, the outcome of the AEI *Soil erosion by water,* greatly depends on natural conditions, such as topography and soil type, although, agricultural practices can enhance/reduce the erosion rate (Panagos *et al.*, 2015). Table 10 presents an overlook of the importance of anthropogenic and natural conditions for each indicator, which can be helpful for understanding the results presented in the following chapters. The focus in this part of the report, will be on the AEIs which can be affected by agricultural practices, as these can describe the difference in environmental performance between countries more accurately than AEIs highly dependent of natural conditions.

Table 10. *Anthropogenic and natural factors. The table demonstrates if the results of AEIs are predominantly influenced by anthropogenic factors or natural conditions. The order of the AEIs listed below should not be regarded as definite, but merely as an indication for if the AEI primarily depends on natural conditions or anthropogenic factors*

	Agri-environmental indicator	**Main influencing factors**
Highly depends on anthropogenic factors	Ammonia emissions	Livestock, manure management
	Mineral fertilizer use	Crop type, crop rotation, nutrient management
	Pesticide use	Crop rotation, weed and pest pressure, weed and pest management
	Livestock density	Market demand, structure of agriculture
	Gross N and P balance on agricultural land	Nutrient management
	GHG emissions from agriculture	Manure and nutrient management, land management, production orientation, soil conditions
	Leaching of N and P	Fertilization, crop type, soil type, tillage, hydrology
	Nitrate pollution of groundwater	Fertilization, crop type, soil type, hydrology
	Agricultural water withdrawal	Climate, water use efficiency, crop type
Highly depends on natural conditions	Soil erosion by water	Precipitation, topography, soil type, land management

4.2 National background data

Table 11 offers an overview of national differences in agricultural production and climate conditions. Data in the table is used as basis for calculating environmental performance of food production. Permanent grassland in table 11 refers to land used for pasture, rough grazing, mown for hay or silage, and used as such for at least 5 consecutive years (Eurostat, 2017g). Length of growing period (LGP) refers to the period of the year when daily average temperature exceeds 5°C (SLU, 2006). Raw milk amounts refers to milk collected from dairy farms, including milk used on the farm. IE, UK and DK are the countries with the highest share of Utilized Agricultural Area (UAA) of total land (TL), which comprises about 70%. However, a large part of the UAA in IE and UK is used as permanent grassland, 79% and 66% respectively. In SE about 7% of TL is used as UAA, and 15% of UAA is used as permanent grassland. SE and FI are the countries with largest forest land area, which makes up about 70% of TL.

Table 11. *National data on natural conditions and production figures. If nothing else is stated, the data comes from the Eurostat production database, for the latest available years. The % of land use*

are calculated by using Eurostat figures on land use (Eurostat, 2018j). *Abbreviations: UAA; Utilized Agricultural Area, LGP; Length of Growing Period, TL; Total Land, PG; Permanent grassland, OF; Organic farming. Root crops refers to potatoes and sugar beet. Organic farming refers to land certified according to EU organic farming or land in transition from conventional to organic farming*

Conditions	**SE**	**FI**	**NO**	**DK**	**DE**	**NL**	**UK**	**IE**
[1]Population (10^6)	10.1	5.5	5.2	5.7	82.8	17.1	66.6	4.8
[2]Precipitation (mm/year)	624	536	1414	703	700	778	1220	1118
[3]Temperature (mean °C/year)	4.7	2.7	4.3	7.5	7.8	9.3	9.3	9.6
[4]LGP (days)	120-240	90-180	60-180	180-240	180-240	210-240	210-300	240-300
UAA (1000 ha)	3036	2258	987	2619	16700	1847	17096	4959
PG (% of UAA)	15	1	18	9	28	41	66	79
Forest (% of TL)	70	68	44	15	32	9	13	11
UAA (% of TL)	7	7	3	61	47	43	69	71
OF (% of UAA)	18,3	10,5	4,9	7,7	6,8	2,9	2,8	1,7
Production (1000t)	**SE**	**FI**	**NO**	**DK**	**DE**	**NL**	**UK**	**IE**
Cereals	5964	3398	1326	9872	45593	1344	22753	2240
Oil seeds	285	95	11	506	4676	10	1823	33
Pulses	195	65	n.s.	55	523	6	886	65
Root crops	1981	939	305	4561	33276	11520	11806	n.s.
Raw milk	2883	2390	1594	5385	31973	15090	14732	6851
Bovine meat	132	85	82	124	1123	441	899	617
Pig meat	240	179	138	1544	5455	1456	900	294
Poultry meat	158	129	98	160	1513	1036	1835	152
Eggs, hen in shell	140	73	68	86	812	716	722	44

n.s.= non-significant value. [1] Wikipedia (2018), [2]FAO (2016a),[3]Weatherbase (2016), [4]FAO (n.d.), [5]SSB (2018)

4.3 National production figures expressed as kcal and protein

To better understand the environmental impact of food production, and link it to the production of crops and animal products, the production of Animal- and Plant Protein and kcal has been calculated for each country, using production figures from Eurostat. Production figures have been converted to protein and kcal using conversion factors, presented in Appendix 1. The commodities presented in table 11 are included in the calculations, i.e. only crops used primarily for human consumption are included in the calculation of Plant-Protein and calories (cereals, oil seeds, pulses and root crops). Sheep and goat meat is not included, as this only comprises a small part of the total production. It has been taken into consideration that about 24% of cereals produced in Europe are used for human consumption, whereas the rest is used as animal feed (60%) or for industrial purposes (11%) (SBA, 2014). As the results for kcal production per country from plant and animal sources closely correspond to the protein production (high protein production will result in high kcal production for any chosen country) only the results for protein production are presented in this section.

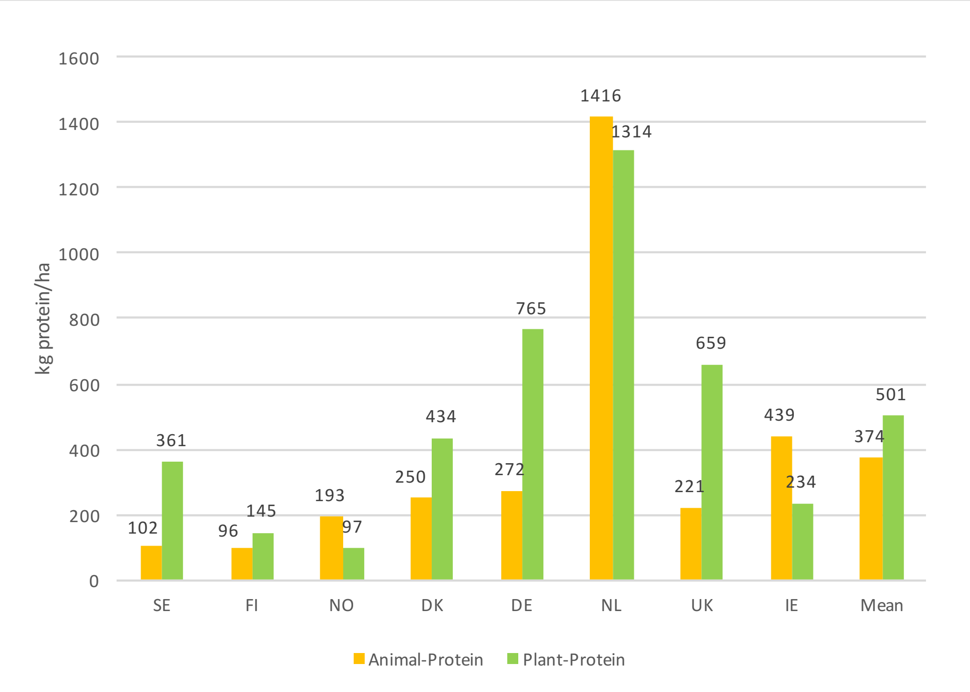

Figure 1. Animal and crop protein production per ha of arable land used for livestock or plant production. Source: figures are reworked data from the Eurostat database (Eurostat, 2018j) and refer to either the year 2016 or 2017 (latest available data).

In figure 1 the animal and plant-protein production has been expressed as kg plant or animal protein produced per ha of arable land used for either livestock production or plant production. Note that arable land is defined as land which is ploughed or tilled continuously, usually included in a crop rotation, and does not include permanent grassland (Eurostat, 2017f). 60% of arable land for SE, is assumed to be used for producing feed for livestock, and 40% for producing crops for human consumption. Shares for the other countries can be found in Appendix 1 as well as how the assumptions have been made.

When expressed in protein production per ha it becomes clear which countries have a high production per area of land used for producing food. NL stands out as the country with highest both animal and plant protein production, as it is a country with a small UAA, but high production figures. Most of the countries are more efficient at producing plant-protein per ha than animal protein per ha, with the probable explanation that crops which are directly used for human consumption do not undergo the energy losses coupled to the conversion of animal feed to milk and meat (Lesschen *et al.*, 2011). In IE, NL and NO the production of animal-protein is higher than the plant-protein. For IE this can be explained by the small number of land used for producing crops for human consumption, while cattle (dairy and non-dairy) production make up a significant share of the total production of agricultural products (table 11). For NO, a short length of growing period and various soil quality for crop production, might explain the low plant-protein production per ha. Grass production and grazing livestock are thus important constituents of the NO agriculture (Lombnaes *et al.*, 2011). For NL both the animal and plant-protein production is high. The high animal-protein production can be linked to a high LSU of 3.6 LSU/ha of UAA, indicating an intensive livestock production (figure 2).

4.4 Livestock density

The indicator livestock density (LSU/ha of UAA), gives an estimation of the intensity of the livestock production of a country. A high livestock production and a relatively small area of UAA will result in high LSU/ha. This is the case with NL, the country with the highest LSU/ha in this study. The size of UAA of NL is about half that of SE, still NL produces about five times the amount of raw milk, and ten times the amount of pig meat (table 11). DK has the second largest LSU/ha of UAA (1.58) and a similar size of UAA as SE, but about six times larger pig production measured in tonnes of produced meat (table 11). A high LSU/ha of UAA is linked to a high national output of animal protein per ha.

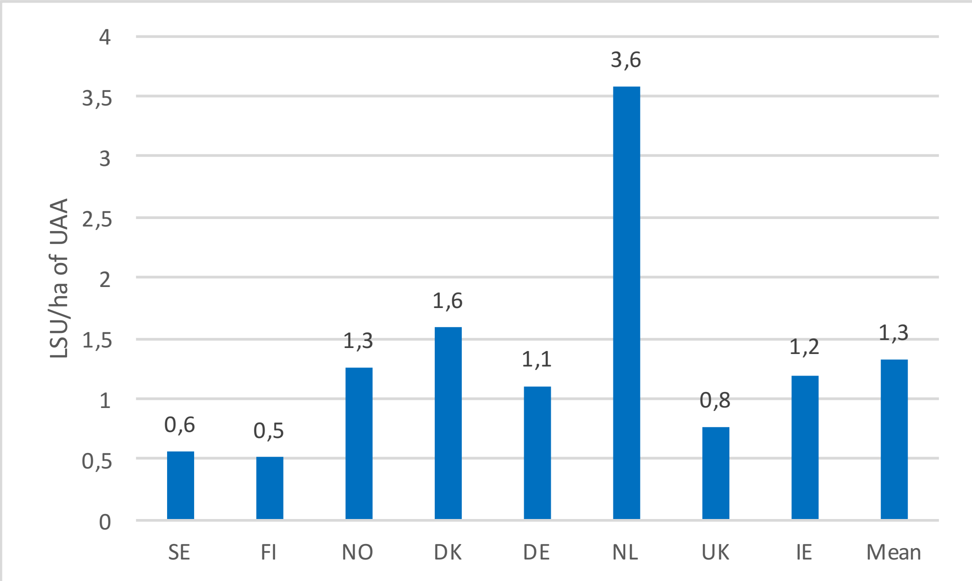

Figure 2. Livestock units/ha of Utilized Agricultural Area (LSU/ha of UAA). Source: Eurostat Agri-environmental indicators (Eurostat, 2018d).

4.5 Mineral fertilizer use

The national input of mineral fertilizer per ha of UAA can indicate a risk of nutrient loss to the environment, but also nutrient depletion of the soil if yields exceed the nutrient inputs. Nutrients are also applied in organic form, commonly as manure (figure 4 and 5).

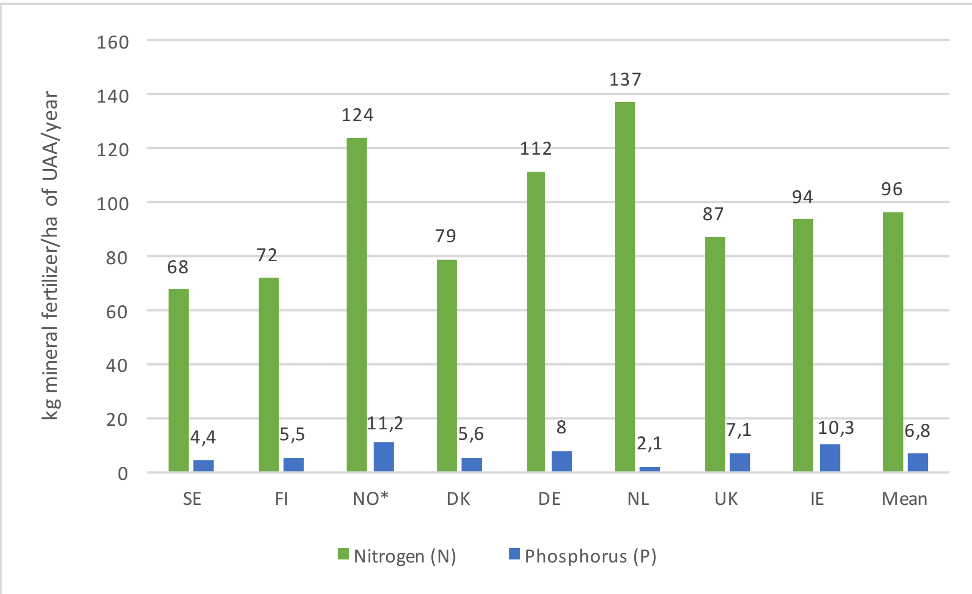

Figure 3. Mineral fertilizer application. The amount of nitrogen (N) and phosphorous (P) mineral fertilizer input per ha of UAA and year. Fallow land and rough grazing have been excluded from UAA. Source: Data was obtained from Eurostat and refers to the year 2015 (Eurostat, 2017b). *As data was lacking for Norway (NO) calculations where made based on Eurostat data on total mineral fertilizer use and FAO data on land use.

NL, NO and DE are the countries with highest mineral N inputs, exceeding 100 kg N/ha. This does not necessarily mean that the nutrient losses are higher from these countries, if nutrients are removed from the soil through harvest. The figures in figure 4 should therefore be complemented with data on nutrient leaching (for phosphorus), ammonia emissions and GHG emissions (for nitrogen) or gross nutrient balances, which account for the nutrients being removed from the field through harvest. NL is the country with the lowest input of mineral P, a possible explanation could be found in figure 6, where it shows that NL is the country with the highest P input from manure, which could further be linked to the high livestock density found in NL. NO has the second highest mineral N input, and highest mineral P input. A possible explanation is that NO has a low price ratio of N fertilizer to yield value, as cereal prices are twice as high as for EU; a result of NO agricultural policy (Korsaeth & Riley, 2006).

SE is the country with lowest application of N and second lowest application rates of P. There are several possible explanations, SE is the country with highest share of organic farming (18%) of the compared ones, land which receives non-significant amounts of mineral fertilizer (Eurostat, 2017i). Further, SE has a high share of grassland and pastures and both national (zero eutrophication) and international environmental policy (Baltic Sea Action Plan) which aims at reducing and making the use of fertilizers in agriculture more efficient through various measures aimed at farmers (SCB *et al.*, 2012). No relationship was found when plotting Eurostat data on farm structure (share of holdings > 50 ha) and data on national mineral fertilizer consumption.

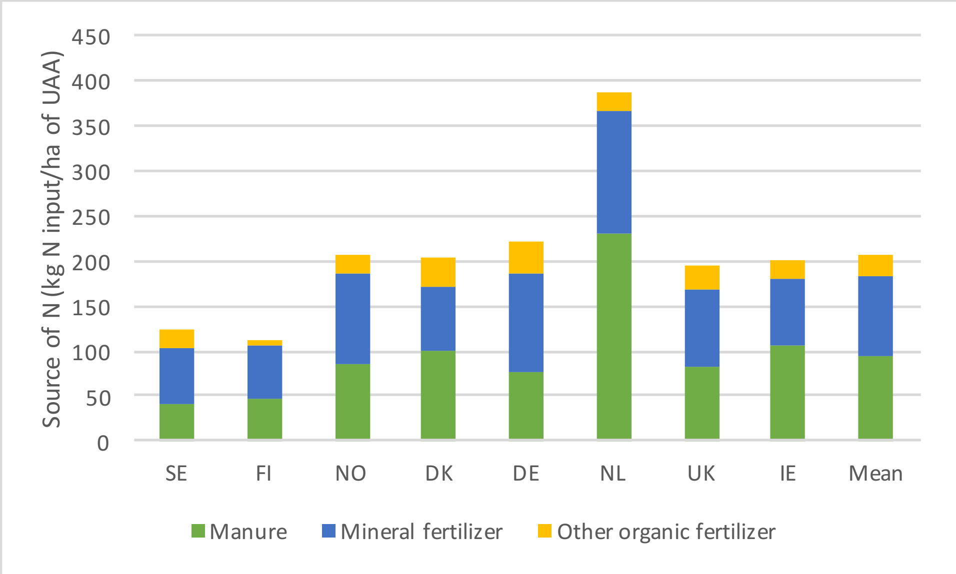

Figure 4. The origin of nitrogen applied to UAA. Source: Eurostat database, with data from the latest available year (from 2013 and onwards) (Eurostat, 2018j).

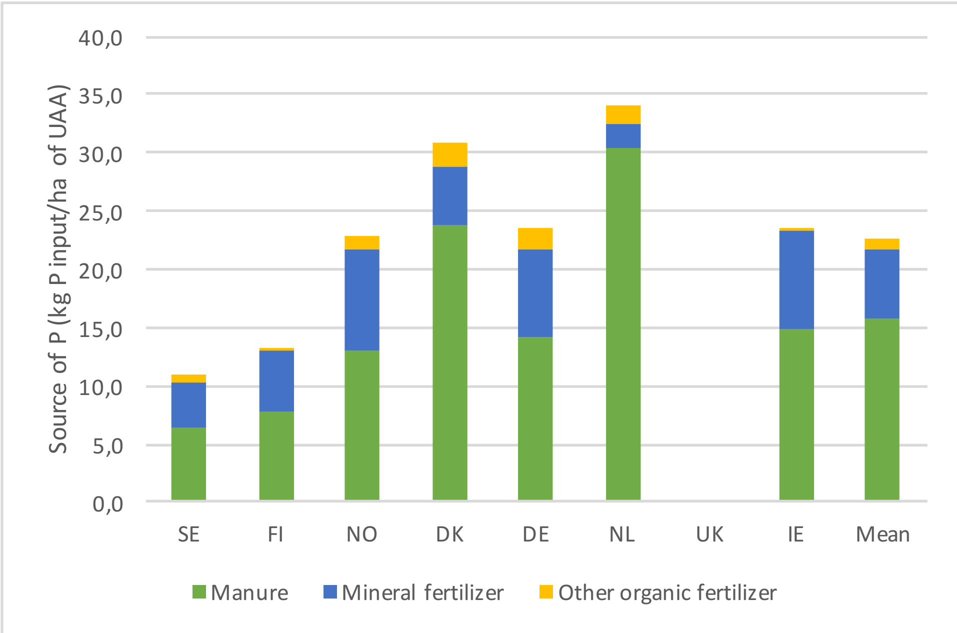

Figure 5. The origin of phosphorus applied to UAA. Source: Eurostat database, with data from the latest available year (from 2013 and onwards) (Eurostat, 2018j). Data for UK is not available.

It can be noted that for all the countries, manure is an important source of P, which makes up the largest share of P input, compared to N where N from mineral fertilizer generally makes up the largest share of N input.

4.6 Gross N and P balance on agricultural land

The *Gross N and P balance* gives further indication of how efficiently nutrient inputs to agricultural land are used, and the risk of the nutrients polluting the environment, compared to the indicator *Mineral fertilizer use*. In figure 7 the gross N and P balances for the year 2015 are presented, according to the Land Budget, described in detail in chapter 3.4. Countries with hight input of N from mineral fertilizer (figure 3), coincide with high gross N balances for all countries except IE (figure 6).

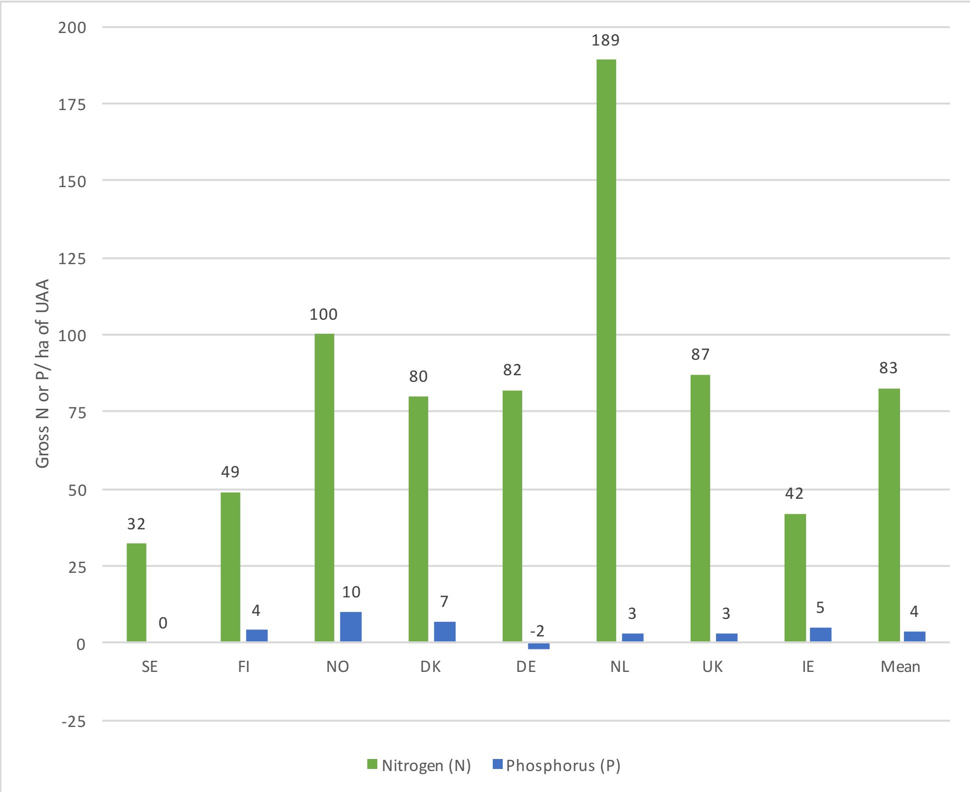

Figure 6. Gross N and P per hectare of UAA, by the Land Budget. The figure shows the excess nutrients when total outputs have been subtracted from the total inputs of N and P. Source: Eurostat data for the year 2015 (Eurostat, 2016b).

A similar relationship between gross P balance and P input could not be found as for N. This would be explained by the different behaviour of P in soil, which is not as mobile as N, but can rather be adsorbed to soil particles and released occasionally (see chapter 3.4 and 3.6 for more details on difference on N and P behaviour in soil). As animal manure is an important source of P it could be anticipated that countries with the highest animal densities would also have the highest gross P balances, however this seems not be the case (figure 5 and 6).

The gross nutrient balance does not reveal how the surplus nutrients are dispersed in the environment. Figure 7 presents the distribution of N, and indicates that a large share of surplus N is emitted to the atmosphere as N_2, and would indicate an economical loss for the farmer rather than an environmental pressure, as N_2 is naturally the most abundant molecule in the atmosphere. N_2O emissions generally make up only a small fraction of N loss, but has a significant environmental impact as it is a potent GHG. NH_3 emissions make up the second largest post (except for NL). Leaching and runoff make up the smallest share. A higher nutrient surplus in the balance seems to be increasing the risk of N lost as NH_3, N_2O and by leaching.

Furthermore, it should be noted that gross N balance has been reduced for all the countries between 2001-2003 and 2015, when comparing figure 9 and figure 7, except for UK where it has increased from 75 kg N/ha to 87 kg N/ha. This development would be in line with the evaluation of the impact of the Nitrates Directive, which has reduced the N loss from agriculture in Europe since its implementation in 1991, mainly by decreasing inputs of manure and mineral fertilizer through various policies, such as the manure application limit of 170 kg N/ha, indirectly reducing livestock numbers (Velthof *et al.*, 2014). As P does not occur in gas form, most of surplus P will be stored in soil stocks, or lost through runoff or leaching, thus a source allocation has not been done for P as for N.

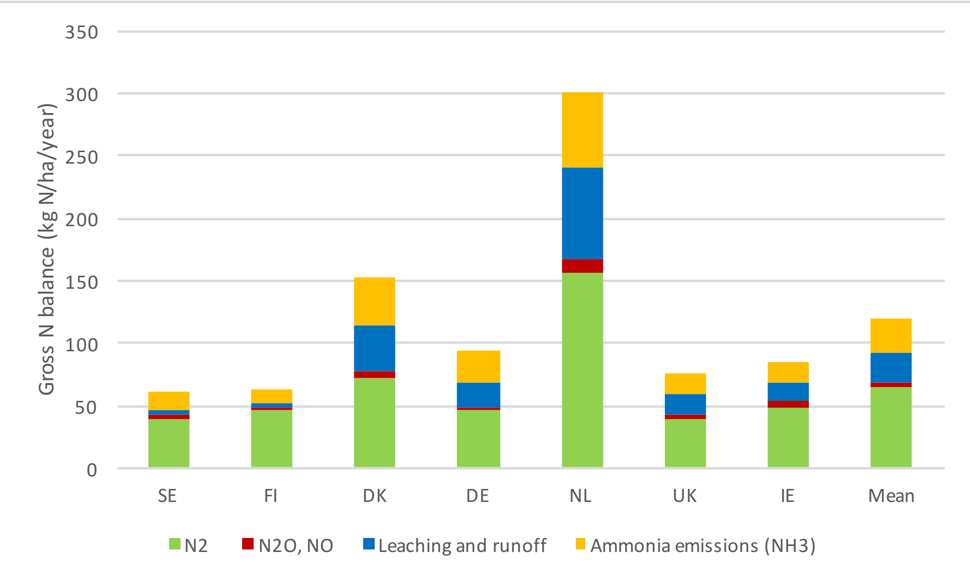

Figure 7. The distribution of surplus N from agricultural land when calculated according to the Land Budget, referring to the years 2001-2003. The post "Leaching and runoff" refers to N loss both from soil and from livestock (housing and manure). Leaching makes up the biggest share of this post. The post "Ammonia emissions" refers to NH_3 lost both from soil and livestock (housing and manure). Soil stock changes have been set to zero. Source: Reworked data from Leip et. al (2011). Only EU Member States are included.

It should be noted that the results in figure 7 are obtained by a different method compared to the gross nutrient balance in figure 6 and seems to consequently be underestimating the leaching from agricultural land when compared to data on leaching in figure 8. SE has a N leaching of 19 kg N/ha, but in figure 7 it seems to be about 5 kg N/ha. DK has a N leaching of 61 kg N/ha, compared to about 30 kg N/ha in figure 7. However, data seems to be overlapping quite well regarding the order of the countries in the gross nutrient balance in figure 6. The gross N balance for all countries appears to have decreased since 2001 (compare figure 6 and 7), which would be in line with the estimated reduced surpluses in the balance since the implementation of the Nitrates Directive (Velthof *et al.*, 2014).

4.7 Leaching of N and P

Nutrient leaching from arable land is commonly expressed as kg nutrient/ha. To link leaching of nitrogen (N) and phosphorus (P) to food production, leaching per ha (figure 8) has been calculated to leaching per produced kcal and protein (figure 9 and 10). The trend is that countries with high production of animal or plant protein/kcal, results in low leaching, proving the importance of functional unit (Salou *et al.*, 2017).

Leaching of N is both coupled to natural conditions, primarily soil type and precipitation patterns and farming practices, were important factors are which crops are cultivated and fertilizer regime. N leaching is more severe from sandy soils, than clay soils. This could partly explain the high leaching figures, for DK, DE and NL which have a high share of sandy soils (Ballabio *et al.*, 2016). For UK, IE and NO, high N leaching might partly be explained by high precipitation (> 1000 mm/year) (table 11). For UK and NO, the high precipitation in combination with high gross N balance (figure 6) might further explain the N leaching figures. SE and FI stand out as countries with lowest N leaching. In contrast with NO, with high leaching, which was indicated by a high mineral fertilizer use (figure 3). This could both be due to agricultural practices, as SE and FI have the lowest gross N balances, indicating that N is used efficiently, as well as smaller shares of sandy soils. Further, SE and FI cultivate almost no maize, and rather small shares of arable land is used for growing potatoes and vegetables, which are crops coupled to high N leaching.

P leaching is primarily influenced by natural conditions such as soil type and hydrology, and P-content in soil (influenced by farming practices). Clay soils are known for leaching P, and the highest share of clay soils will be found in UK, which has a high P leaching. DK, with the lowest P leaching, might partly be explained by a small fraction of clay soils. High P-leaching figures for NL could be explained by high input of P from manure (figure 5), and clay soils found in the southwestern parts of the country. SE has clay rich soils in the area Mälardalen, and clay soils make up a large part of southern FI (Ballabio *et al.*, 2016).

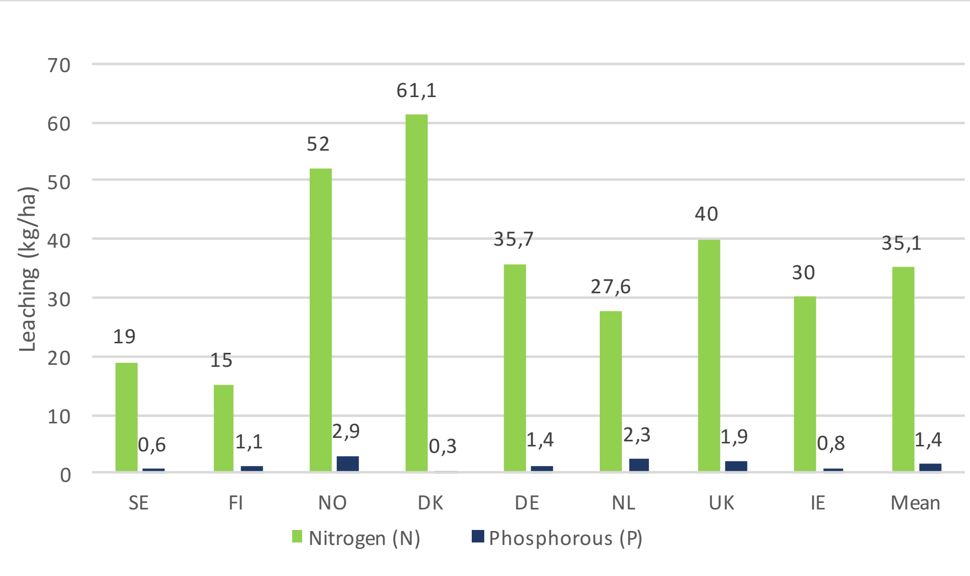

Figure 8. Nutrient leaching from arable land expressed as kg N or P per ha and year. The figures originate from both simulations and monitoring, depending on the source. Sources (when there are separate sources for N and P, N is given first within the bracket): SE; (SMED, 2013), FI; Huttunen (2018)[1], NO; (Bechmann *et al.*, 2017), DK; (Hutchings *et al.*, 2014; Klinglmair *et al.*, 2015), DE; (Behrendt *et al.*, 2003; FEA, 2014), NL; (PBL, 2017), UK; (Lillywhite & Rahn, 2005; White & Hammond, 2006), IE; Mellander (2018)[2].

In figure 9-10, leaching of N and P has been expressed as g N and P/kg of Animal-Protein, Plant-Protein and per 10 000 kcal of Animal kcal and Plant kcal. Thus, countries with high production figures per ha (figure 1) will perform better than countries with low production figures. Figure 8 and 9 seems to be the invert of figure 1; countries with high Animal-protein production per ha, have low leaching per kg Animal-protein, whereas a low Animal-Protein production results in high leaching per kg Animal-Protein. The same is true for N leaching/kg Plant-Protein. IE and NL are the countries with highest Animal-Protein production per ha (figure 1), and consequently have the lowest result for leaching when expressed as g N/kg Animal-Protein (red staple figure 9). NO stands out as the country with highest N leaching per produced kg of plant protein, explained by low plant production figures (figure 1)(note: the total national animal-protein figure is about 3 times higher than the plant-protein production), in combination with high leaching figures (figure 8).

[1] Markus Huttunen, Hydrologist at the Finnish Environment Institute. Communicated 2018-02-26.
[2] Per-Erik Mellander, Senior Research Officer in Catchment Science, Department of Environment, Soils and Land use. Wexford, Ireland. Communicated 2018-04-16.

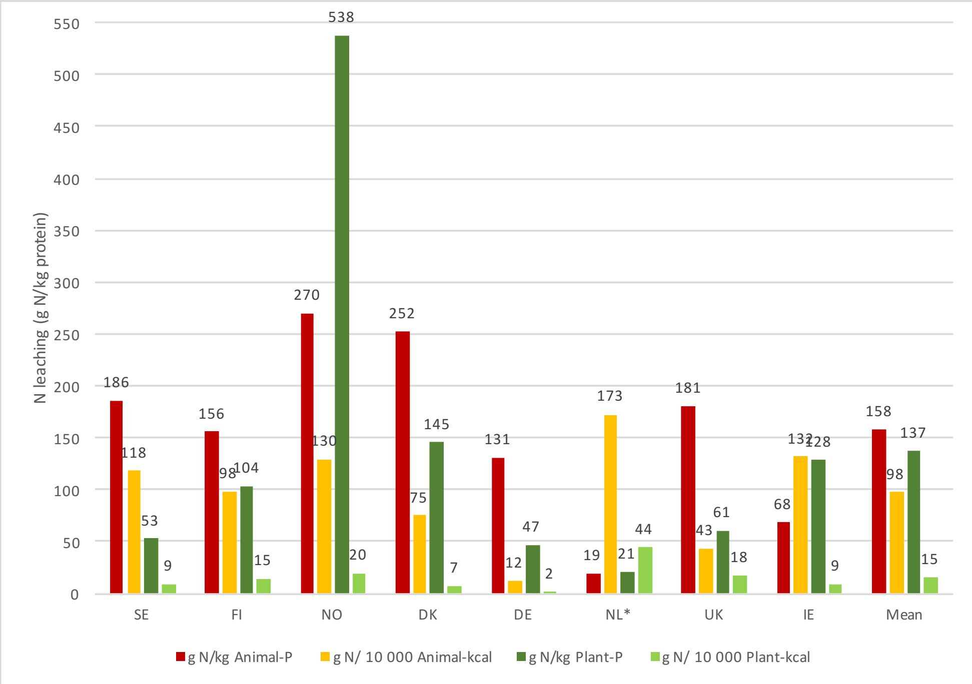

Figure 9. Nitrogen leaching per produced quantity of animal or plant products, expressed as either protein or calories (kcal). Source: Calculations based on data from figure 1 and 8. See Appendix 1 for methodological details.

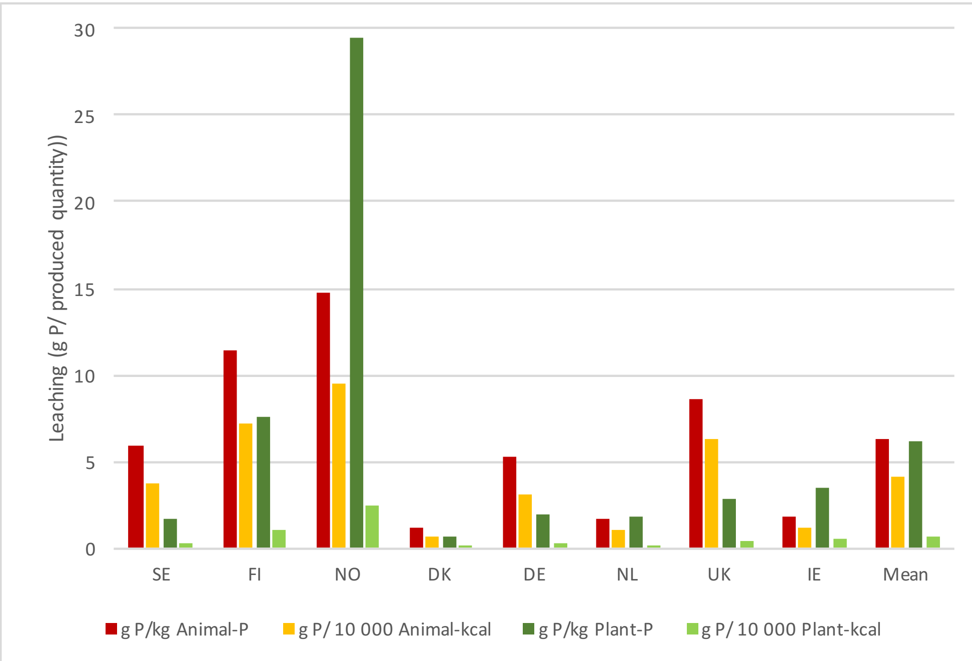

Figure 10. Phosphorus leaching per produced quantity of animal or plant products, expressed as either protein or calories (kcal). Source: Calculations based on data from figure 1 and 8. See Appendix 1 for methodological details.

The results from figure 9 and 10 are in line with the findings of Salou et al. (2017). Their report concluded that intensification of dairy systems had increased environmental impact for all impacts, when expressed per ha. When expressed per mass unit, there was a decrease in the environmental impacts eutrophication and area of land needed for production of one unit of milk.

4.8 Ammonia emissions

As discussed in chapter 3.3 agriculture represents 94% of total ammonia (NH_3) emissions in EU, of which livestock excreta contributes to 80-90% of the total agricultural emissions. In figure 11, NH_3 emissions are presented as kg NH_3/ha of UAA. NL has the highest emissions per ha of UAA, correlated to high livestock densities (figure 2). FI, SE and UK are the countries with lowest livestock densities, and have the lowest NH_3 emissions per ha of UAA.

Figure 11. NH_3 emissions from the agricultural sector, expressed as kg NH_3 per ha of UAA. Source: Eurostat, referring to NH_3 emissions for year 2015 reported under the National Emissions Ceiling Directive. Total NH_3 emissions have been divided by ha of UAA (Eurostat, n.d.).

In figure 12-15 NH_3 emissions are presented per ton of produced food. Generally, the production of beef meat is the most emission intensive, followed by production of pig meat > poultry > milk, when expressed per produced quantity. The results have been calculated using official emission figures, reported by countries under the EU National Emission Ceilings Directive for certain pollutants (NEC). The emissions are calculated by using standard Emission Factors (EFs) or using developed country specific EFs (see chapter 3.3 for further details). The sum of NH_3 emissions is reported for the categories "manure management – per animal category", "manure to soil" and "urine and dung deposited by grazing". These figures have been allocated to non-dairy cattle, dairy cattle, pig and poultry and distributed on the produced quantity of meat (beef, pig and poultry) and milk (Linderholm, 2017), presented as kg NH_3/ton of product in figure 12-15.

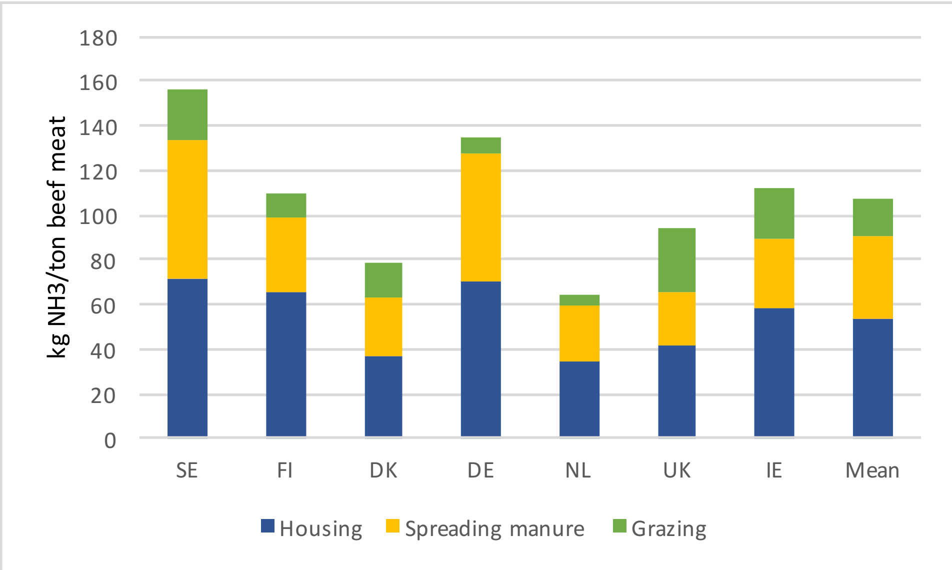

Figure 12. NH_3 emissions per produced ton of beef meat. Source: Linderholm (2017).

As seen in figure 13, SE is the only country where grazing contributes to $NH3_3$ emission from milk production. This is explained by the method of reporting NH_3 emissions and how the results have been calculated in figure 12-15. NH_3 emissions which arise from grazing are reported under the category "urine and dung deposited by grazing animals", which has been allocated to beef production for all countries, except SE (figure 12), as SE is the only country in EU with a general legal requirement for a grazing period for all dairy cattle, older than six months (LRF, 2013). For this reason, SE is the only country where a share of the emissions from grazing has been allocated to dairy cattle. Other countries might also have dairy cattle on grazing, but without legal requirements, or legal requirements coupled to type of housing. In e.g. FI, tethered dairy cattle are legally required to have access to outdoor recreation 60 days per year (LRF, 2013). Thus, a share of the green staple from figure 12 should most likely be allocated to figure 13 for all countries, which would reduce emissions for beef production, and increase emissions from milk production.

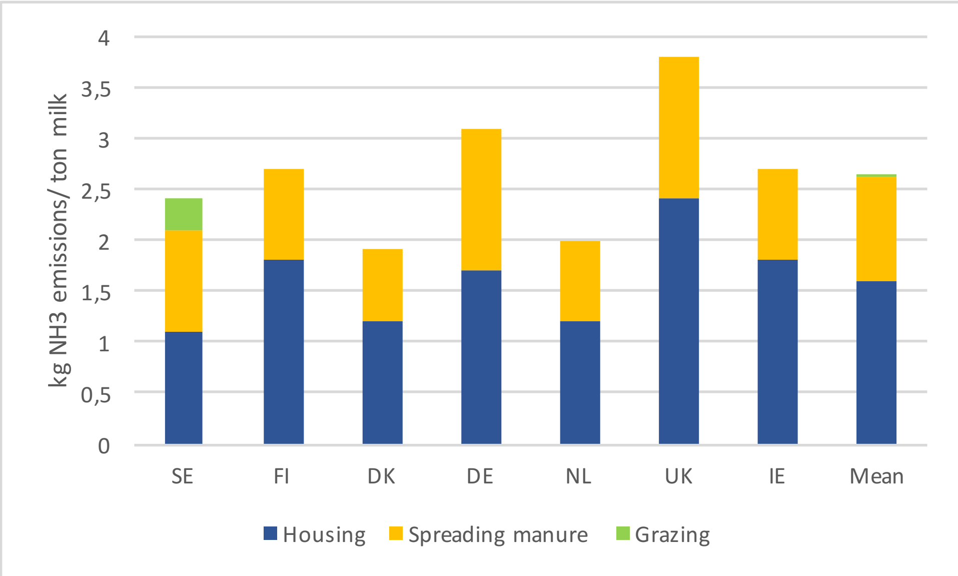

Figure 13. NH_3 emissions per produced ton of milk. Source: Linderholm (2017)

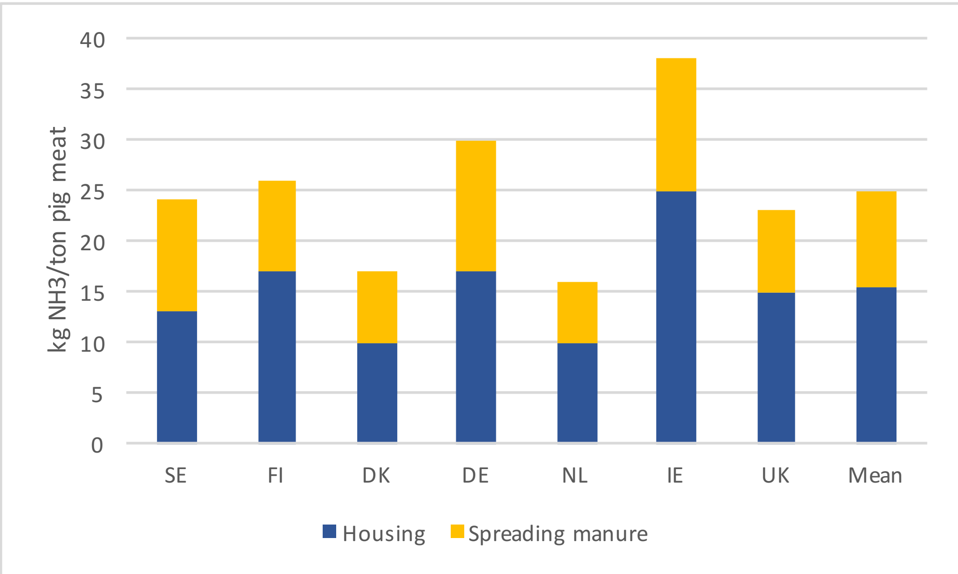

Figure 14. NH_3 emissions per produced ton of pig meat. Source: Linderholm (2017)

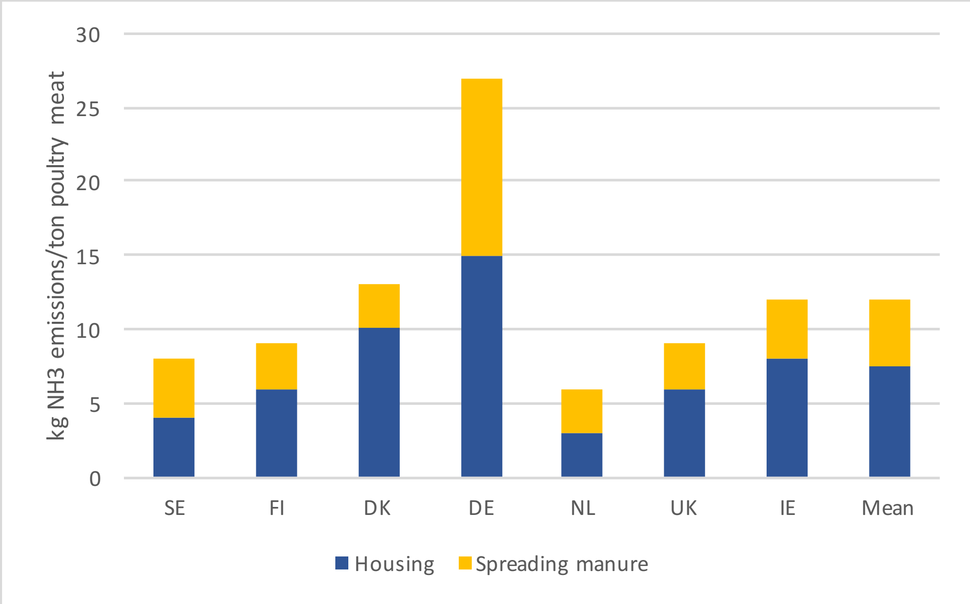

Figure 15. NH_3 emissions per produced ton of poultry meat. Source: Linderholm (2017)

According to figure 13-15 SE is performing relatively well in milk, pig and poultry production from an NH_3 emission perspective (being in line with or under the mean value). Figure 12 shows that SE has the highest NH_3 emissions out of the compared countries for production of beef meat. This cannot be explained by lower production figures, as SE and DK produce about the same quantity of meat from beef (table 11). Possible differences in production which explain this could be the acidification of manure in DK. In DK 18% of all liquid manure is being acidified, which has proven to potentially reduce NH_3 emissions up to 70% (Vibeke Vestergaard, 2015). Further, according to a Danish national emissions inventory report, the national NH_3 emissions from agriculture have been reduced by 38% since 1985 mainly due to improved feed efficiency and a transition to slurry injection instead of band spreading of manure (Mikkelsen Hjorth *et al.*, 2011). Another possible explanation for high emissions in SE could be the common use of deep litter housing in SE, which has a high EF, concluded from discussion with Knut-Håkan Jeppsson[3].

A possible explanation for the overall differences between countries would be the impact of using different EF, as it is known that EFs can have high influence on the results (Velthof *et al.*, 2012). Countries have the option to use either pre-set NH_3 emission factors (EF) according to EEA guidelines, or produce country specific EFs, which might affect the outcome of the results. Countries known to have put a lot of work to develop their own country specific EFs are DK and NL (Jeppsson, 2018).

[3]Knut-Håkan Jeppsson, Researcher at the Institution of Biosystems and Technology, Swedish University of Agricultural Science, Alnarp. Communicated 2018-05-05

This is however something that would require additional investigation to be able to draw further conclusions. NL is the country with lowest emissions overall per produced product (figure 12-15), but the country with highest emission per ha of UAA (figure 11), probably explained by a high production on a small surface.

4.9 GHG emissions from agriculture

Greenhouse gas emissions (GHG) and associated climate change, is one of the biggest threat to modern society (Rockström *et al.*, 2009). This has been the driver for implementation of the Kyoto protocol, binding all its parties to set national GHG emission reduction targets, and report on national GHG inventories to UNFCCC. In figure 16 the GHG emissions from the agricultural sector are presented as CO_2 equivalents/ha of UAA, for year 2015. The agricultural sector does not include emissions from energy use and transport, nor emissions associated to land use and land use change. The main sources for GHG emissions from the agricultural sector are thus CH_4 and N_2O, derived from enteric fermentation from ruminants and emissions from soil, respectively (see chapter 3.6 for more details).

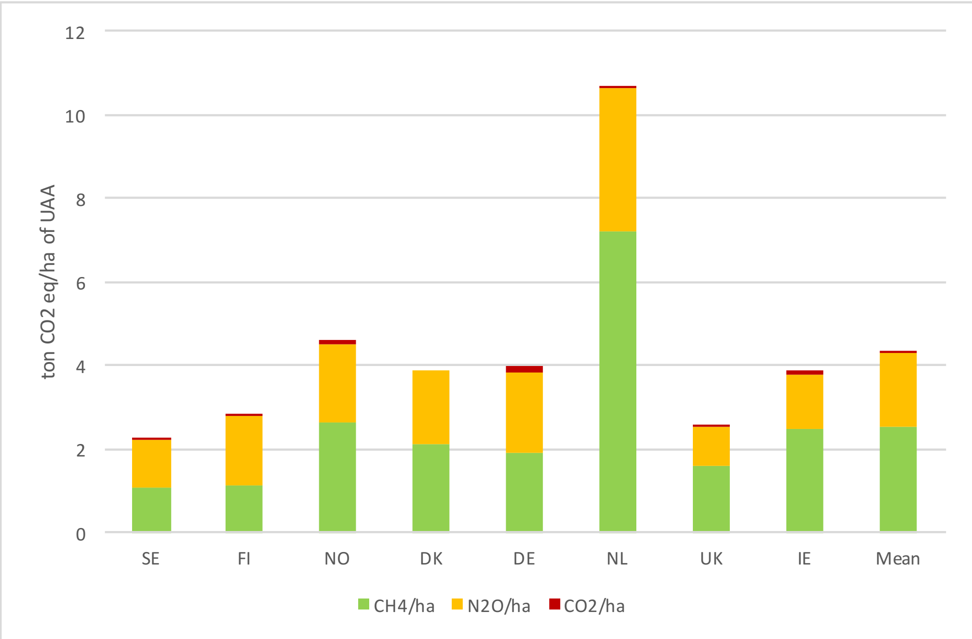

Figure 16. GHG emissions from the agricultural sector (CO_2 equivalents/ha of UAA). Source: Data for ha of UAA are obtained from Eurostat and referring to year 2013 (Eurostat, n.d.). Data on GHG emissions refers to year 2015 and is derived from national reporting to UNFCCC (EEA, 2017a)

Countries with low CH_4 emissions (green staple) (SE, FI and UK) correlates with low livestock densities (compare with figure 2). Enteric fermentation by ruminants is the main source of CH_4, which would also explain why NL has the distinctively highest result. N_2O emissions (yellow staple) correlate well to gross N balance per ha of UAA (figure 6). Countries with high gross N balances, have higher N_2O emissions (NL, NO, DE, DK), which would be in line with the strong correlation of excess N in soil and N_2O emissions. The exception is UK which has a high gross N balance, 87 kg N/ ha of UAA, but the lowest N_2O emissions (0.93 ton CO_2 eq/ha of UAA). FI, SE and IE have the lowest gross N balances and the lowest N_2O emissions per ha of UAA. To give further explanation of the results, requires more detailed knowledge about agricultural practices in the compared countries.

In figure 17-18 emission per kg of food product, obtained by FAO, are presented as CO_2 equivalents. However, it should be noted that these results vary greatly from other reports on GHG emissions per kg product (Lesschen *et al.*, 2011), emphasizing the impact of different methods for this type of calculations. FAO emission figures seem to be consequently 2-3 times lower than emission figures reported by Lesschen et. al (2011). A probable explanation would be that FAO considers the emissions coupled to excreta and enteric fermentation (emissions from manure management systems, application of manure from soils and manure left on pastures and emissions from enteric fermentation), thus the emission associated to the production of feed required for livestock seems not to be included (FAO, 2018b). However, further research in the methods used, is required to draw accurate conclusions.

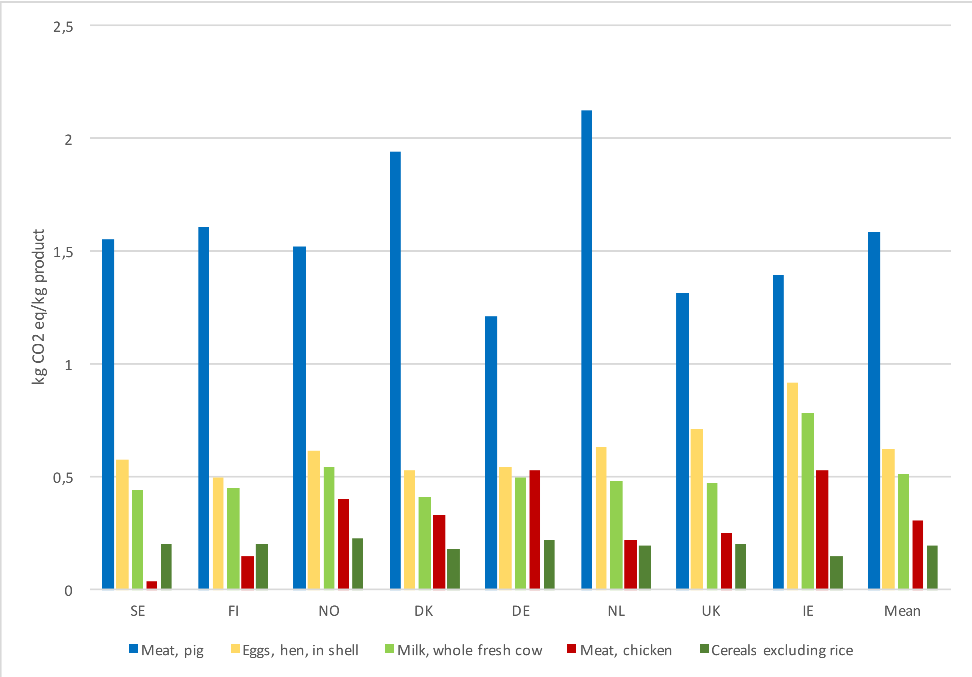

Figure 17. Kg CO_2 eq/kg product. Source: FAO, 2018b, referring to year 2016.

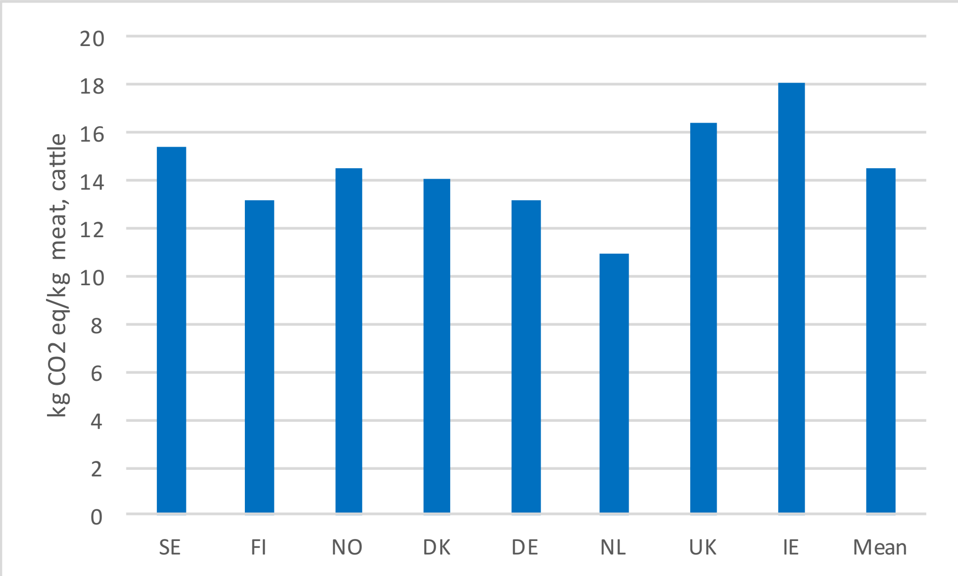

Figure 18. Kg CO_2 eq/kg of cattle meat. Source FAO, 2018b, referring to year 2016.

4.10 Pesticide use

The pesticide use of a country indicates the risk of environmental pollution of pesticides to the surrounding water bodies. No comparable data could be found on national level on the occurrence of pesticides in water bodies, to validate this indication. However, data is presented on the pesticide residues found in food products, from domestic production, and exceeding maximum residue levels (MRL). No correlation was found between the countries with highest application rates of pesticides and residues of pesticides in domestic food products (figure 19). Overall MRL values are low for all countries, explained by solid control programs with frequent sampling for residue levels in food products, concluded from discussion with Jan Eksvärd[4].

SE, FI and NO have the lowest application rates of kg active ingredient/ha. A part of the explanation is climate, as colder climate reduces the pressure of insects and fungi. SE and FI have high shares of ley, which does not have a high pesticide requirement. Further, NO has a high share of small-scale farming, contributing to lower pesticide use (Eksvärd, 2018). DE, UK and IE have long growing periods, which increased the pressure of fungal disease and insects.

The amount of organic production, as this production form utilizes less pesticides and relies more heavily on mechanic weed control, could also play a role, as UAA under organic production is included in the denominator for this indicator.

[4]Jan Eksvärd, Senior Expert, Sustainable Development, LRF. Communicated 2018-05-15.

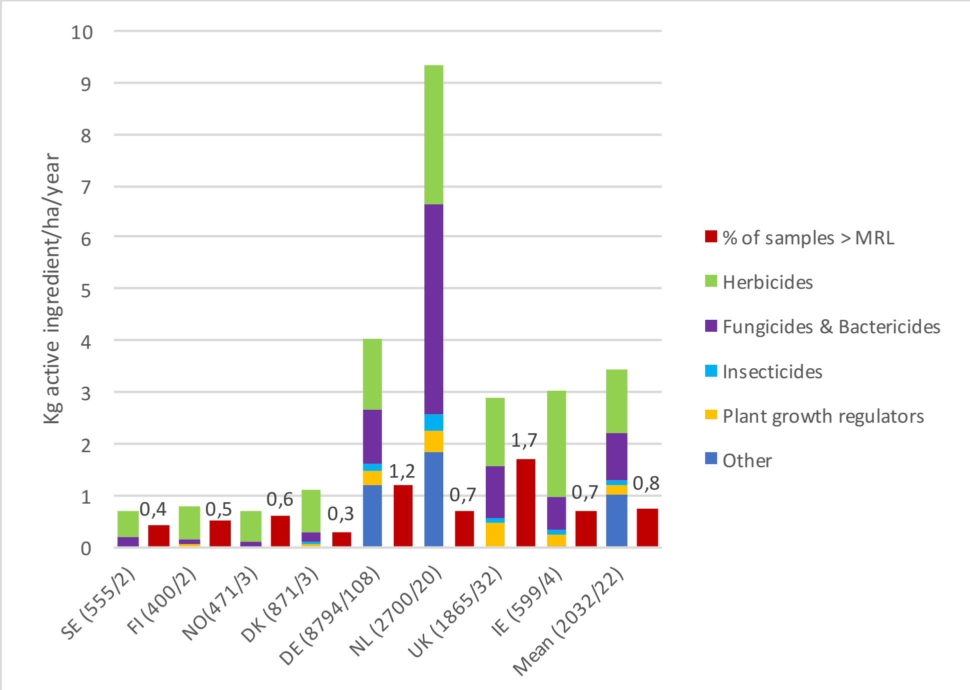

Figure 19. Pesticide use. Use of pesticides per ha of cropland (arable land + permanent crop) and % of samples exceeding maximum residue levels from the EU-coordinated control programme and results of national control programmes on pesticide residues in domestic food products. Numbers in brackets present total number of samples/number of samples > MRL. Both stacks refer to the year 2015. The category "other" includes inert gases for DE and soil disinfectants for NL. Source: Pesticide use (FAO, 2016c) and MRL exceedance rates (EFSA, 2017).

Difference in agricultural structure and cropping patterns probably explain much of the differences between the countries. Crops vary in their susceptibility to plant pathogens and insects. Wine production requires high doses of fungicides, as well as potatoes, while sugar beets require high doses of herbicides (SCB, 2010). SE, NO and FI use a small share of their UAA for growing potatoes and sugar beet (even though it might be a significant part of the countries production). DK has the double size of ha used for cultivating potatoes. DE is the only country with a significant wine production (99 000 ha). DE, NL and UK have high production of potatoes and sugar beet (Eurostat, 2018j.). Furthermore, the figures on pesticide use includes pesticides used for cultivating vegetables and fruits, which usually require high doses of herbicides and fungicides (SCB, 2010). This explains the high figures for NL, which has a high production of field vegetables, greenhouse vegetables and potatoes on a small area (Eksvärd, 2018).

4.11 Nitrate pollution of ground water

Nitrate is an important pollutant of ground and surface water. The threshold level for ground, surface and drinking water is 50 mg NO_3^-/l, where concentrations above 25 mg NO_3^-/l indicate a level of concern. According to figure 20, DK, DE and NL are the countries with highest share of monitoring sites exceeding the level of concern. This is in line with figure 9, showing the share of excess N which is lost through leaching and runoff and where NL, DK and DE show the greatest N loss by leaching and runoff. It could be expected that NO, which has high N leaching rates, should have a higher rate of monitoring sites exceeding the thresholds. There are high variations in number of monitoring sites among the countries. NO had 60 monitoring sites and UK had 2150 monitoring sites in year 2012.

Further, little is known about the location of the monitoring sites, except for SE, where monitoring sites are located in areas unaffected by human activities (SGU, n.d.). This implies that the exceedance rate would likely be higher for SE if monitoring was conducted in areas affected by agricultural activities. Furthermore, the concentration of nitrates found in the groundwater is highly influenced by soil type, and the nitrate leaching will be significantly greater from a loess or sandy soil than from a clay soil (Baumann *et al.*, 2012).

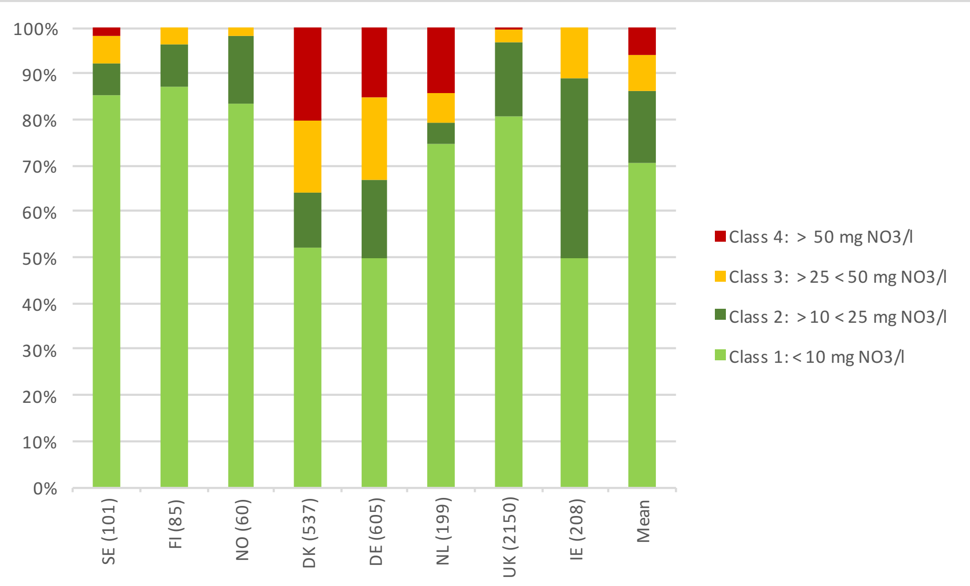

Figure 20. Nitrate in groundwater. Share of groundwater monitoring sites with nitrate concentration within the different nitrate classes. Numbers within brackets present the number of monitoring sites. Source: EEA member state reporting for year 2012 (EEA, 2015)

As NL is known for a shallow groundwater table, one could expect that NL would have the highest share of monitoring sites exceeding threshold levels. This is however not the case. A shallow ground water table could also increase denitrification rates, thus reduce N lost by leaching to groundwater (Baumann *et al.*, 2012), which could perhaps act as an explanation.

4.12 Soil erosion by water

On national scale, soil erosion by water (t/ha/year), is relatively low in the countries in this study, which are all well below the EU annual average soil loss of 2.67 t/ha/year from arable land. The mean annual soil formation rate in Europe is 0.3-1.41 t/ha/year. Erosion rates for all the countries exceed the lower soil formation rate of 0,3 t/ha/year (figure 21)(Verheijen *et al.*, 2009). A low soil erosion rate does not mean that the countries lack regional areas of concern, and might be losing soil at a higher rate than the soil formation rate in erosion prone areas. SE has a mean annual soil erosion rate of 1.21 t/ha/year, still soil erosion is an important source of particulate P loss to water bodies in certain areas.

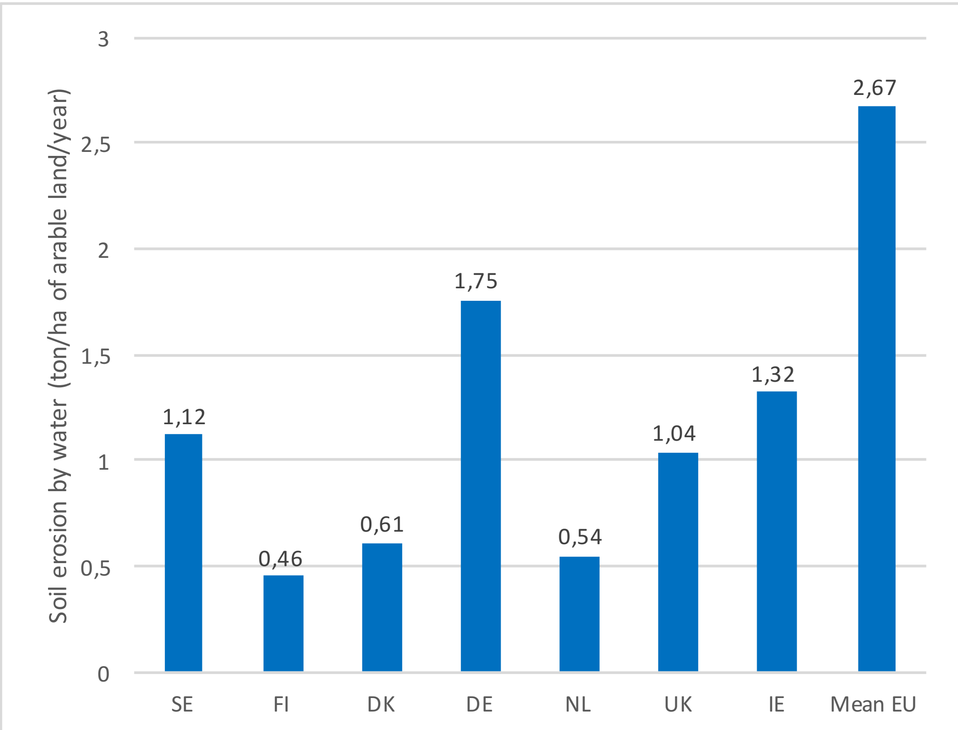

Figure 21. Soil erosion. National comparison of mean annual soil erosion rates by water on arable land. Average soil erosion from arable land in EU is 2,67 t/ha/year. Source: Data is reproduced from Panagos et al. (2015). Data for Norway is lacking as it is not an EU Member State.

Soil erosion by water is mainly decided by precipitation patterns, soil type and topography, therefore the countries with highest national mean annual erosion rates by water are found in other regions of Europe, then the countries compared in this study (i.e. around the Alps and the Mediterranean area) (Panagos *et al.*, 2015). However, the erosion rate is often higher on arable land than on other land, as erosion is also affected by land use and land management. For example, the mean soil erosion for all land in SE is 0.41, compared to 1.12 t/ha/year for arable land. Panagos et al. (2015) estimated that soil erosion rates by water could be reduced by approximately 20% when applying soil protection measures such as minimizing bare soil, conservation tillage and contour farming. DE is the country with the highest erosion rates by water on arable land, one part of the explanation could be the high share of arable land used for production of maize (18% in 2015). The increasing erosivity when growing maize is coupled to a short growing period, as maize is sown late in spring with wide interrow spacing, leaving the soil bare or poorly covered for a long period. Furthermore, maize often replaces grassland as it is commonly grown as feed for cattle, thus, often being sown on steep land, increasing erosion risk (Laloy & Bielders, 2010). It should be noted that the results presented in figure 21 are retrieved from a model (see chapter 3.9) which has received critique for under and over estimating actual erosion rates by water (Evans & Boardman, 2016).

4.13 Agricultural water withdrawal

Countries in southern Europe primarily experience issues with over exploitation of freshwater resources, however, as discussed in chapter 3.10, changed precipitation patterns and increasing periods of droughts are also raising concerns in western and northern Europe regarding sustainable use of fresh water resources for producing food (EEA, 2012, 2018b). In figure 22, the percentage of total national water use, used for agriculture is presented, as well as the percentage of UAA equipped for irrigation. The difference in share of agricultural water use of total national water use, can largely be explained by how much water is used for industrial and municipal purposes. The low share of agricultural water use of total water use in SE is due to a high industrial use of water, which is also the case for FI and NL, whereas DE has a high municipal water use, reducing the agricultural share of water use (FAO, 2016a). For SE, FI, NO, DK and UK irrigation is mainly used for potatoes, as this is an intensive crop, meaning that it is a high value cash crop where irrigation is profitable for maintaining yields and quality. For NL, which has the highest share of irrigable UAA, this can probably be explained by a high production of vegetables, usually classed as intensive crops (Eurostat, 2018c).

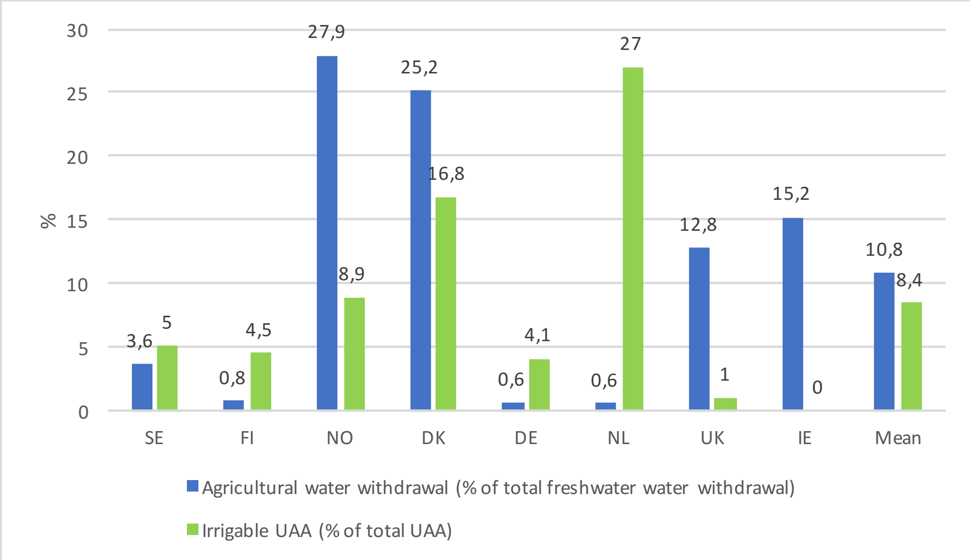

Figure 22. Agricultural water withdrawal. The % of total national water use, used for agriculture (including ground and surface water, as well as direct use of agricultural drainage water, desalinated water and direct use of treated waste water) and the % of total UAA equipped for irrigation. Actual irrigated area is usually significantly smaller than irrigable area (area equipped for irrigation). The irrigable area does not include ha under glass. Source: Agricultural water withdrawal referring to latest available years (1994-2012) (FAO, 2016a). Irrigable UAA, referring to year 2013, (Eurostat, 2016c).

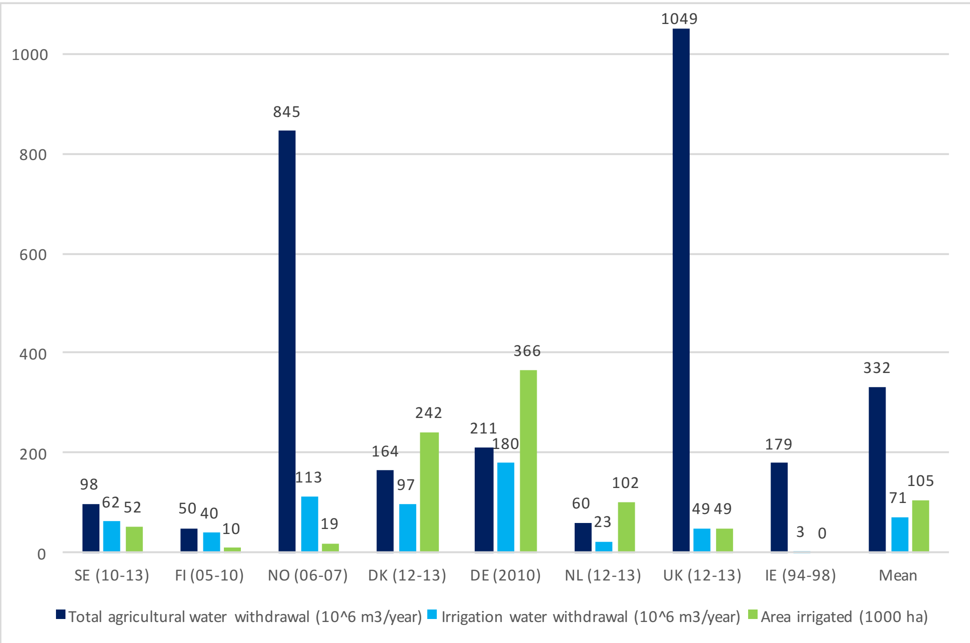

Figure 23. Total agricultural water withdrawal, withdrawal for irrigation and actual irrigated area. Figures within brackets represents the year of the data. Source: Latest values from FAO(FAO, 2016a).

In figure 23 it can be noted that NO and UK stand out as the countries with highest water withdrawal for agriculture. For NO this is explained by the inclusion of aquaculture in the statistics, which makes up 660 out of 845 million cubic meters of water (Undelstvedt *et al.*, 2008). No explanation has been found for the high share of agricultural water use for UK, where the agricultural water withdrawal is about the same size as the industrial water withdrawal.

Water withdrawal for agriculture is mainly used for irrigation as shown in figure 23. The second largest post is commonly livestock husbandry. In SE for example, in 2015, 64% of total water withdrawal for agriculture was used for irrigation and 36% for livestock husbandry (SCB, 2015c). Data in figure 23 is of various age, indicating that water reporting is not a main priority among the countries in this study, which could likely be explained by that these are all countries with sufficient water supplies.

5 Discussion

The aim of this report was to analyze the performance of Swedish food production, by doing an inventory and analysis of available agri-environmental indicators (AEIs) as well as to develop new AEIs which would better link agri-environmental performance to food production. The discussion is divided into three parts to simplify the interpretation of the results:

- Discussion of advantages and shortcomings of AEIs included in the study
- The performance of Swedish food production
- Evaluation of the new AEI “nutrient leaching/kg product”

5.1 Advantages and shortcomings of AEIs

An indicator should preferably be: easy to measure or obtain data for; easy to interpret; scientifically sound. With the work of this paper, it was discovered how these three objectives are not always easy to fulfill. Generally, the simplest indicators (livestock density, pesticide use and mineral fertilizer use), i.e. driving force indicators, rely on data simple to obtain such as sold/imported quantities or data obtained from farm structure surveys. This type of indicator is the most common. However, they cannot alone be used to explain environmental performance, as it is not necessarily true that a high use of pesticides or mineral fertilizers, or a high livestock density will lead to reduced environmental performance. Other factors, not included in these indicators, such as yield levels, agricultural practices, and handling of manure and natural conditions can have great impact. For Norway, a country with a high economical optimum value for N input, the mineral N fertilizer use, corresponded well to a high N leaching/ha. Denmark on the other hand, which has the lowest mineral N inputs, after Sweden and Finland, N leaching was non-correspondently high, explained partly by natural conditions (Denmark has a high share of sandy soils), but also by anthropogenic factors i.e. high N input from manure, combined with moderately high N input from mineral N fertilizer, resulting in a high total N input, compared to Sweden. These indicators, should thus preferably be complemented with indicators describing the state or pressure of the environment. For example, the Netherlands had a remarkably high livestock density, which corresponded well to high NH_3 and GHG emissions, but leaching figures were comparably low. The indicator *Gross nutrient balance*, which could be regarded as a more detailed and accurate indicator than *Mineral fertilizer use*, as it includes both nutrient inputs and outputs, also failed to predict leaching or emission of ammonia. It did, however, correlate well to emissions of N_2O (countries with high gross nutrient balance had high N_2O emissions).

The pressure indicators *NH_3 emissions* and *GHG emissions from agriculture*, are easy to obtain data for as there are international treaties requiring parties to compile and officially present data on emissions. These types of indicators are easy to communicate, as they present an actual pressure caused by agriculture. The downside of these two indicators is the methodology for calculating them. The methodology is developed as such that it should be usable for countries also with small resources, and allow for countries with greater resources to customize the method by developing country specific emission factors. This is essentially a good idea, but complicates the comparison of the results, as shown in chapter 4.8. Further, when GHG emissions are presented as kg CO_2 eq/kg product, the results can vary greatly depending on method and system boundaries, as discovered when comparing figures from FAO (figure 17 and 18) with Lesschen et. al (2011).

The pressure indicator *Agricultural water withdrawal*, illustrates how countries do not consequently collect or report data on environmental topics not regarded as relevant for a country. Some of the data for this indicator was more than 20 years old (data for Ireland was from the years 94-98). Furthermore, it was difficult to link the results of the indicator to agricultural practices, rather the results where explained by how much water was consumed by industry or municipal water use. However, as it is possible that water scarcity will increase in the future, water indicators will likely become more used and important also for north European countries. The indicator *Soil erosion by water* is relatively new and based on a specific model applied to all the countries, meaning that the methodology is harmonized. However, as is the case with many models, it has received critique for not overlapping with field conditions, illustrating the downside of models. On the other hand, models are very useful for systemizing calculations, and making the results easier to compare and interpret. According to the indicator *Soil erosion*, and to the Eurostat classification, all the countries in this study were classified as having low erosion rates, although it is known that erosion is an important pathway for P loss, and it is thus one of the factors contributing to eutrophication in the Nordic countries, and most likely also for other countries in the study.

The state indicators included in this review *Leaching of N and P* and *Nitrate pollution of groundwater*, directly reflect on the state of the environment, and how much of the actual nutrient surpluses are lost to waterbodies. This type of indicators would be more appropriate to use for policy making, or for setting agri-environmental goals. The shortcoming of these indictors is that they require environmental monitoring and models, which can both be expensive and time consuming to produce. Another shortcoming of environmental monitoring would be the impact of non-harmonized methodology on the results. As is the case with the indicator *Nitrate pollution of groundwater*, where the range of monitoring sites being included varied between 60 and 2150, in the different countries. The monitoring is integrated in already existing national monitoring programs, indicating that there might be variations between countries on how the sampling is conducted. E.g. in Sweden monitoring is conducted on waters unaffected by human activities. For the indicator *Nutrient leaching*, data was obtained from multiple sources, including oral communication from institutes and reports, with various methods for calculating the leaching, as there is no international harmonized methodology, reducing the comparability of the results.

Only two out of ten indicators in this study, *Nutrient leaching* and *Nitrate pollution of groundwater*, were state indicators, whereas most of the indicators fell under the category pressure and driving force. This complicates the evaluation of national environmental performance, as an indicator such as pesticide use should preferably by followed up by a state indicator, to prove causation. Even though state indicators can be found in the OECD and Eurostat indicator database, they often lack data, thus rarely making it possible to do national comparisons.

5.2 Performance of Swedish food production

What could be concluded from this report was that, on a national scale, Sweden was one of the countries, accompanied by Finland, that had the lowest environmental impact regarding nutrient leaching, ammonia emissions and GHG emissions, when expressed per ha of UAA (figure 8, 11 and 16). Most likely explained by lower nutrient inputs, a low livestock density, i.e. less intensive agriculture. It should be noted, however, that cultivation of organic soils, is a major emitter of GHG in Sweden, and this emission post is not included in the GHG emission from agriculture, but under the IPCC category Land Use and Land Use Change. A review of the magnitude of the other countries emissions from cultivation of organic soils has not been conducted in this study. Further, Sweden has one of the lowest uses of pesticides, most likely explained by what type of crops are grown in Sweden, i.e. the countries with high pesticide use have a larger production of vegetables and fruits for which the pesticide use is higher than for e.g. cereals. A high/low pesticide use could however not be coupled to high or low occurrence of pesticide residues in food products (figure 19). The Swedish performance for the indicator *Agricultural water withdrawal* was high and for the indicators *Soil erosion by water from arable land* the performance was average. Further, Sweden was one of the countries with a low share of groundwater monitoring sites that exceeded threshold levels of 25 mg NO_3-/l (figure 20). The exceedance rate would probably have been higher if monitoring was done in areas affected by agricultural activities, a result somewhat hard to explain as Sweden has amongst the lowest N leaching figures (figure 8).

When the functional unit was switched from land unit (ha) to mass unit (kg or kcal), the Swedish performance was impaired. This was true for the indicators *NH_3 emissions/ton of product*, FAO figures for *GHG emissions per kg product* and *Nutrient leaching/kg product*. One explanation are the low production figures expressed as kg protein/ha, especially low are the Animal-Protein figures. The low production figures might be a result of lower inputs, but also climatic constraints, structure of agriculture and to some extent country specific regulations such as the maximum allowance of applying 22 kg P/ha/year, indirectly regulating number of animals. Further the nominator in the equation for calculating leaching per kg product, was total nutrient leaching from arable land. This figure was calculated by multiplying ha of arable land with kg N or P leaching/ha. Consequently, a country such as Sweden, which has a relatively low production from a large area, will get high results (the Netherlands produces 10 more animal protein per ha, and the total arable land is half that size of Sweden's arable land). For NH_3 emissions, the results are more difficult to explain. Sweden has an average performance for milk, poultry and pig meat, and stands out with highest NH_3 emissions from beef meat out of the eight countries compared. As Denmark and Sweden produce about the same quantity of beef meat, a low production cannot explain the difference. Once again this might be a result of different emission factors, but also of the implementation of NH_3 abatement techniques, such as acidification of liquid manure, a common practice in Denmark. Further investigations would be needed to be able to explain the differences.

5.3 Evaluation of new AEI "nutrient leaching/kg product"

The developed agri-environmental indicator *Nutrient leaching/ kg product* in this thesis, expresses nutrient leaching in a mass unit instead of the more commonly used land unit. The switch of land based functional unit (ha) to a mass based functional unit (kg) favored countries with high production figures and gave radically different results from the indicator *Nutrient leaching/ha.* The performance of Sweden and Finland, which had the lowest N leaching/ha, was significantly reduced with this indicator, and Netherlands which has a higher leaching per ha, had by far the best results when leaching was expressed per kg Animal or Plant-Protein. The results are in line with other findings, that suggest that an increased environmental impact, when a result of intensification of agriculture (i.e. producing more per ha of land), can be masked by high production figures. However, the benefits of expressing environmental impact per mass unit, is the growing demand for food. The amount of produced food is therefore relevant, and should somehow preferably also be included in the total performance. Consequently, in a context of intensified agriculture, where it might be beneficial to express environmental performance in mass units, it should be complemented with indicators that express environmental impact per land unit, to ensure that the impact is within environmentally acceptable levels.

This reasoning can be recognized when discussing environmental benefits from organic agriculture. Organic agriculture, which has lower inputs, and often lower outputs, usually performs well when the functional unit is land based, whereas conventional agriculture (high inputs, high outputs) often benefits from the functional unit being mass based. However, regardless of production form, if the aim is to produce as much as possible, within environmentally accessible limits, both environmental impact and production should be regarded.

For this indicator to be useful on larger scale, additional development is required:

- For correct allocation of leaching load to animal and plant products, it would require country specific figures on the share of arable land used for producing plant products used for human consumption and the share of arable land used for producing animal feed. In this study 40% of arable land was allocated to the production of crop products for human consumption and 60% for animal feed (with some minor adjustments for countries known to use higher shares for producing animal feed). An improvement like this would most likely benefit plant products, as a higher share of the leaching load would be allocated to animal products, since the share of land used for producing animal feed likely exceeds 60% for many of the countries included in this report.
- National figures on leaching/ha from arable land where obtained from different sources, where different methods have been applied to calculate or estimate leaching, and different system boundaries have been used. Thus, there is high uncertainty coupled to these figures.
- Some of the protein and kcal conversion factors, such as for sugar beet and potatoes should be revised.
- More animal and crop categories could be included, e.g. meat from goat and sheep.

5.4 Conclusions

- The choice of functional unit is important as it can generate very different results. When the functional unit is mass unit (kg etc.) it generally benefits countries with high production figures. A high production figure can mask a high negative environmental impact. For demonstrating sustainable intensification, indicators expressed in mass units should be complemented with indicators expressing environmental performance in land units or total national emissions. Doing so will ensure that increased production figures are not on behalf of reduced environmental performance. However, the mass unit is likely easier to communicate to consumers
- Even if indicators with a functional mass unit are not suitable when evaluating environmental impact, they are still important, as the fact is that we will need to produce more food as the average daily calorie demand per capita is expected to continue to increase. Thus, a high production, within environmentally acceptable limits should be desirable.
- A shortcoming of indicators in general when doing national comparisons, is that regional variance which can be high, is ignored. This should be kept in mind when doing international comparisons. The indicators *Greenhouse gas emissions* and *NH_3 emissions* are most suitable on national scale as they relate to national emission reduction goals, although NH_3 emissions can have a high regional impact even if total national emissions are low.
- The results of the indicator *Pesticide Use* are rather explained by climatic conditions (length of growing period, pesticide pressure) and structure of agriculture (i.e. what crops are grown) than sustainable use of pesticides. This does however not mean that countries cannot impact their use of pesticides through for e.g. crop rotations. Lower overall use of pesticides will generate a lower pressure on the environment, and the structure of agriculture can be regulated so that the need for pesticides is reduced, e.g. by adjusted crop rotations, increased use of biological and mechanical plant protection methods etc.
- The indicator *Livestock density (LSU/ha of UAA)* correlates well with *NH_3 emissions (kg NH_3/ha).* The livestock density can thus be used as an indicator to predict NH_3 emissions, based on data for the 8 countries in this thesis. This statement should be validated by doing a comparison that includes more countries.
- The prediction of nitrogen leaching based on the indicators *Mineral fertilizer use* and *Gross nutrient balance* should be done cautiously. Even if high gross nutrient balance overlapped quite well with high leaching for some countries, it was not true for all countries. The Netherlands had by far the highest gross nutrient balance, not corresponding to its leaching figures which where average. This could be a result of the high impact of climate and structure of agriculture (i.e.

what crops are grown). P leaching could not be predicted by any of the indicators, most likely due to that P leaching is highly linked to natural conditions. The most important factors in this study which could indicate risk of P loss, would be livestock density and P input from manure, as this is the biggest source of P input.

- For an indicator such as *Soil erosion by water from arable land,* it can be difficult to make international comparisons, as regional variances can be high and a national figure might mask great issues mainly caused by erosion, such as P loss by erosion.
- As established in this thesis, driving force and pressure indicators are easier to obtain data for, and most indicators belong to these categories. State indicators are often more difficult and require rigorous work to produce. Driving force and pressure indicators will thus probably continue to be the most common indicators. As demonstrated in this thesis, the results of driving force and pressure indicators do not always correspond to the anticipated environmental state, as natural conditions might have high impact. Therefore, it would be important to recognize the correlation between pressure and state under different conditions. This could be based on knowledge obtained from monitoring, and generate correlation for different e.g. climate and soil conditions.

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Appendix 1

Table 12. *Input data. Data used for calculating the AEI "Nutrient leaching/kg product". ALAAP stands for arable land allocated to animal production. ALAPP stands for arable land allocated to plant production for human consumption.*

		SE	FI	NO	DK	DE	NL	UK	IE	Source
Arable land	(1000 ha)	2583	2223	808	2392	11876	1038	6269	1042	Eurostat (2013)
	ALAAP* (share)	0,6	0,57	0,57	0,57	0,57	0,57	0,65	0,72	Lesschen et al. (2011)
	ALAPP* (share)	0,4	0,43	0,43	0,43	0,43	0,43	0,35	0,28	Lesschen et al. (2011)
Production	(1000t)									
Plant	Cereals	5964	3398	13264	9872	45593	1344	22753	2240	Eurostat (2017)
	Oil seeds	285	95	11	506	4676	10	1823	33	Eurostat (2016)
	Pulses	195	65	n.s.	55	523	6	886	65	Eurostat (2016)
	Potatoes	1954	587	10772	352	363	861	5373	6534	Eurostat (2017)
	Sugar beet	1696	433	25497	0	n.s.	1988	5687	5502	Eurostat (2015)
Animal	Raw milk	2883	2390	1594	5385	31973	15090	14732	6851	Eurostat (2016)
	Pig meat	240	179	138	1544	5455	1456	900	294	Eurostat (2017)
	Poultry meat	158	129	98	160	1513	1036	1835	152	Eurostat (2017)
	Eggs, hen in shel	140	73	68	86	812	716	722	44	FAOSTAT (2016)
Leaching	kg N/ha	19	15	52	61,1	35,7	27,6	40	30	
	kg P/ha	0,6	1,1	2,9	0,3	1,4	2,3	1,9	0,8	

Table 13. *Conversion factors. Conversion factors used to convert production figures to kcal and protein.*

	Beef	Pig meat	Poultry	Milk	Eggs	Cereals	Pulses	Oil crops	Potatoes	Sugar beet
Protein (g/kg)	138	105	127	33	111	80	215	146	18	80
Kcal/kg	1513	2786	144	560	1425	3208	3412	2908	830	427
HC*						0,24				
Carcass to meat	0,7	0,78	0,75							

Demonstration of calculation of the values for the AEI "Nutrient leaching/kg product", as an example the method for calculating kg N/kg Animal-Protein and Plant-Protein will be used.

1. The first step was converting the production figures, for plant products respectively animal products, in table 12 using the protein conversion factors in table 13 for all countries. Total kg Plant-Protein was summed to a total sum of kg plant protein produced in each country, respectively total kg Animal-Protein produced per country.
2. In the second step, total arable land was allocated to either "Arable land allocated to producing plant products for human consumption (ALAPP)" or "Arable land allocated to producing animal products (ALAAP)". This was simply the share of arable land estimated to be used for either producing plant products for human consumption, or animal products, including land used for producing animal feed. The shares (see table 12) where multiplied with total ha of arable land per country. It was estimated that 40% of arable land was used for producing crops for human production (SBA, 2014), and this figure was somewhat adjusted to country specific values based on a map illustrating "feed related area (% of UAA)" found in Lesschen et al. (2011). This resulted in ha of ALAAP and ha of ALAPP.
3. In the third step, total kg N leaching from ALAAP and ALAPP was calculated by using data on N leaching (kg N/ha) and multiplying this figure with number of ha ALAAP respectively ha of ALAPP. This resulted in total figures on kg N leaching from ALAAP and ALAPP from each country.
4. In the last step, total kg N leaching from ALAAP/ALAPP was divided by total kg of Animal-Protein produced per country respectively total kg of Plant-Protein produced per country, resulting in figures on kg N/kg Animal-Protein and kg N/kg Plant-Protein. The same methodology was used for calculating kg N/ 10 000 Animal-kcal and kg N/10 000 Plant-kcal, only using the kcal conversion factors instead of protein conversion factors.

Table 14. Compilation of the agri-environmental indicators encountered during the work of this thesis.

Category	Indicator	Supporting-indicators	Unit	Source of data	Reference
Land use	Agricultural land	Arable land	ha	OECD + Eurostat	OECD, EEA
		Permanent crop	ha	OECD + Eurostat	OECD, EEA
		Permanent pasture	ha	OECD + Eurostat	OECD, EEA
		Yield of cereals	ton/ha		ELISA
		Share of farms with more than 50% cereals	%	FADN	ELISA
		Share of grassland in total UAA	%	FADN	ELISA
		Conservation agriculture	ha	Eurostat	OECD
		Agricultural commitments	% of ha	DG-AGRI	Eurostat
Livestock	Livestock patterns	Total livestock density	LSU/ha of UAA	Eurostat	EEA
		Grazing livestock density	LSU/ha of UAA	Eurostat	EEA
	Manure storage	Holdings with livestock and manure storage facilities	%	Eurostat	EEA
Nutrients	Nitrogen	Nutrient inputs	ton	OECD + Eurostat	OECD
		Total inorganic fertilizer	ton	OECD + Eurostat	OECD
		Total organic fertilizer (excluding manure)	ton	OECD + Eurostat	OECD
		Net input manure (divided by species)	ton	OECD + Eurostat	OECD
		Other nutrient inputs (deposition, biological fixation)	ton	OECD + Eurostat	OECD
		Nutrient outputs (total harvested, crop by crop, pasture)	ton	OECD + Eurostat	OECD
		Balance (inputs - outputs)	ton	OECD + Eurostat	OECD
		Gross balance per hectare	kg	OECD + Eurostat	OECD
	Phosphorus	Nutrient inputs	ton	OECD + Eurostat	OECD
		Total inorganic fertilizer	ton	OECD + Eurostat	OECD
		Total organic fertilizer (excluding manure)	ton	OECD + Eurostat	OECD
		Net input manure (divided by species)	ton	OECD + Eurostat	OECD
		Other nutrient inputs (deposition, biological fixation)	ton	OECD + Eurostat	OECD
		Nutrient outputs (total harvested, crop by crop, pasture)	ton	OECD + Eurostat	OECD
		Balance (inputs - outputs)	ton	OECD + Eurostat	OECD
		Gross balance per hectare	kg	OECD + Eurostat	OECD
	Fertilizer use	N/P	ton	Eurostat	EEA
		Average use per area of cropland (N, P, K)	kg/ha	FAOSTAT	FAO
		Critical load exceedance for nitrogen	eq/ha	CCE IMPACT	EEA
Emissions	NH3 emissions		ton	OECD + Eurostat	OECD
		% of total NH3 emissions from agriculture	%	EMEP	EEA
		NH3 emissions from agriculture	kton/yr	Eurostat	EEA
	GHG emissions	Total CH4 emissions from agriculture	kton CO2 eq	OECD + Eurostat	OECD
		Total emissions agriculture (CO2 eq)	Gg of CO2 eq	FAOSTAT	FAO
		Total N2O emissions from agriculture	kton of CO2 eq,	OECD + Eurostat	OECD
		CO2 emissions from cultivation of organic soils	Gg of CO2 eq	FAOSTAT	FAO
		GHG emissions per commodity	kg CO2 eq/kg prod	FAOSTAT	FAO
		Divided by categories (manure, animal species etc)	Gton CO2 eq/yr	FAOSTAT	FAO
Soil quality	Soil	Organic content	% carbon	HWSD	FAO
		Soil erosion	GLASOD	GLASOD	FAO
		Soil cover arable land	%	Various	EEA
		Heavy metals	n.d.	n.d.	ELISA
		Soil salinisation	n.d.	n.d.	ELISA
		Land degradation	GLASOD	GLASOD	FAO
		Soil moisture	litres/m3/10 yrs	EEA	EEA
	Tillage practices	Arable areas under conventional, conservation and zero tillage	%	Eurostat	EEA
	Soil erosion	Total agricultural land affected by wind erosion	%	OECD + Eurostat	OECD
		Total agricultural land affected by water erosion	%	OECD + Eurostat	OECD
		Soil water erosion rate by country	ton/ha/yr	JRC	EEA
Water Use	Resources	Agriculture freshwater abstraction	m3, millions	OECD + Eurostat	OECD
		Ground water abstraction	m3, millions	OECD + Eurostat	OECD
		Surface water abstraction	m3, millions	OECD + Eurostat	OECD
		Water withdrawl agriculture (km3)	km3	AQUASTAT	FAO
		Water Exploitation Index (WEI)	%	EEA	EEA
		Agricultural water withdrawl as % of TRWR	%	AQUASTAT	FAO
	Irrigation	Irrigation area	ha	OECD + Eurostat	OECD, EEA
		Irrigable area	ha	OECD + Eurostat	OECD, EEA
		Irrigated area of total UAA (Utilized Agricultural Area)	%	Eurostat	EEA
		Irrigation water withdrawl	km3/yr	AQUASTAT	FAO
		Pressure on freshwater resources due to irrigation	%	AQUASTAT	FAO
		Water requirement ratio; irrigation efficency	%	AQUASTAT	FAO
Quality	Water quality				
		% of agriculture in total emissions of N/P in ground water	%	OECD + Eurostat	OECD
		% of agriculture in total emissions of N/P in surface water	%	OECD + Eurostat	OECD
		% of agriculture in total emissions of N/P in coastal water	%	OECD + Eurostat	OECD
		Pesticides in monitoring sites in agricultural areas (surfacewater)	%	OECD + Eurostat	OECD
		Pesticides in monitoring sites in agricultural areas (groundwater)	%	OECD + Eurostat	OECD
		Monitoring sites > drinking water limits for N/P/pesticides (groundwater)	%	OECD + Eurostat	OECD
		Monitoring sites > drinking water limits for N/P/pesticides (surfacewater)	%	OECD + Eurostat	OECD
	NO3 pollution	Rivers and groundwaters with NO3 concentrations above 50mg NO3/l	%	EEA	EEA
	Water use	Water intensity of crop production	%	Eurostat	EEA
Pesticides	Pesticides	Average use per area of cropland	kg a.i./ha	FAOSTAT	FAO
	Pesticide sales	Insecticides	ton	OECD + Eurostat	OECD
		Herbicides	ton	OECD + Eurostat	OECD
		Fungicides	ton	OECD + Eurostat	OECD

Popular scientific summary

When the environmental performance of Swedish food production was compared in a study with other European countries, Sweden showed to perform well, when looking at environmental factors such as greenhouse gas emissions, ammonia emissions, pesticide use and leaching of the nutrients nitrogen and phosphorus. The positive results for Sweden could be explained by low number of animals per agricultural surface area, as well as low nutrient inputs and to some extent national environmental policy. Countries with high nutrient inputs and high livestock densities were generally coupled to lower environmental performance.

In this report, the environmental performance of Swedish food production was evaluated through an international comparison with other European countries, with similar agricultural and climatic conditions as Sweden, by analysis of agri-environmental indicators. The study shows that there might be difficulties when using some of the indicators, as countries might have different methods for calculating or collecting data, reducing the comparability of the indicators. Further, national figures might be misleading, as the regional variance within a country can be high. Thus, the environmental impact might be unevenly distributed, and some areas might have high environmental pressure due to agricultural activities, while others have low environmental pressure.

Agri-environmental indicators are used as important tools for developing and evaluating the progress of agri-environmental policy. Indicators are used to be able to predict variables which can be difficult to measure, such as eutrophication. An indicator such as *Mineral fertilizer use* can be used to predict the environmental performance instead of doing actual measurements. It is therefore important to explore if this type of indicators manage to predict the environmental performance. In this study, it was concluded that a high mineral fertilizer use does not necessarily correlate to a high nutrient leaching, especially not for phosphorus, as phosphorus loss is highly dependent of natural conditions and can be lost very occasionally.

This study also demonstrated the importance of functional unit, when the functional unit was mass based (kg) instead of land based (ha), for the environmental impact *Nutrient leaching*, the Swedish performance was reduced. A mass unit favours countries with high production, generally coupled to an intensive production with high livestock densities and nutrient inputs. In this context, Sweden has an extensive agriculture with low nutrient inputs and low animal densities. It is however important to regard also production figures as we do need to produce more food for a growing population in the coming decades.