Food and Agriculture
Organization of the
United Nations

VERSION 1

Nutrient flows and associated environmental impacts in livestock supply chains

Guidelines for assessment

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FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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Contents

Foreword vii
Acknowledgements ix
Multi-step review process xiii
Sponsors, advisors and networking xv
Abbreviations and Acronyms xvii
Glossary xix
Summary of the LEAP Guidelines xxxiii
LEAP and the preparation process xxxvii
Nutrient TAG and the preparation process xxxix

PART 1
OVERVIEW AND GENERAL PRINCIPLES 1

1 OBJECTIVES AND INTENDED USERS 3

2 SCOPE AND IMPACT CATEGORIES 5
2.1 Application 6
2.2 Livestock species and production systems 6
2.3 Normative references 6

PART 2
METHODOLOGY FOR QUANTIFICATION OF NUTRIENT FLOWS AND ENVIRONMENTAL IMPACTS FOR EUTROPHICATION AND ACIDIFICATION IN LIVESTOCK SUPPLY CHAINS 9

3 GOAL AND SCOPE DEFINITION 11
3.1 Goal and scope of the study 11
3.2 Functional unit and system boundary 11
3.3 Nutrient flows to consider 11
3.4 Scale consistent assessment 13

4 LIFE CYCLE INVENTORY 15
4.1 Overview 15
4.2 Feed production 16
4.2.1 Introduction 16
4.2.2 Input flows to feed production systems 17
4.2.3 Output flows 20
4.2.4 Internal flows 23
4.2.5 Attributing emissions and resource use to single production units 26
4.2.6 Field-to-Gate assessment 27
4.3 Animal Husbandry: Confined or Housed, Grazing and Mixed Animal Systems 28
4.3.1 Introduction 28
4.3.2 Quantification of input flows 29
4.3.3 Quantification of output flows 31

4.3.4 Allocation of emissions to manure 36
4.4 Animal processing 37
4.4.1 Nitrogen and phosphorus output in products 38
4.4.2 Residues, waste and wastewater treatment 38
4.5 Upstream processes and transportation 40
4.5.1 Fertilizer production 40
4.5.2 Production and use of cleaning chemicals, refrigerants and other consumables 40
4.5.3 Generation and use of energy 41
5 LIFE CYCLE IMPACT ASSESSMENT (LCIA) 43
5.1 Impact categories 43
5.1.1 Eutrophication: environnemental cause-effet chain 45
5.1.2 From the inventory of nutrient emissions to impact assessment for eutrophication 45
5.1.3 Acidification 47
5.2 Generic versus site-specific assessment 47
5.3 Recommendations for Impact Assessment in LCA 48
5.3.1 Eutrophication 48
5.3.2 Freshwater Eutrophication 48
5.3.3 Marine Eutrophication 49
5.3.4 Acidification 49
5.3.5 Sensitivity analysis and current developments 49
6 RESOURCE USE ASSESSMENT 51
6.1 Nutrient use efficiency at each production stage 52
6.2 Life cycle nutrient use efficiency 53
7 INTERPRETATION OF RESULTS 55
7.1 Data quality 55
7.2 Significant issues 55
7.3 Evaluation 55
7.3.1 Completeness check 55
7.3.2 Sensitivity check 56
7.3.3 Consistency check 59
7.4 Additional indicators to support the interpretation of nutrient budget analysis 60
7.4.1 Nitrogen and phosphorus footprints 60
7.4.2 Gross nutrient surplus 60
7.4.3 Circularity indicator 60
7.5 Conclusions, recommendations and limitations 61
7.5.1 Good practice in reporting LCA results 62
7.5.2 Report elements and structure 62
7.5.3 Critical review 63
8 REFERENCES 65
APPENDICES 77
APPENDIX 1
Tiered approaches 79

APPENDIX 2
Nutrient assessment – relevant guidelines 81

APPENDIX 3
Biological nitrogen fixation 83

APPENDIX 4
Estimating the soil non-labile phosphorus pool 87

APPENDIX 5
Nitrogen and phosphorus losses from feed storage 91

APPENDIX 6
Estimation of the N and P content of animals and animal products 97

APPENDIX 7
Approach to allocate upstream livestock emissions to manure and livestock products 103

APPENDIX 8
Excreta deposition and spatial variability, source/site factors affecting N and P loss, and index methods for estimating nutrient losses 113

APPENDIX 9
Fertilizer production 119

APPENDIX 10
Example data for upstream processes for fertilizer manufacturing emissions and for energy use and emissions for animal product processing and electricity 123

APPENDIX 11
Background principles for eutrophication and acidification 125

APPENDIX 12
Use of Biosolids as fertilizer in agriculture 133

APPENDIX 13
Construction of the matrices for the calculation of the life cycle nutrient use efficiency 137

CASE STUDIES TO ILLUSTRATE INVENTORY DATA AND RESULTS FROM A RANGE OF LIVESTOCK SYSTEMS 139

CASE STUDY 1
Lamb production in New Zealand to consumption in the United Kingdom 141

CASE STUDY 2
Beef and sheep extensive grazing system in Uruguay 147

CASE STUDY 3
Egg (medium size) production, in combination with pigs and cereal production in Sweden 151

CASE STUDY 4
Fully grazing dairy cattle supply chain in Rwanda 153

Foreword

The aim of the methodology developed in these guidelines is to introduce a harmonized international approach assessing nutrient flows and impact assessment for eutrophication and acidification for livestock supply chains taking the specificity of the various production systems involved into consideration. The methodology strives to increase understanding of nutrient use efficiency and associated environmental impacts and to facilitate improvement of livestock systems' environmental performance. The guidelines are a product of the Livestock Environmental Assessment and Performance (LEAP) Partnership, a multi-stakeholder initiative whose goal is to improve the environmental sustainability of livestock sector through better metrics and data.

Nutrient use in livestock production systems increased over the last decades due to the increased demand for livestock production. This demand is mainly driven by the increase in the population growth, population income, and urbanization. Consequently, in livestock supply chains, nutrient losses into the environment have contributed to environmental burdens such as climate change, air and water pollution, degradation of soil quality, loss of biodiversity and human health issues. Therefore, there is strong interest in measuring nutrient flows to improve the environmental performance of the livestock sector.

The objectives of these guidelines are:

- To develop a harmonized, science-based approach resting on a consensus among the sector's stakeholders;
- To recommend a scientific, but at the same time practical, approach that builds on existing or developing methodologies;
- To promote an harmonised approach to assess nutrient flows and impact assessment, relevant for global livestock supply chains;
- To identify the principal areas where ambiguity or differing views exist concerning the methodological framework.

During the development process, these guidelines were submitted for technical review and public review. The purpose is to strengthen the advice provided and ensure it meets the needs of those seeking to improve nutrient use efficiency and environmental performance through sound assessment practice. This document is not intended to remain static. It will be updated and improved as the sector evolves and more stakeholders become involved in LEAP, and as new methodological frameworks and data become available.

The guidelines developed by the LEAP Partnership gain strength because they represent a multi-actor coordinated cross-sectoral and international effort to harmonize assessment approaches. Ideally, the harmonization leads to greater understanding, transparent application and communication of metrics, and, not least, real and measurable improvement in environmental performance.

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Abbreviations and acronyms

BD	Bulk density
DM	Dry Matter
EEA	European Environment Agency
FAO	Food and Agriculture Organization of the United Nations
GDP	Gross domestic product
GHG	Greenhouse gas
HAB	Harmful algal blooms
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LEAP	Livestock Environmental Assessment and Performance (LEAP) Partnership
N	Nitrogen
NUE	Nutrient use efficiency
OECD	Organisation for Economic Co-operation and Development
P	Phosphorus
TAG	Technical Advisory Group
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change

Glossary

Acidification	Impact category that addresses impacts due to acidifying substances in the environment. Emissions of NO_x, NH_3 and SO_x lead to release of hydrogen ions (H^+) when the gases are mineralised. The protons contribute to the acidification of soils and water when they are released in areas where the buffering capacity is low, resulting in forest decline and lake acidification. [Product Environmental Footprint Guide, European Commission, 2013].
Activity data	Data on the magnitude of human activity resulting in emissions or removals taking place during a given period of time [UNFCCC, 2014].
Agricultural land	Arable crops (e.g. cereals), permanent crops (e.g. orchards) and permanent pasture (i.e. land devoted to livestock grazing for periods longer than 5 years) [OECD, 2001].
Agro-Ecological Zones (AEZ)	A framework for the characterization of climate, soil and terrain conditions relevant to agricultural production.
Allocation	Partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems [ISO 14044:2006, 3.17].
Ammonification	Amino acids released during proteolysis undergo deamination in which nitrogen containing amino (-NH_2) group is removed. Thus, process of deamination which leads to the production of ammonia is termed as “ammonification”. The process of ammonification is mediated by several soil microorganisms. Ammonification usually occurs under aerobic conditions (known as oxidative deamination) with the liberation of ammonia (NH_3) or ammonium ions (NH_4) which are either released to the atmosphere or utilized by plants or microorganisms and under favourable soil conditions oxidized to form nitrites and then to nitrates.
Annual plants	Crops established annually, usually with annual plants, and generally involves soil disturbance, removal of existing vegetation, and other cultivation practices e.g. feed grain, fodder root crops.
Attributional modelling approach	System modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule [UNEP/SETAC Life Cycle Initiative, 2011].

Background system	The background system consists of processes on which no or, at best, indirect influence may be exercised by the decision-maker for which an LCA is carried out. Such processes are called "background processes." [UNEP/SETAC Life Cycle Initiative, 2011].
Biomass	Material of biological origin excluding material embedded in geological formations and material transformed to fossilized material, and excluding peat [ISO/TS 14067:2013, 3.1.8.1].
By-product	Material produced during the processing (including slaughtering) of a livestock or crop product that is not the primary product of the activity (e.g. oil cakes, meals, offal or skins). Most of the by-products have economic value.
Capital goods	Capital goods are final products that have an extended life and are used by the company to manufacture a product; provide a service; or sell, store, and deliver merchandise. In financial accounting, capital goods are treated as fixed assets or as plant, property, and equipment (PP&E). Examples of capital goods include equipment, machinery, buildings, facilities, and vehicles (GHG Protocol, Technical Guidance for Calculating Scope 3 Emissions, 2013, Chapter 2).
Catch crops	Catch crops are grown in the period between two main crops to retain nutrients in the root zone.
Cover crops	Cover crops are grown in the period between two main crops to protect the soil against erosion and minimise the risk of surface runoff by improving the infiltration.
Characterization	Calculation of the magnitude of the contribution of each classified input/output to their respective impact categories, and aggregation of contributions within each category. This requires a linear multiplication of the inventory data with characterisation factors for each substance and impact category of concern. For example, with respect to the impact category "climate change", CO_2 is chosen as the reference substance and kg CO_2-equivalents as the reference unit (Adapted from: Product Environmental Footprint Guide, European Commission, 2013).
Characterization factor	Factor derived from a characterization model which is applied to convert an assigned life cycle inventory analysis result to the common unit of the category indicator (ISO 14044:2006, 3.37).

Circularity	Circularity is a measure of the degree that nutrients are not used in the final product(s) but are re-used in the processes substituting for input of new/external nutrient inputs.
Classification	Assigning the material/energy inputs and outputs tabulated in the Life Cycle Inventory to impact categories according to each substance's potential to contribute to each of the impact categories considered (Adapted from: Product Environmental Footprint Guide, European Commission, 2013).
Combined production	A multifunctional process in which production of the various outputs can be independently varied. For example in a backyard system the number of poultry and swine can be set independently.
Compound feed/concentrate	Mixtures of feed materials which may contain additives for use as animal feed in the form of complete or complementary feedstuffs.
Conservation tillage	A tillage system that creates a suitable soil environment for growing a crop and that conserves soil, water and energy resources mainly through the reduction in the intensity of tillage, and retention of plant residues (OECD, 2001).
Content	Content is a fraction, here usually mass per mass (for example kg N kg $soil^{-1}$) (Campbell & Schilfgaarde, 1981).
Conventional tillage	A tillage system using cultivation as the major means of seedbed preparation and weed control. Typically includes a sequence of soil tillage, such as ploughing and harrowing, to produce a fine seedbed, and also the removal of most of the plant residue from the previous crop. In this context the terms cultivation and tillage are synonymous, with emphasis on soil preparation (OECD, 2001).
Co-production	A generic term for multifunctional processes; either combined- or joint-production.
Co-products	Any of two or more products coming from the same unit process or product system (ISO 14044:2006, 3.10).
Cover crop	A temporary vegetative cover that is grown to provide protection for the soil and the establishment of plants, particularly those which are slow growing. Some cover crops are introduced by under-sowing and in due course provide permanent vegetative cover to stabilise the area concerned. The term can include an intermediate crop that can be removed using selective herbicides (OECD, 2001).

Cradle-to-gate	Life-cycle stages from the extraction or acquisition of raw materials to the point at which the product leaves the organization undertaking the assessment (PAS 2050:2011, 3.13).
Crop residues	Materials left in an agricultural field after the crop has been harvested.
Data quality	Characteristics of data that relate to their ability to satisfy stated requirements (ISO 14044:2006, 3.19).
Denitrification	It is the reverse process of nitrification. During denitrification, nitrates are reduced to nitrites and then to nitrous oxide and dinitrogen gas. Thus, reduction of nitrates to gaseous nitrogen by microorganisms in a series of biochemical reactions is called "denitrification". The process is wasteful as available nitrogen in soil is lost to atmosphere.
Dung	Faeces from mammalian livestock (Pain and Menzi, 2011).
Economic value	Average market value of a product at the point of production possibly over a 5-year time frame (Adapted from PAS 2050:2011, 3.17). Note 1: whereas barter is in place, the economic value of the commodity traded can be calculated based on the market value and amount of the commodity exchanged.
Ecosystem	An ecosystem is a system in which the interaction between different organisms and their environment generates a cyclic interchange of materials and energy (OECD, 2001).
Edible offal	In relation to slaughtered food animals, offal that has been passed as fit for human consumption.
Elementary flow	Material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation (ISO 14044:2006, 3.12).
Emission factor	Amount of substance (e.g. nitrogen, nitrous oxide, phosphorus) emitted, expressed a unit equivalent and relative to a unit of base input (e.g. kg N_2O per kg N input) (Adapted from UNFCCC, 2014).
Emission intensity	Emission intensity is the level of emissions per unit of economic activity or product. Usually the term 'emission intensity' is used in relation to CO_2 emissions of a country, measured at the national level as GDP (Baumert *et al.,* 2005) or for specific economic outputs (kilowatt-hours, or tonnes of steel produced).

Emissions	Release of substance to air and discharges to water and land.
Enrichment	Enrichment is the addition of nitrogen, phosphorus and carbon compounds or other nutrients into a different ecosystem (water, air, soil), thereby increasing the potential for growth of algae and other aquatic plants. Most frequently, enrichment results from the inflow of sewage effluents or from agricultural run-off (OECD, 2001).
Environmental impact	Any change to the environment, whether adverse or beneficial, wholly or partially resulting from an organization's activities, products or services (ISO/TR 14062:2002, 3.6).
Erosion	Loss of surface soil due to the action of wind or water (including from rainfall and glaciers).
Eutrophication	Nutrient output (mainly nitrogen and phosphorus), such as from sewage outfalls and fertilized farmland, that accelerates the growth of algae and other vegetation in water. The degradation of organic material consumes oxygen resulting in oxygen deficiency and, in some cases, fish death. Eutrophication translates the quantity of substances emitted into a common measure expressed as the oxygen required for the degradation of dead biomass (Product Environmental Footprint Guide, European Commission, 2013).
Excreta	Waste expelled from the body: faeces plus urine (Pain and Menzi, 2011).
Extrapolated data	Refers to data from a given process that is used to represent a similar process for which data is not available, on the assumption that it is reasonably representative for all aspects (Product Environmental Footprint Guide, European Commission, 2013).
Faeces	Solid waste or undigested material voided by animals (Pain and Menzi, 2011).
Flow	Nutrient flows describe the transport of nutrient over time between the various pools of a nutrient, or between the sub-pools within a pool. Flows of nutrient can occur as reactive nitrogen (Nr) or phosphorus. Flows must be represented in the same unit, e.g. in kg of N per year (Adapted from UNECE, 2013).
Flow diagram	Schematic representation of the flows occurring during one or more process stages within the life cycle of the product being assessed (Product Environmental Footprint Guide, European Commission, 2013).

Flux Flow density or flow of N or P over a unit area. Often the term of "fluxes" is used as a synonym of "flux rates" thus the time dependency is implicitly included. If the flux transports nitrogen to an environmental pool, the term emission flux can be used. Depending on the scale of the assessment, a flux is measured on a hectare-basis (e.g. if referring to agricultural area) or on a basis of a square metre (measurements or plot/field-scale averages) or square kilometre (for large-scale regional averages).

Footprint Footprints are metrics used to report life cycle assessment results addressing an area of concern (Ridoutt *et al.*, 2016). They represent the sum of emissions that are caused by the production of one unit of final product, scaling processes such that the quantity of intermediate products produced equals the quantity required in the subsequent supply chain stages (Heijungs and Suh, 2002).

Foreground system The foreground system consists of processes directly influenced by the decision-maker for which an LCA is carried out. Such processes are called "foreground processes" (UNEP/SETAC Life Cycle Initiative, 2011).

Freshwater Naturally occurring water on the earth's surface (e.g. in rivers, lakes, glaciers) and underground as groundwater, with low concentrations of dissolved solids and salts (e.g. < 1000 ppm) (American Meteorological Society 2011).

Full grazing Production system for livestock in which the animals receive no additional roughage and consume grassland plant material directly by grazing to reduce production costs. The system is usually combined with calving/lambing/kidding in spring to synchronise feed requirements with plant growth (Pain and Menzi, 2011).

Functional unit Quantified performance of a product system for use as a reference unit (ISO 14044:2006, 3.20). It is essential that the functional unit allows comparisons that are valid where the compared objects (or time series data on the same object, for benchmarking) are comparable.

Grasslands Forage that is established (imposed grazing-land ecosystem) with domesticated introduced or indigenous species that may or may not receive periodic cultural treatment such as renovation, fertilization or weed control. The vegetation of grassland in this context is broadly interpreted to include grasses, legumes and other forbs, and at times woody species may be present.

Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned (ISO 14044:2006, 3.39).
Impact category indicator	Quantifiable representation of an impact category (ISO 14044:2006, 3.40).
Inactive nitrogen	Some forms of nitrogen may be considered inactive or inert as they are inaccessible to bio-substrates. This regards primarily molecular nitrogen (N_2), which is the dominant N species. Flows of N_2 between different pools do not need to be quantified in a nitrogen budget. N_2 requires considerable amount of energy to become bio-available. This activation process then constitutes a flow bringing Nr from this origin into a nitrogen budget. Other inactive natural forms of N are excluded from the nitrogen budget until being activated (e.g. N contained in mineral oil and its products) (UNECE, 2012).
Input	Product, material or energy flow that enters a unit process (ISO 14044:2006, 3.21).
Land use change	Change in the purpose for which land is used by humans (e.g. between crop land, grass land, forestland, wetland, industrial land) (PAS 2050:2011, 3.27).
LCA	See Life Cycle Assessment.
LCI	See Life Cycle Inventory.
LCIA	See Life Cycle Impact Assessment.
Leaching	The downward transport of nutrient (e.g. nitrate-nitrogen) in soil solution with drainage water.
Life cycle	Consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal (ISO 14044:2006, 3.1).
Life Cycle Assessment	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 14044:2006, 3.2).
Life Cycle Impact Assessment (LCIA)	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential impacts for a product system throughout the life cycle of the product (Adapted from: ISO 14044:2006, 3.4).

Life Cycle Interpretation	Phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are evaluated in relation to the defined goal and scope to reach conclusions and recommendations (ISO 14044:2006, 3.5).
Life Cycle Inventory (LCI)	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle (ISO 14046:2014, 3.3.6).
Liquid manure	A general term that denotes any manure from housed livestock that flows under gravity and can be pumped. There are several different types of liquid manure arising from different types of livestock housing, manure storage and treatment (Pain and Menzi, 2011).
Manure	A general term to denote any organic material that supplies organic matter to soils together with plant nutrients, usually in lower concentrations compared to synthetic fertilizer (Pain and Menzi, 2011).
Manure management	The collection, storage, transport and application of manures to land. May also include treatment (Pain and Menzi, 2011).
Manure surplus	An amount of manure containing plant nutrients in excess of those required by crops (Pain and Menzi, 2011).
N-forms	Nitrogen can occur in various forms, some of which are reactive, and some of which are inactive (UNECE, 2012).
Nitrification	Nitrification is a biological process involving the conversion of nitrogen-containing organic compounds into nitrates and nitrites. It is part of the nitrogen cycle and considered to be beneficial because it converts organic nitrogen compounds into nitrates that can be absorbed by green plants (OECD, 2001).
Nitrogen fixation	The conversion of dinitrogen (N_2) to nitrogen combined with other elements; specifically, regarding soils, the assimilation of atmospheric nitrogen from the soil air by soil organisms to produce nitrogen compounds that eventually become available to plants (OECD, 2001).
Nutrient	Substance required by an organism for growth and development. Key crop nutrients are nitrogen, phosphorus and potassium (OECD, 2001).

Nutrient Balance	A Nutrient balance consists of the quantification of all major nutrient flows across the boundaries of a given system, as well as the changes of nutrient stocks within the system. Flows that are internal of the system are not considered. The balance equation is 'Output + Stock change – Input = 0' (adapted from UNECE, 2012).
Nutrient Budget	A Nutrient budget consists of the quantification of all major nutrient flows across all sectors and media within given boundaries, and flows across these boundaries, in a given time frame (typically one year), as well as the changes of nutrient stocks within the respective sectors and media. Nutrient Budgets can be constructed for any geographic entity, for example at regional level (e.g. Europe), for country, for watersheds or even individual farm (adapted from UNECE, 2012).
Organic wastes	A general term for any carbon-containing wastes from organic rather than inorganic origin (e.g. Livestock manure, sewage sludge, abattoir wastes) (Pain and Menzi, 2011).
Output	Product, material or energy flow that leaves a unit process (ISO 14044:2006, 3.25).
Particulate matter	Impact category that accounts for the adverse health effects on human health caused by emissions of Particulate Matter (PM) and its precursors (NO_x, SO_x, NH_3) (Product Environmental Footprint Guide, European Commission, 2013).
Pools	Nutrient pools are elements in a nutrient budget. They represent "containers" which serve to store quantities of nutrient (these quantities may be referred to as nutrient stocks). Exchange of nutrient occurs between different pools via nutrient flows. Nutrient pools can be environmental media (e.g. atmosphere, water), economic sectors (e.g. industry, agriculture) or other societal elements (e.g. humans and settlements). Selection of pools may differ between budgets (Adapted from UNECE, 2012).
Primary activity data	Quantitative measurement of activity from a product's life cycle that, when multiplied by the appropriate emission factor, determines the emissions arising from a process. Examples of primary activity data include the amount of energy used, material produced, service provided, or area of land affected (PAS 2050:2011, 3.34).
Primary data	Quantified value of a unit process or an activity obtained from a direct measurement or a calculation based on direct measurements at its original source (ISO 14046:2014, 3.6.1).

Product system	Collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product (ISO 14044:2006, 3.28).
Reactive nitrogen	Reactive nitrogen (Nr) is any form of nitrogen that is available relatively easily to living organisms via biochemical processes. These compounds include ammonia (NH_3), nitrogen oxide (NO_x), nitrous oxide (N_2O), nitrate (NO_3), organically-bound N in plants, animals, humans and soil – and many other chemical forms (UNECE, 2012).
Releases	Emissions to air and discharges to water and soil (ISO 14044:2006, 3.30).
Residue or Residual	Substance that is not the end product(s) that a production process directly seeks to produce (Communication from the European Commission 2010/C 160/02). More specifically, a residue is any material without economic value leaving the product system in the condition as it created in the process, but which has a subsequent use. There may be value-added steps beyond the system boundary, but these activities do not impact the product system calculations. Note 1: Materials with economic value are considered co-products (see by-product). Note 2: Materials whose economic value is both negligible relative to the annual turnover of the organization and is also entirely determined by the production costs necessary divert materials from waste streams are to be considered as residues from an environmental accounting perspective. Note 3: Those materials whose relative economic value volatility is high in the range of positive and negative value, and whose average value is negative are residues from an environmental accounting perspective. Materials economic value volatility is possibly calculated over a 5-year time-frame at the regional level.
Resource depletion	Impact category that addresses use of natural resources either renewable or non-renewable, biotic or abiotic (Product Environmental Footprint Guide, European Commission, 2013).
Runoff	The portion of precipitation not immediately absorbed into or detained on soil and which thus becomes surface water flow (OECD, 2001).

Secondary data	Data obtained from sources other than a direct measurement or a calculation based on direct measurements at the original source (ISO 14046:2014, 3.6.2). Secondary data is used when primary data is not available, or it is impractical to obtain primary data. Some emissions, such as methane from litter management, are calculated from a model, and are therefore considered secondary data.
Secondary packaging materials	Containers/packaging and materials, which are used in raw materials acquisition, production and distribution but which do not reach consumers.
Sediment	Material of varying size, both mineral and organic that is being, or has been, moved from its site of origin by the action of wind, water, gravity, or ice, and comes to rest elsewhere on the earth's surface (OECD, 2001).
Sensitivity analysis	Systematic procedures for estimating the effects of the choices made regarding methods and data on the outcome of a study (ISO 14044:2006, 3.31).
Sewage	Liquid domestic and municipal waste (Pain and Menzi, 2011).
Sewage sludge	By-product of sewage treatment that concentrates solids. It contains significant quantities of various essential nutrients for plants and animals (Pain and Menzi, 2011).
Silage	Forage harvested and preserved (at high moisture contents, generally >500 g kg^{-1}) by organic acids produced during partial anaerobic fermentation.
Sludge	The liquid or semi-solid fraction arising from the sedimentation or flocculation of liquid waste or liquid manure (Pain and Menzi, 2011).
Slurry	Faeces and urine produced by housed livestock, usually mixed with some bedding material and some water during management to give a liquid manure with a dry matter content in the range from about 1 – 10 percent (Pain and Menzi, 2011).
Soil Organic Matter (SOM)	The measure of the content of organic material in soil. This derives from plants and animals (adapted, Product Environmental Footprint Guide, European Commission, 2013).

Soil quality	Encompasses two distinct, but related parts: inherent quality, the innate properties of soils such as those that lead to soil formation; and dynamic quality, covering the main degradation processes (physical, chemical and biological) and farm management practices (OECD, 2001).
Stock	Stocks represent real-world accumulations. Each pool can store a quantity of nutrient, for example, as mineral or organic nitrogen in soils (for instance as in agriculture or semi-natural lands/pools). This quantity is the nutrient stock. Nutrient stocks may be very large with respect to nutrient flows (e.g. for soil pools), and often nutrient stocks are difficult to quantify. However, the most relevant parameter for the nutrient budget is a potential stock change, i.e. a variation over time of the respective accumulation, rather than the nitrogen stock itself. Nutrient stocks can be composed of nutrient in any form (Adapted from UNECE, 2012).
Sub-pools	Pools can be further divided into sub-pools if sufficient data is available. For example, the pool "inland water" can be divided into groundwater, lakes, rivers, etc., with additional nutrient flows across these sub-pools to be quantified (adapted from UNECE, 2012).
System boundary	Set of criteria specifying which unit processes are part of a product system (ISO 14044:2006, 3.32).
System expansion	Expanding the product system to include additional functions related to co-products.
Techno-sphere	The part of the physical environment affected through building or modification by humans.
Tier-1 method	Simplest method that relies on single default emission factors (e.g. kg excreta-nitrogen per animal).
Tier-2 method	A more complex approach that uses detailed country-specific data (e.g. gross nitrogen intake less nitrogen in products for specific livestock categories).
Tier-3 method	Method based on sophisticated mechanistic models that account for multiple factors such as diet composition, product concentration, and seasonal variation in animal and feed parameters.
Uncertainty analysis	Systematic procedure to quantify the uncertainty introduced in the results of a life cycle inventory analysis due to the cumulative effects of model imprecision, input uncertainty and data variability (ISO 14044:2006, 3.33).

Unit process	Smallest element considered in the life cycle inventory analysis for which input and output data is quantified (ISO 14044:2006, 3.34).
Upstream	Occurring along the supply chain of purchased goods/ services prior to entering the system boundary (Product Environmental Footprint Guide, European Commission, 2013).
Volatile Solids (VS)	Volatile solids (VS) are the organic material in livestock manure and consist of both biodegradable and non-biodegradable fractions. The VS content of manure equals the fraction of the diet consumed that is not digested and thus excreted as faecal material which, when combined with urinary excretions, constitutes manure.
Volatilization	Gaseous loss of the volatile form of a nutrient (e.g. ammonia).
Waste	Substances or objects which the holder intends or is required to dispose of (ISO 14044:2006, 3.35). Note 1: Deposition of manure on a land where quantity and availability of soil nutrients such as nitrogen and phosphorus exceed plant nutrient requirement is considered as a waste management activity from an environmental accounting perspective. See also: **Residual and Economic value**.
Wastewater	A general term for contaminated water e.g. with faeces, urine, milk, chemicals etc., so posing a risk of pollution (Pain and Menzi, 2011).
Water body	Entity of water with definite hydrological, hydrogeomorphological, physical, chemical and biological characteristics in a given geographical area. Examples: lakes, rivers, groundwaters, seas, icebergs, glaciers and reservoirs. Note 1 to entry: In case of availability, the geographical resolution of a water body should be determined at the goal and scope stage: it may regroup different small water bodies (ISO 14046:2014, 3.1.7).

Summary of the LEAP Guidelines

NUTRIENT FLOWS AND ASSOCIATED ENVIRONMENTAL IMPACTS IN LIVESTOCK SUPPLY CHAINS: GUIDELINES FOR ASSESSMENT

The methodology in these guidelines aims to introduce an internationally-harmonised approach to assess the potential environmental impacts associated with nutrient use in livestock supply chains, while considering the different nutrient flows in the various production systems involved. *These guidelines aim to increase understanding of nutrient flows in livestock supply chains and their impact assessment in relation to eutrophication and acidification*. These guidelines are a product of the Livestock Environmental Assessment and Performance (LEAP) Partnership, a multi-stakeholder initiative committed to improving the environmental performance of livestock supply chains, whilst ensuring its economic and social viability. LEAP builds up consensus on comprehensive guidance and methodology for understanding the environmental performance of livestock supply chains, in order to shape evidence-based policy measures and business strategies. The table below summarises the major recommendations for the *environmental quantification of nutrient flows and impact assessment in livestock supply chains*. The table is intended to provide a condensed overview and information on the location of specific guidance within the document.

All LEAP guidance documents use a normative language to indicate which provisions of the guidelines are requirements, which are recommendations, and which are permissible or allowable options that the intended user may choose to follow. The term “shall” is used in this guidance to indicate what is required. The term “should” is used to indicate a recommendation, but not a requirement. The term “may” is used to indicate an option that is permissible or allowable. In addition, as a general rule, assessments and guidelines claiming to be aligned with the present LEAP guidelines should flag and justify with reasoning any deviations.

Topic	Summary	Section
TARGETED PRODUCTION SYSTEMS		
Livestock supply chains	These guidelines are intended to be relevant to all varieties of livestock species and production systems.	2.2
Life cycle stages: modularity	The guidelines support modularity to allow flexibility in modelling systems. The 3 main stages are feed production, animal production, and primary animal processing. Upstream processes and transportation are also included. Additional guidance is on post-processing stages, in view of their significance to nutrient cycling and potential environmental impacts.	3.2
GOAL AND SCOPE DEFINITION		
Goal and scope of the study	The goal shall be clearly articulated, describing the objective, purpose, intended use, audience and limitations. For LCA study, the inventory of N or P pressure per unit of product will be used for the impact assessment for eutrophication and acidification. For a nutrient use efficiency study, the goal is mostly to understand the dynamics of nutrient flows in livestock supply chains and the efficiency in which nutrient from inputs are converted into useful end products.	3.1
SYSTEM BOUNDARIES		
Life cycle stages	These guidelines cover the system boundary from the cradle-to-primary-processing gate. Additional guidance is given on post-processing stages through to final waste stages. Other guidance of the system boundary are provided in previous LEAP guidelines on feed and animal supply chains (FAO, 2016a,b,c,d).	3.2
Functional unit	Users shall consult other LEAP guidelines on feed and animal supply chains for the definition of the functional unit (FAO, 2016a,b,c,d).	3.2
Nutrient flows to consider	All inputs, outputs, losses, and recycling flows shall be quantified for life cycle stages considered.	3.3
Scale consistent assessment	Quantification of flows shall be done using a 'Tier 2' approach. Tier 1 methods should only be applied for flows which amount to a maximum 1 percent of the total embedded input flows or for which no data for a Tier 2 method is available. The methods for specific supply chains and regional scale assessment are principally the same, even though generic (representative) data might be used for regional scale assessment, whereas measured data should be used for specific supply chain assessments.	3.4
LIFE CYCLE INVENTORY – FEED PRODUCTION		
Collection of data	In general, data sources for N and P flows shall be based on section 10 – Compiling and recording inventory data - of the existing LEAP guidelines for feed and livestock supply chains (FAO, 2016a, b, c, d), which offers guidance on the collection of data.	4.1
FEED PRODUCTION		
Input flows to feed production systems	Data on the following input flows shall be collected: N and P input from atmospheric deposition, seeds, irrigation water and waste water, synthetic fertilizers, manure and other organic residues. Further N input from biological N fixation shall be accounted for. P input from bedrock weathering may be considered zero, unless site-specific data are available.	4.2.2
Output flows from feed production systems	Data on the following output flows shall be collected: N and P in harvested biomass, N volatilization as NH_3 and NO_x, N emissions from biomass burning, N emissions as N_2O and N_2, N and P losses via soil erosion, leaching and run-off.	4.2.3
Internal flows in feed production systems	Data on the following internal flows shall be collected: N and P in crop residues and green manure, soil N and P stock changes.	4.2.4
Allocation between multiple crops in crop sequences	Emissions of nutrients shall be allocated in proportion to the share of nutrient remaining in the soil at a defined cut-off date. The cut-off date is defined as the start of land preparation for a crop. Thus, the temporal boundaries for the allocation of emissions to a feed crop are from land preparation for the feed crop to land preparation for the following crop.	4.2.5.1

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Topic	Summary	Section
Emissions from direct land use change	Emissions of nutrients that are caused by land use change and occur before land preparation for the first crop or grassland should be allocated to the crops grown until a new equilibrium is reached (using a default period of 20 years), allocating 1/20 of the emissions to the crops grown each year.	4.5.2.2.
Field-to-Gate assessment	Losses during harvest, storage and feed processing before the feed is sent to the livestock production unit, as well as emissions from feed processing before the feed is sent to the livestock production unit, shall be quantified.	4.2.6
ANIMAL HUSBANDRY: CONFINED OR HOUSED, GRAZING AND MIXED ANIMAL SYSTEMS		
Input flows to animal husbandry systems	Data on the following input flows shall be collected: N and P in animal intake, bedding material, and live animal inputs.	4.3.2
Output flows from animal husbandry systems	Data on the following output flows shall be collected: N and P in body live weight and animal products, and N and P in excreta and manure, and losses from manure in gaseous forms (NH_3, NO_x, N_2O, N_2) or losses through leaching and runoff from manure management and grazing systems.	4.3.3
Allocation of emission to manure	Manure shall be considered as a co-product, with some exceptions (i.e. landfilling or dumping, application in excess of crop need, incineration without energy recovery). Allocation of upstream emissions of livestock production systems between manure that leaves the production system and animal co-products can be done with two methods. In most cases, method 1 (biophysical allocation) will be preferable due to its robustness and simplicity. However, it is recommended that when sufficient data is available, method 2 (economic allocation) is evaluated.	4.3.4
ANIMAL PROCESSING		
Animal processing	Data on the following flows in animal processing systems shall be collected: N and P in products, and N and P in residues and waste including recycling flows and emissions to the environment, e.g. from waste water treatment plants.	4.4
UPSTREAM PROCESSES AND TRANSPORTATION		
Upstream processes and transportation	Data on the following flows shall be collected: N and P emissions from the production of fertilizers and consumables, generation and use of energy in all steps including transportation.	4.5
LIFE CYCLE IMPACT ASSESSMENT		
Eutrophication	CML method should be used for the generic midpoint assessment of eutrophication potential (aquatic and terrestrial). Due to its absence of fate and effects modelling of nutrient emissions, CML is considered a Tier 1 approach. In case this impact category is a hotspot, additional efforts shall be undertaken to characterize more the impact in the region or location receiving the emissions.	5.3.1
Freshwater Eutrophication	In case the specific region under analysis are known to be P and N limited, CML should be used at midpoint as Tier 1. Further methods such as TRACI, ReCiPe should be used for more robust assessment, but the practitioners shall explain the basis for selection of the final choice of LCIA method(s) used	5.3.2
Marine Eutrophication	ReCiPe 2008 model should be used to evaluate marine eutrophication (midpoint indicator).	5.3.3
Acidification	CML is recommended for the midpoint assessment of acidification potential (aquatic and terrestrial).	5.3.4
RESOURCE USE ASSESSMENT		
	Resource use shall be assessed based on the Life-Cycle Material Use Efficiency concept, building on the concepts of “inputs” and “useful outputs”.	6
Nutrient use efficiency at each production stage	Nutrient use efficiency (NUE) at each stage or process of a supply chain shall be calculated as the total of N or P in useful outputs (products, recycled nutrients, and stock changes) divided by the total of N or P in external or recycled inputs.	6.1
Life cycle nutrient use efficiency	Life Cycle NUE shall be calculated as one unit of nutrient in the sum of products of the ‘last’ stage of a supply chain that produced the end-products of interest, divided by the amount of external nutrient additions to the supply chain to produce it.	6.2

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Topic	Summary	Section
INTERPRETATION OF RESULTS		
Data quality	Comprehensive assessment of nutrient flows involves the collection and integration of data regarding the products, process or activity under study. This data is gathered from different sources; as such, the management of data quality shall be an integral part of the overall process.	7.1
Significant issues	The results of inventory and impact assessment phases are structured to help determine the significant issues in accordance with the goal and scope definition. The main contributors to the inventory and impact assessment can be assessed through contribution analysis. The contribution of the methodological choices (e.g. allocation rules, assumptions) should be assessed.	7.2
Evaluation	Evaluation shall be performed to establish and enhance the confidence in, and the reliability of, the results of the inventory and LCA. The evaluation involves a completeness check, sensitivity check in combination with scenario analysis and uncertainty analysis, and consistency check.	7.3
ADDITIONAL INDICATORS TO SUPPORT THE INTERPRETATION OF NUTRIENT BUDGET ANALYSIS		
	Several specific N and P indicators are commonly used to inform environmental policies and improving farm practices and livestock supply management. Additional N and P indicators, therefore, should be quantified to enhance interpretation of the results.	7.4
Nitrogen and phosphorus footprints	N and P footprints shall be calculated as the sum of emissions that are caused by the production of one unit of final product, considering all processes, scaled so that the quantity of intermediate products produced equals the quantity required in subsequent supply chain stages.	7.4.1
Gross nutrient surplus	The gross nutrient surplus (GNS) indicator is an agri-environmental indicator used as a proxy for agricultural pressure on the environment from agricultural production. It is calculated be calculated as the difference between total nutrient inputs and total nutrient outputs at a system or production unit level.	7.4.2
Circularity indicator	The nutrient circularity indicator from a perspective of either input or output flows shall be calculated as the quotient of recycled inputs over total inputs, or recycled outputs or residues over total outputs, respectively.	7.4.3
Conclusions, recommendations and limitations	Conclusions derived from the study should summarize supply chain "hotspots" derived from the contribution analysis and the improvement potential associated with possible management interventions. Conclusions should be given in the strict context of the stated goals and scope of the study, and any limitation of the goals and scope can be discussed a posteriori in the conclusions.	7.5

LEAP and the preparation process

The LEAP Partnership is a multi-stakeholder initiative launched in July 2012 with the goal of improving the environmental performance of livestock supply chains. Hosted by the Food and Agriculture Organization of the United Nations, LEAP brings together the private sector, governments, civil society representatives and leading experts who have a direct interest in the development of science-based, transparent and pragmatic guidance to measure and improve the environmental performance of livestock products. The first phase of the LEAP Partnership (2013-2015) focused mainly on the development of guidelines to quantify the greenhouse gas (GHG) emissions, energy use and land occupation from feed and animal supply chains as well as on the principles for biodiversity assessment. The second phase (2016-2018), known as LEAP+, broadened the scope and is focusing on water footprinting, nutrient flows and impact assessment, soil carbon stock changes, quantification of the impact of livestock on biodiversity, etc.

In the context of environmental challenges such as climate change and increasing competition for natural resources, the projected growth of the livestock sector in the coming decades places significant pressure on livestock stakeholders to adopt sustainable development practices. In addition, the identification and promotion of the contributions that the sector can make towards a more efficient use of resources and better environmental outcomes is also of great significance.

Currently, many different methods are used to assess nutrient flows and their associated environmental impacts as well as the performance of livestock products. This may raise confusion and makes it difficult to compare results and set priorities for continuing improvement. With increasing demands in the marketplace for more sustainable products, there is also the risk that debates about how sustainability is measured will distract people from the task of making real improvement in environmental performance. There is the added danger that either labelling or private standards based on poorly developed metrics could lead to erroneous claims and comparisons.

The LEAP Partnership addresses the urgent need for a coordinated approach to develop clear guidelines for environmental performance assessment based on international best practices. The scope of LEAP is not to propose new standards but to produce detailed guidelines that are specifically relevant to the livestock sector and to refine guidance concerning existing standards. The three groups that form the LEAP Partnership, have an equal say in deciding work plans and approving outputs from LEAP, thus ensuring that the guidelines produced are relevant to all stakeholders, widely accepted and supported by scientific evidence.

The work of LEAP is challenging yet vitally important to the livestock sector. The diversity and complexity of livestock farming systems, products, stakeholders and environmental impacts can only be matched by the willingness of the sector's practitioners to work together to improve performance. LEAP provides the essential backbone of robust measurement methods to enable assessment, understanding and improvement in practice. More background information on the LEAP Partnership can be found at: www.fao.org/partnerships/leap/en/

Nutrient TAG and the preparation process

The nutrient TAG of the LEAP Partnership was formed in April 2016. The core group included 38 experts in animal sciences, crop sciences, soil sciences, life cycle assessment, environmental science, and livestock production systems. Their backgrounds, complementary between systems and regions, allowed them to understand and address different perspectives. The TAG was led by Stewart Ledgard (AgResearch, New Zealand) and Adrian Leip (EU Joint Research Centre, Italy), who were assisted by Aimable Uwizeye (FAO, Rome, Italy), Technical Secretary of the TAG. The role of the TAG was to:

- develop guidelines to quantify nutrient flows in livestock supply chains;
- develop guidelines to quantify the environmental impact of eutrophication and acidification;
- select the relevant indicators to understand the nutrient use and associated environmental impacts in livestock supply chains.

The TAG met in two workshops. The first one was held from 12 to14 July 2016 at FAO, in Rome, Italy, and the second one was held from 16 to18 November 2016 at Nobleza Hotel, in Kigali, Rwanda. Between the workshops, the TAG worked via online communications and teleconferences.

PERIOD OF VALIDITY

It is intended that these guidelines will periodically be reviewed to ensure the validity of the information and methodologies on which they rely. At the time of development, no mechanism is in place to ensure such review. The user is invited to visit the LEAP website to obtain the latest version at: www.fao.org/partnerships/leap

STRUCTURE OF THE DOCUMENT

This document adopts the main structure of ISO 14040:2006 and the four main phases of the Life Cycle Assessment – goal and scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA), and interpretation. Part 2 of this methodology sets out the following:

- Section 1 describes the objectives and intended users.
- Section 2 describes scope and impact categories covered.
- Section 3 includes requirements and guidance to help users define the goals and scope, and system boundary of the study.
- Section 4 presents requirements and guidance on the collection and assessment of the quality of inventory data as well as the equations for inventory.
- Section 5 outlines the life cycle impact assessment and recommendations.
- Section 6 provides additional indicators for resource use assessment.
- Section 7 provides guidance on the interpretation and summarizes the various requirements and best practice for reporting, including the uncertainty analysis.

A glossary providing a common vocabulary for practitioners has been included. Additional information is presented in the appendices.

Throughout the document, we refer to several case studies and appendices. The case studies are not intended to be representative of the global distribution of livestock systems, nor are they necessarily representative of all aspects of nutrient flows in global livestock systems. Nevertheless, they do provide useful and practical examples of nutrient use assessment. Most importantly they serve to highlight nutrient use and impact assessment indicators and methods that have been used to assess nutrient flows in contrasting livestock supply chains.

PRESENTATIONAL CONVENTIONS

These guidelines are explicit in indicating which requirements, recommendations, and permissible or allowable options users may choose to follow. The term "shall" is used to indicate what is required for an assessment to conform to these guidelines. The term "should" is used to indicate a recommendation, but not a requirement. The term "may" is used to indicate an option that is permissible or allowable. Commentary, explanations and general informative material (e.g. notes) are presented in footnotes and do not constitute a normative element. Examples illustrating specific areas of the guidelines are presented in boxes.

PART 1

OVERVIEW AND GENERAL PRINCIPLES

1. Objectives and intended users

The methodology and guidance developed here can be used by stakeholders in all countries to assess the sustainability of nutrient use in livestock supply chains. The guidelines are directed at individuals or organizations with a good working knowledge of environmental assessment of livestock systems based on life cycle thinking. The main purpose of the guidelines is to provide sufficient definition of calculation methods and data requirements to enable consistent assessment of nutrient flows and associated impacts in livestock supply chains.

This guidance is relevant to a wide array of livestock stakeholders including:

- livestock producers who wish to develop inventories of their nutrient use and to have the environmental performance of their production systems assessed.
- supply chain partners such as feed processors, livestock farming organizations, processors of animal products as well as retailers pursuing a better understanding of the environmental performance of their production processes.
- policy makers interested in developing nutrient use accounting and reporting specifications for livestock supply chains.

The benefits of this approach include:

- use of recognized, robust and transparent methodology developed to take account of the specificity of nutrient use in contrasting production systems;
- identification of nutrient loss hotspots and opportunities to improve supply chain performance and to reduce environmental impacts;
- identification of opportunities to increase efficiency and productivity;
- Ability to benchmark performance internally or against industry or government standards;
- supporting reporting and communication requirements; and
- raising awareness and supporting action on environmental sustainability.

2. Scope and impact categories

Nutrients are essential elements for the growth of organisms and thus must be added intentionally to the production chain of products based on living substrates if they are not available in sufficient quantity or quality for production to cover the nutritional demand of livestock. Nitrogen (N) and phosphorus (P) are of relevance as they belong to the four elements with (global) biogeochemical cycles (N, P, Carbon and water), which are regularly recycled around the planet at various temporal scales. During the industrial age, these previously stable and self-sustained cycles were perturbed. This is of concern for N and P, which contribute to agricultural production, but also for many environmental and socio-economic impacts. Biogeochemical flows, encompassing both N and P cycles, are two of the planetary boundaries that has been surpassed (Steffen *et al.*, 2015). Additionally, P natural resources are becoming depleted due to human activities.

In contrast to the assessment of livestock supply chains with a focus on impact categories, a more thorough description of all flows involved is required when the area of concern is the assessment of impacts on nutrient cycles. This assessment includes not only the flows that directly lead to the emission of a pollutant, but also others which divert nutrients from the product. The analysis of these flows offers potential opportunities to improve nutrient management and thus increase nutrient use efficiency and reduce environmental impacts.

The existing LEAP Guidelines on animal feed and animal supply chains (FAO, 2016a, 2016b, 2016c, 2016d) focus on the assessment of greenhouse gas emissions and quantification of resource use (e.g. fossil energy use) during the production of feed materials and animal products. They also include associated environmental impacts (mainly climate change). The animal feed and large ruminants' guidelines provide additional recommendations on other impact categories, including eutrophication and acidification, but they do not give detailed recommendations on the estimation of nutrient flows and losses along livestock supply chains. Due to the inherent characteristics of nutrients (particularly N and P) to cycle within the environment and techno-sphere, the environmental assessment of livestock supply chains should account for the impacts linked to losses of polluting nutrient forms, and the efficiency with which nutrients are used in the supply chain.

The objective of this document is to provide additional recommendations to the existing feed and livestock supply chain guidelines by including recommendations for the life cycle impact assessment of livestock supply chains, including methods to estimate flows of N and P. Environmental impact categories are restricted to acidification and eutrophication (freshwater, marine and terrestrial). It is also recognised that N and P losses to water, soil and air play a dominant role in ozone depletion, soil quality impoverishment, or biodiversity loss. These environmental impacts, however, are not covered in this guideline. The impact of nutrients on biodiversity is covered in the LEAP principles on biodiversity, whereas the assessment of the impact of nitrous oxide (N_2O) on ozone is excluded in these guidelines because of strong interactions between N_2O and other greenhouse gases such as CO_2 and CH_4, which are not covered in these guidelines. This document also provides additional indicators for nutrient use efficiency (NUE) along the livestock supply chains (e.g. life-cycle nutrient use efficiency, Uwizeye *et al.*, 2016). In many studies, this indicator is calculated at animal or farm level using farm-gate nutrient balance (e.g. Powell *et al.*, 2010). Details for the assessment of climate change impacts have already been covered in the existing guidelines, although this document provides additional guidance on the calculation of emissions of N_2O. Several specific N and P indicators (e.g. N and P surplus, N and P footprints) are commonly used to inform environmental policies and improving farm practices and livestock supply management. These indicators are also discussed. Regarding the impact assessment, the potential impact of particulate matter and photochemical ozone formation are also excluded from these guidelines. This document does not provide guidance on full assessment of environmental performance, nor on the social or economic aspects of livestock supply chains.

2.1 APPLICATION

Some flexibility in methodology is desirable to accommodate the range of possible goals and special conditions arising in different sectors. This document strives for a pragmatic balance between flexibility and rigorous consistency across scale, geographic location, and project goals.

A stricter prescription on the methodology, including allocation and acceptable data sources, is required for product labelling or comparative performance claims. Users are referred to ISO 14025 for more information and guidance on comparative claims of environmental performance.

These LEAP guidelines are based on the attributional approach to life cycle accounting. The approach refers to process-based modelling, intended to provide a static representation of average conditions.

Due to the limited number of environmental impact categories covered here, results should be presented in conjunction with other environmental metrics to understand the wider environmental implications, either positive or negative. It should be noted that comparisons between final products should only be based on full life cycle assessment. Users of these guidelines shall not employ results to claim overall environmental superiority of livestock production system over another.

The methodology and guidance developed in the LEAP Partnership is not intended to create barriers to trade or contradict any WTO requirements.

2.2 LIVESTOCK SPECIES AND PRODUCTION SYSTEMS

These guidelines are intended to be relevant to all varieties of livestock species and production systems.

2.3 NORMATIVE REFERENCES

The following referenced documents are indispensable in the application of this methodology and guidance.

- ISO 14040:2006 *Environmental management – Life cycle assessment – Principles and framework* (ISO, 2006a)
- These standards give guidelines on the principles and conduct of LCA studies, providing organizations with information on how to reduce the overall environmental impact of their products and services. ISO 14040:2006 define the generic steps which are usually taken when conducting an LCA, and this

document follows the first three of the four main phases in developing an LCA (Goal and scope, Inventory analysis, Impact assessment and Interpretation).

- ISO14044:2006 Environmental management – Life cycle assessment – Requirements and guidelines (ISO, 2006b)
- ISO 14044:2006 specifies requirements and provides guidelines for life cycle assessment including: definition of the goal and scope of LCA, LCI, LCIA, the life cycle interpretation, reporting and critical review of the LCA, limitations of the LCA, relationship between the LCA phases, and conditions for use of value choices and optional elements.
- ISO 14025:2006 Environmental labels and declarations – Type III environmental declarations – Principles and procedures
- ISO 14025:2006 establishes principles and specifies procedures for developing Type III environmental declaration programmes and Type III environmental declarations. It specifically establishes the use of ISO 14040:2006in the development of Type III environmental declaration programmes and Type III environmental declarations.

PART 2

METHODOLOGY FOR QUANTIFICATION OF NUTRIENT FLOWS AND ENVIRONMENTAL IMPACTS FOR EUTROPHICATION AND ACIDIFICATION IN LIVESTOCK SUPPLY CHAINS

3. Goal and scope definition

3.1 GOAL AND SCOPE OF THE STUDY

The first step required when initiating a nutrient flow's analysis study is to clearly set the goal or statement of purpose. This statement describes the goal pursued and the intended use of results. The goal can be to perform an LCA for N or P flows or to analyse the N or P use efficiency in livestock supply chains. In case of an LCA, the inventory of nutrient pressure per unit of product will be used as input in the impact assessment for eutrophication and acidification. This assessment would serve the goal of nutrient use management or to understand the nutrient loss hotspots to prioritise the management interventions along the supply chains. For a nutrient use efficiency study, the goal would be to understand the dynamics of nutrient flows in livestock supply chains and the efficiency in which nutrient from inputs are converted into useful end products. This assessment is important for benchmarking and monitoring improvement and can support reporting on nutrient losses/pressures. This approach can also be used to inform environmental policy and best practices. It is therefore of paramount importance that the goal and scope is given careful consideration because these decisions define the overall context of the study. A clearly articulated goal helps ensure that aims, methods and results are aligned. For example, fully quantitative studies will be required for benchmarking or reporting, but somewhat less rigour may be required for hotspot analysis. Finally, approaches used will depend on the goal and scope, system boundary, scale and site characteristics.

Interpretation is an iterative process occurring at all steps of the nutrient flow assessment and ensuring that calculation approaches and data match the goal of the study (see section 7). Interpretation includes completeness checks, sensitivity checks, consistency checks and uncertainty analyses. The conclusions (reported or not) drawn from the results and their interpretation shall be strictly consistent with the goal and scope of the study.

3.2 FUNCTIONAL UNIT AND SYSTEM BOUNDARY

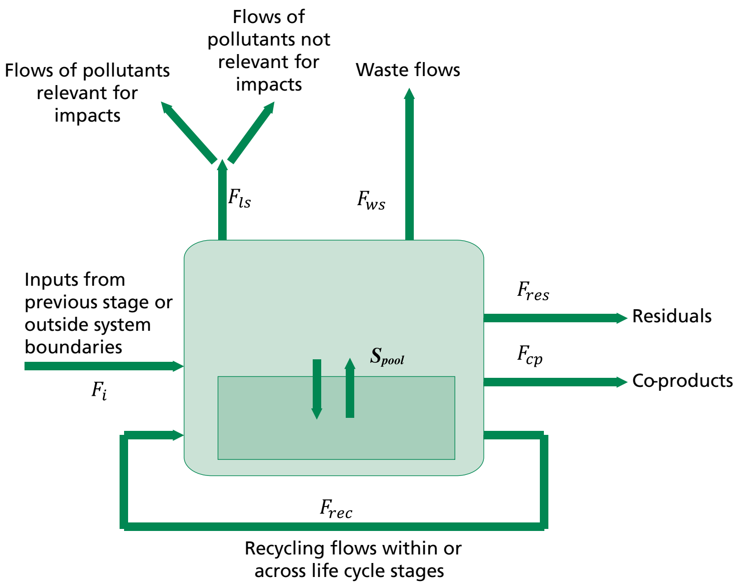
These guidelines cover the system boundary from the cradle-to-primary-processing gate, representing the life cycle stages detailed in the existing LEAP guidelines. However, additional guidance is provided on post-processing stages, in view of their significance to nutrient cycling and environmental impacts. Regarding the functional unit for LCA, consult LEAP guidelines (FAO, 2016a, 2016b, 2016c, 2016d).

3.3 NUTRIENT FLOWS TO CONSIDER

Figure 1 shows schematically one stage in the life cycle of a project indicating which kind of flows must be quantified:

- **Input flows** F_i include both flows that link the life cycle stage with previous stages (carrying on to the product(s) and new input flows required. Based on a modular LCA, input flows carry with them all upstream burdens and are thus equivalent.

Figure 1
Generalised diagram showing relevant flows for an individual life cycle stage; for the explanation of the acronyms, please see the main text



- **Output flows** in terms of co-products F_{cp} carrying burden to the next stage and residual flows F_{res} that have further use but do not carry the burden with them.
- **Loss flows** that carry nutrients out of the system boundaries without leading to any benefit are nutrient losses F_{ls}. This includes emission flows (F_{em}) that are losses of nutrients to the environment (both atmosphere and hydrosphere). Emissions such as non-reactive N (N_2) do not cause any environmental impact; emissions of reactive N (all other forms of un-locked N compounds[1]; Nr) or P that is not re-captured and purposefully used in a supply chain are relevant for environmental impact. Nutrient losses also include waste flows (F_{ws}) which might generate further emissions that are to be considered in the burden allocated to co-products. Waste flows include food losses and wastes (HLPE, 2014) that are not recycled. Nutrient losses are the sum of nutrient emissions and nutrient wastes; $F_{ls} = F_{em} + F_{ws}$.
- **Recycling** flows F_{rec} are used in a supply chain; this can include composted, digested or incinerated food losses or wastes, sewage sludge, wastewater, or re-captured emissions of Nr and P that are recovered and reused. Recycling flows can be classified as either co-products or residual flows.

The distinction between loss and recycling flows is often difficult, and the quantification of the share of "potential" recycling flows which is actually recycled is a challenge and is addressed in this document. For example, data on communal organic waste is not easily available. Similarly, the estimation of atmospheric deposition from an agricultural origin that serves as a fertilizer is complex; the effect of

[1] Locked nitrogen is nitrogen bound e.g. to fossil fuel which is not available to organisms but is "activated" when the fuel is burned. See definition of terms in UN-ECE (2013).

riparian and wetland zones for removing aquatic and atmospheric pollutants is of particular challenge. These examples are important "mechanisms" to improve the nutrient efficiency of livestock supply chains and reducing adverse effects.

All flows of the budget must thus be quantified (Tier 2, see Appendix 1 for details on the Tier levels) so that their balance is "closed" according to Equation 1 (see UNECE, 2013).

Equation 1

$$F_o + S_{pool} - F_i = 0$$

with the total output flow (F_o) calculated as $F_o = F_{cp} + F_{res} + (F_{em} + F_{ws})$ as indicated in Figure 1, and S_{pool} being the stock changes of the pool (generally also regarded as "useful" output [Leip *et al.*, 2011b]). For a feed production system, stock changes refer mainly to nutrients in the soil. In practice, a budget is often unbalanced due to (i) data gaps (ii) inconsistent data sources, or (iii) knowledge gaps leading to the omission of relevant flows.

3.4 SCALE CONSISTENT ASSESSMENT

Recommendations in these guidelines cover:

- specific supply chain assessment (e.g. cradle to farm gate)
- regional scale assessment.

Recommendations are given for "Tier 2" methodologies (see Glossary and Appendix 1), while default values ("Tier 1") are suggested for certain flows in additional appendices. However, efforts shall be undertaken to use the Tier 2 methods, as Tier 1 methods should only be applied for flows which amount to a maximum 1 percent of the total embedded input flows and for which no data for a Tier 2 method is available. If available, Tier 3 methods can provide the most accurate estimates. Tier 3 methods usually are based on process-based simulation models that run at higher temporal resolution. Tier 3 models must be widely accepted by peer-reviewed publications. If a Tier 3 model is available, validation of the model for conditions encountered in the supply chain assessed must be demonstrated.

The methods for specific supply chains and regional scale assessment are principally the same, even though generic (representative) data might be used for regional scale assessment, whereas measured data should be used for specific supply chain assessments. For most of the nutrient flows that need to be quantified in feed supply chains, existing guidelines have defined relevant methods. These include previous LEAP guidelines and guidelines for reporting of GHG inventories IPCC (2006), air pollution inventories (EEA, 2016), Gross Nutrient Balances (Eurostat 2013), and national N budgets for agriculture (Leip *et al.*, 2016). Details are given in Appendix 2.

4. Life cycle inventory

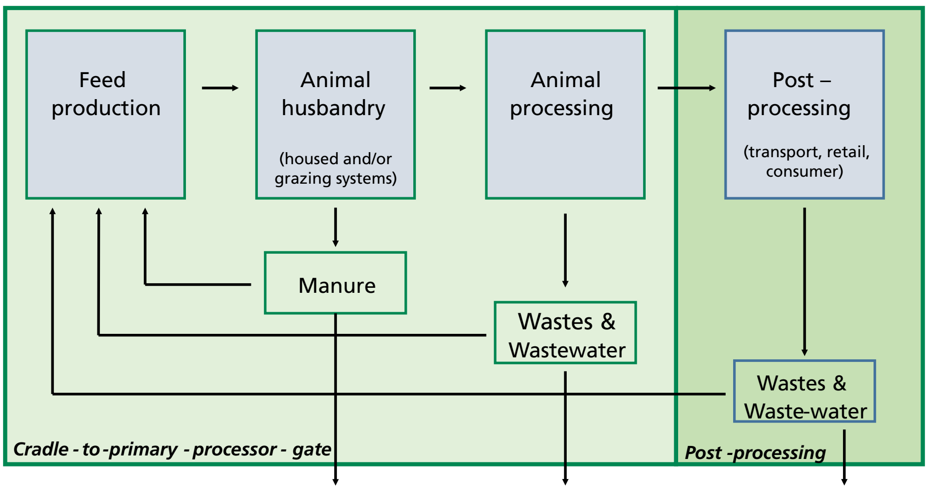
4.1 OVERVIEW

LCI analysis involves the collection and quantification of inputs and outputs throughout the life cycle stages covered by the system boundary of a study. These guidelines refer to quantification of nutrient flows, covering inputs, products, recycling and losses, and refer to the existing LEAP guidelines for animal feed and livestock supply chains (for small ruminants, large ruminants, poultry and pigs) whenever possible. The most recent existing guidelines were organised in a modular structure so that animal feed guidelines covered all stages from the production of feeds to the animal's mouth, while livestock supply chain guidelines covered the animal production and primary processing stages.

These current guidelines are similarly structured so that they align to the existing animal feed and livestock supply chain guidelines. They are structured in relation to the production of feed and livestock production systems for housed animals and for grazing animals (Figure 2), followed by sections covering animal processing, post-processing life cycle stages to the final waste stage, and upstream processes.

For the compilation of the inventory data, methods for collection of data for N and P flows are described in the subsequent sections, but in general, data sources for N and P flows shall be based on section 10 – Compiling and recording inventory data - of the existing LEAP guidelines for feed and livestock supply chains (FAO, 2016a, b, c, d), which offers guidance on the collection of primary data. Where primary data cannot be obtained, relevant data from models or scientific literature can be used and their basis justified. Data for N and P flows from technosphere should include information on practices and be country- or site-specific.

Figure 2
Generalised system diagram showing the life cycle stages covered in these guidelines



The environmental assessment of the impact and resource efficiency dimensions is discussed in the following sections. Finally, guidance is given on the interpretation of results.

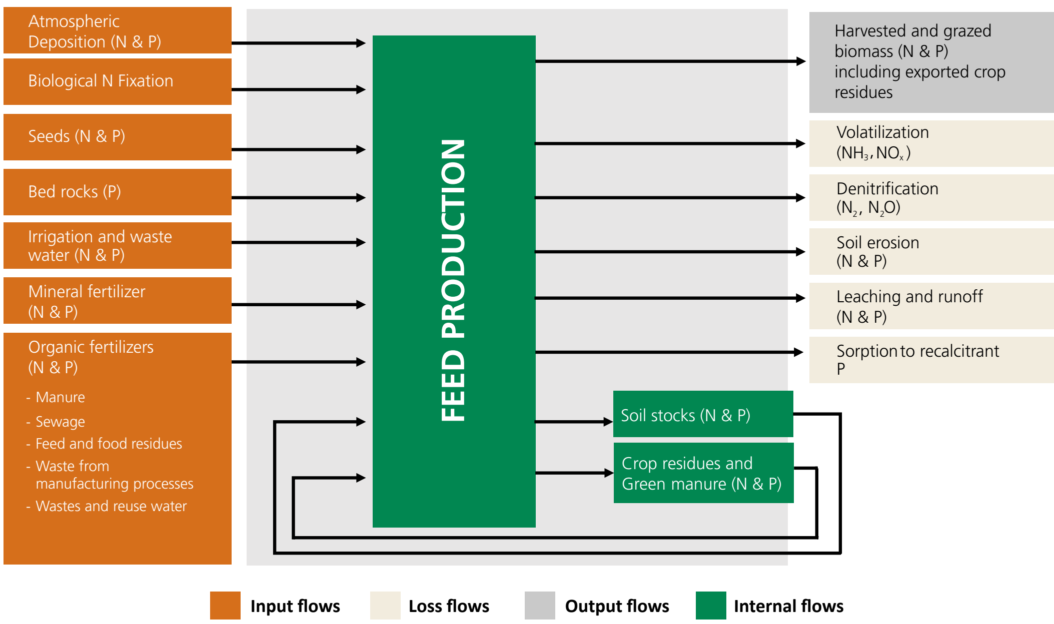
4.2 FEED PRODUCTION

4.2.1 Introduction

Feed production systems are a relevant part of agricultural systems across the world, and they are a critical part of livestock supply chains. Details on feed types, systems, and material flows have been covered in the LEAP Environmental Performance of Animal Feed Supply Chains guidelines. The soil-crop continuum is a highly complex system in which inputs of nutrients undergo many transformation processes. Figure 3 shows a schematic representation of relevant N and P flows in feed production systems. System boundaries extend to the soil and below the root zone. Temporal boundaries are from the land preparation for the feed crop to the land preparation for the following crop (see Section 4.2.5.1).

Only some of the nutrients available from external input or release from soil pools are used by the feed crop. Nutrient turnover in soils is mainly driven by microbiological processes; some of them (e.g. mineralisation, residual N and P in soluble forms, and P solubilization) improve N and P availability to plants for uptake, but also increase the chances for losses to the environment. Conversely, other processes such as immobilization of N with organic inputs of high C:N ratio (>25), immobilization of N and P by soil microorganisms, and P sorption would temporarily reduce the availability of N and P for plant uptake and loss to the environment. Physico-chemical processes lead to losses of N and P from soils.

Figure 3
Diagram of relevant N and P flows in feed production systems



This includes gaseous emissions (N), volatilization (N), leaching (N, P), runoff (N, P) and erosion (N, P). Relevant flows are shown in Figure 3.

The quantification of these N and P flows may have high uncertainties. The practitioner should collect additional information on these uncertainties and quantify their impact on the results based on the recommendations in Section 7.

4.2.2 Input flows to feed production systems

4.2.2.1 Nitrogen and phosphorus from atmospheric deposition

Inputs of nutrients from atmospheric deposition include both wet and dry deposition of N and deposition of P in aerosols or dust.

Data on deposition rates shall be collected in kg N ha^{-1} yr^{-1} or kg P ha^{-1} yr^{-1}. For wet deposition, the concentration of N in precipitation in mg L^{-1} should be multiplied by the total precipitation in L during the feed production reference period (between the start of land preparation of feed crop and the start of land preparation of the following crop). Constant deposition rates of N in dry deposition and of P can be assumed, taking into consideration that deposition rates vary with the land cover (Simpson *et al.*, 2011). Global deposition maps are available for N (Dentener, 2006) and P (Mahowald *et al.*, 2008). Gridded maps exist as well, for example for the region covered by the UNECE (Simpson *et al.*, 2014). For deposition of P in dust, global maps indicating areas susceptible to P deposition and P concentrations in dust are available (Das *et al.*, 2013).

4.2.2.2 Biological nitrogen fixation

N_2 fixation from the atmosphere is achieved by rhizobia bacteria, in most cases in nodules associated with legume roots. All legumes can fix N, but some are more efficient than others, and the maximum percentage of N derived from fixation by legumes varies between species, from about 65 percent (e.g. bean) to 100 percent (most fodder legumes e.g. alfalfa, clovers). N fixation rates in grasslands depend on grazing management (grazing vs. cutting), external sources of mineral N, and the share of legumes in the field (Høgh-Jensen and Schjoerring, 1997; Ledgard, 2001; Ledgard *et al.*, 2001; Vinther, 1998). Furthermore, free-living N_2-fixing organisms can provide additional input of N.

The Tier 2 approach consists of calculating N_{fix} (kg N ha^{-1}) by multiplying crop yield Y [kg dry biomass ha^{-1}] by the concentration of N in the crop C_N (kg N [kg dry biomass]$^{-1}$) and the fraction of total crop-N that is derived from atmospheric N-fixation N_{dfa} (Equation 2). To account for fixed N in non-harvested biomass (e.g. plant material below cutting/grazing height and roots), a "whole-plant-factor" F_{yield} is also used (Anglade *et al.*, 2015; Jørgensen and Ledgard, 1997; Appendix 3, Table A3.1).

Equation 2

$$N_{fix} = Y \cdot C_N \cdot N_{dfa} \cdot f_{yield}$$

For humid and tropical climates, N-input from free-living organisms can be substantial and shall be considered as well (see Appendix 3).

Legume yield in grazing pasture systems can be estimated based on pasture intake by animals $GP_{intake,}$ (kg dry biomass ha^{-1}), a utilization factor $f_{untilizadion}$ and an estimated share of legumes in the pasture $f_{legumes}$.

Equation 3

$$Y_{legumes} = GP_{intake} \cdot f_{untilizadion} \cdot f_{legumes}$$

A method for GP_{intake} is given in the LEAP Guidelines for small and large ruminants (FAO, 2016b, c). The $f_{untilizadion}$ and $f_{legumes}$ shall be estimated for the studied system; typical values are given in Appendix 3. The $f_{untilizadion}$ factor recognizes that intake by animals is less than the amount of pasture production (e.g. approximately 50-80 percent, giving $f_{utilization}$ factor of 1.25-2.0) and an estimate of total pasture production requires accounting for this to avoid underestimation.

Default Tier 1 data is available (Herridge *et al.*, 2008; Peoples *et al.*, 2009) and should be used only when Tier 2 data is unavailable and N fixation is minor.

4.2.2.3 Nitrogen and phosphorus from seeds

Data on seed plant material as kg ha^{-1} should be collected (see section 11.2.3a in FAO, 2016a) and multiplied by its nutrient content in kg N (kg seed)$^{-1}$ or kg P (kg seed)$^{-1}$. The N and P content varies among plant species (e.g. Lamont and Groom, 2013).

4.2.2.4 Phosphorus from bedrock weathering

Weathering of rock can release P into the soil system. It is a slow process (Gardner, 1990) and can be estimated from the geological assessment of bedrock, including its P content (percent) (Hartmann *et al.*, 2014). Using available data from three basins, Gardner (1990) reported P release from rock into the soil system to represent 25 to 50 percent of the P from atmospheric deposition. Young soils may contain natural apatite and provide a larger contribution from weathering. Hence P from rock could be of agronomic significance depending on the geochemical processes in the reference area.

Most of the guidelines for P inputs did not include P from rock weathering. This could be due to the assumption that this P release could be very slow and negligible relative to the overall P budget in the soil system for relatively short periods, particularly when P input from various fertilizer products is high enough to meet the crop P requirement. No Tier 1 or Tier 2 method is available. Thus, it can be considered as zero unless country-specific or site-specific data is available. Alternatively, estimates of P release by weathering could be based on values derived for various regions globally of between 0.1 and 0.7 kg P ha^{-1} yr^{-1}, varying with rock type and site conditions (Hartmann *et al.*, 2014).

4.2.2.5 Nitrogen and phosphorus in irrigation water including wastewater

Irrigation water may contain a significant amount of N, which should be considered in the fertilization programme. For crop production, restrictions on the use of irrigation water might apply, e.g. at high nitrate concentrations (Abrol *et al.*, 1988; Bauder *et al.*, 2011).

Data on N and P input in irrigation water shall be collected by multiplying applied volumes of irrigation water in L ha^{-1} yr^{-1} by its nutrient content in kg N L^{-1} or kg P L^{-1}.

4.2.2.6 Nitrogen and phosphorus from synthetic fertilizers

N and P synthetic fertilizers, also known as inorganic fertilizers, are intentionally applied to both feed and food crops to improve nutrient availability. The formulations are solid (powder or granules) or liquid. Depending on the storage conditions, application techniques and weather conditions, N and/or P can be lost before being available to plants.

The feed guidelines (FAO, 2016a) recommend that data on application rate of synthetic N and P fertilizers shall be collected, expressed as kg N or P ha^{-1} yr^{-1}. The Tier 2 approach consists of the collection of fertilizer application rates by fertilizer type and feed type. This information can be deduced from the fertilizer "label" or through laboratory analysis. Depending on methods available to quantify further flows, the application technique, a form of application (e.g. coated, together with urease or nitrification inhibitors), timing and placement of applications, should be collected concurrently. Some countries may have fertilizer recommendations which determine a quantity of fertilizer that is given to crops, often as a function of previous fertilizer applications, soil type, and climate. In case no crop-specific fertilizer application data is available, recommended application rates that fit with the specific situation should be used. Additional information on synthetic fertilizer application is described in the LEAP global database of GHG emissions related to feed crops[2]. Further details on regional assessment are given in Eurostat (2013).

4.2.2.7 Nitrogen and phosphorus from manure

Availability of N and P from manure for crop uptake depends on temperature and moisture, manure type (animal type and housing and storage systems), and the existence of pre-treatment during storage and degree of manure decomposition during the storage period. In general, between 30 percent and 90 percent of the total N content of solid manures and slurries is present in organic forms (e.g. Goss *et al.*, 2013).

Data on nutrients intentionally applied with manure or deposited by grazing animals in kg N ha^{-1} yr^{-1} or kg P ha^{-1} yr^{-1} shall be collected. Nutrient content is to be considered net of nutrient losses occurring in housing and manure storage and treatment systems. Methods are provided in section 4.3.

The Tier 2 approach consists of the collection of nutrient input rates by manure type and feed type. Depending on the methods available to quantify further flows, the application technique (spreading, incorporation, etc.), form of application (e.g. together with nitrification inhibitors), and timing of applications should be collected concurrently. Countries may have nutrient policies which determine upper limits for manure applications. In case no crop-specific nutrient application data is available, recommended application rates that fit with the specific situation should be used. For additional information on regional assessment e.g. Eurostat (2013) or UNECE (2013).

4.2.2.8 Nitrogen and phosphorus from other organic residues

A large variety of organic residues can be applied to soils to support crop production. Besides animal manures, they fall essentially into four main categories, i.e. (i) municipal biosolids and sewage sludge, (ii) feed and food residues and waste (see section 11.3.3. in FAO [2016a]), (iii) waste from manufacturing processes (section 11.3.3. in FAO [2016a]), and (iv) green manure and crop residues (see section 4.2.4.1) (Goss *et al.*, 2013). The detailed description of the use of biosolids as fertilizer in agriculture is provided in Appendix 12.

[2] http://www.fao.org/partnerships/leap/database/ghg-crops/en/

Data on nutrients applied in organic residues in kg N ha^{-1} yr^{-1} or kg P ha^{-1} yr^{-1} shall be collected. Methods are provided in the sections indicated above. The Tier 2 approach consists of the collection of nutrient input rates by residue type and feed type. Depending on the methods available to quantify further flows, the forms of N and P in the product should be differentiated because the N and P forms in the residue to determine the extent of the mineralisation rate. The C:N ratio, application technique (surface application, incorporation, etc.), and timing of applications should be concurrently collected as they influence N and P potential availability. Some countries may have policies that restrict the application of certain organic residues such as municipal biosolids or sewage sludge, caused by the existence of potentially pollutant elements.

4.2.3 Output flows

The intended output flow in feed production systems is the uptake of nutrient in harvested or grazed biomass. Biomass below harvest/grazing height (roots, rhizomes, stolons, and stubble) are not considered as an output if not harvested or grazed. Plant residues, such as straws for cereals and grain legumes, can be exported (outputs) or returned to the soil, as well as lost at harvest. The associated N and P flows shall be taken into account.

4.2.3.1 Nitrogen and phosphorus in harvested biomass

N in crop products and co-products are estimated according to FAO (2016a, section 11.2.3), by multiplying the harvested yield of each co-product by the content of N or P (kg [kg dry biomass]$^{-1}$). Crop protein content data is published annually by governments and global organizations (e.g. FAO statistics). For annual and perennial grasslands, N content varies largely with growth stage, species composition and soil nutrient status, between about 1.5 percent (late hay) to more than 3.5 percent (well N-fertilized or grass-clover pastures); but if management information is not available, a mean value can be considered. N content is less variable for maize silage and most forage crops (e.g. fodder beet, sorghum, fodder rape, etc.), while P content varies between about 0.1 and 0.4 percent[3]. Feedipedia provides information on N and P contents of all feed materials used around the world (Feedipedia, 2012).

4.2.3.2 Volatilization (NH_3, NO_x)

Ammonia (NH_3) emissions from soil occur due to manure application, grazing (excreta deposited on pastures), application of mineral fertilizers, application of other organic fertilizers, post-flowering plant losses, crop residues and field-burning of agricultural wastes. NH_3 emissions are equal to the N amounts that are applied from these N sources multiplied by NH_3 emission factors for each source (IPCC, 2006; Leip *et al.*, 2016; Webb *et al.*, 2014). Ammonia emissions depend on the type of livestock manure and mineral fertilizer type, application technique (Bittman *et al.*, 2014; Webb *et al.*, 2014), soil properties (Goulding *et al.*, 2008), and meteorological conditions.

If no peer-reviewed model to estimate NH_3 and NO_x emissions, validated on site-specific data, or site-specific primary measurement data is available, NH_3 emission factors for each source from the EEA 2016 Guidebook can be used (EEA, 2016).

[3] http://corn.agronomy.wisc.edu/Silage/S006.aspx

Note should be taken of possible mitigation options described in the Framework Code of Good Agricultural Practice for Reducing Ammonia Emission (Bittman *et al.*, 2014; UNECE, 2014). Furthermore, default emission factors are provided in IPCC guidelines (IPCC, 2006).

4.2.3.3 Nitrogen emissions from burning of agricultural residues

The approach to determine the contribution of N emissions from burning agricultural residues is considering the area burnt, dry matter of available crop residue (see section 4.2.4.1), as well as emission and combustion factors for vegetation types. The emission factor of NO_x (in g kg^{-1} dry matter burnt) for agricultural residues is 2.5 (Andreae and Merlet, 2001 referred to in IPCC, 2006). Emission factors for NO_x and NH_3 are also provided by the EMEP/EEA air pollutant emissions inventory Guidebook (EEA, 2016). The mass of residue burnt is calculated from the area burnt, the mass of fuel available for combustion, and a dimensionless combustion factor. Values of the combustion factor for agricultural residues post-harvest are given in the IPCC 2006 guidelines and are 0.8 for maize, rice and sugarcane and 0.9 for wheat.

4.2.3.4 Denitrification (N_2O, N_2)

Microbial nitrification (stepwise oxidation of ammonia to nitrate) and denitrification (reduction of nitrates to molecular dinitrogen, N_2) in agricultural and natural soils represent approximately 70 percent of the global N_2O emissions (Syakila and Kroeze, 2011). Denitrification represents a source of Nr and is one of the largest loss pathways for N in agricultural soils (Leip *et al.*, 2015, 2011a). Emissions of N_2O are highly variable in space and time and depend on the N source, a large number of management practices, soil type and meteorological conditions (Butterbach-Bahl *et al.*, 2013, 2011). At the field scale, process-based models can predict N_2O fluxes accurately (Beheydt *et al.*, 2007; Giltrap and Ausseil, 2016; Grosso *et al.*, 2010; Qin *et al.*, 2013), but upscaling to the regional scale remains a challenge (Leip, 2010; Leip *et al.*, 2011c). Measurements of N_2 fluxes are very difficult and costly, and no methodology to estimate them exists (Butterbach-Bahl *et al.*, 2011; Leip *et al.*, 2016).

If no validated peer-reviewed model using site-specific or representative data to estimate N_2O and N_2 emissions or site-specific primary measurement data is available, the N_2O emission factor from the IPCC (2006) may be used. Note that for environmental assessment from a nutrient perspective, only *direct* N_2O emissions shall be quantified, while *indirect* N_2O emissions following leaching and run-off or volatilization of NH_3 and NO_x are required if the impact on climate change is also being studied. Suitable country-specific emission factors and other parameters might be available from national GHG inventories and can be downloaded from the United Nations Framework Convention on Climate Change (UNFCCC) website.

N_2 fluxes should be estimated as a "residual" flow from the soil N-balance (Leip *et al.*, 2016, 2011b; Winiwarter and Leip, 2016). The plausibility of the N_2 flux estimate should be evaluated based on the N_2:N_2O emission ratio (Butterbach-Bahl *et al.*, 2011; Leip, 2011; Seitzinger *et al.*, 2006).

4.2.3.5 Nitrogen and phosphorus losses by soil erosion

The Revised Universal Soil Loss Equation (RUSLE; see Appendix 8) can be instrumental to calculate the N and P losses via soil erosion by water. RUSLE predicts soil losses in a unit of soil mass which should be multiplied by the soil N and P concentrations to obtain the net amount of N and P lost via runoff. Losses of P from soils due to wind erosion can be substantial in agricultural areas with dry climates. However, methods to estimate this loss are not yet available (Katra *et al.,* 2016).

Scherer and Pfister (2015) provide regionalised estimate of P loss to water for 169 crops at the country scale (in kg P [kg crop]$^{-1}$). Their modelling of P from erosion combines the Universal Soil Loss Equation (USLE) model and soil P concentration via the Swiss Agricultural Life Cycle Analysis (SALCA) model. The P erosion component accounts for slope, soil erodibility, a crop factor (effectiveness of a crop to prevent soil loss), a tillage factor and a practice factor (based on the Human Development Index and the Environmental Performance Index for Agriculture). SALCA modelling showed that the site-dependent P concentration in soil was one of the most important parameters influencing P emissions to water from agriculture by erosion.

4.2.3.6 Nitrogen and phosphorus leaching and runoff

The non-gaseous N losses include leaching (nitrate, dissolved organic nitrogen (DON)) and runoff (NH_4^+, N_{org}), while P losses also occur via leaching (phosphate) and mainly runoff (phosphate, organic phosphorus, sediment-bound P). The addition of water in excess of the soil's water-holding capacity leads to the downward transport of N and P in the soil solution. Leaching rates depend on the availability of N and P in soils, the water balance (rainfall and irrigation vs. evapotranspiration), and soil characteristics (in particular depth and texture). Soils with fine texture (high clay content) are in general less susceptible to leaching than those with coarse texture (high sand content) because they are much less permeable to water, although this can be bypassed on clay soils with preferential flow pathways. N and P runoff occur with surface movement of water, which displaces soil sediments and depends on slope, rainfall patterns, soil properties, such as infiltration rates and soil drainage.

If no peer-reviewed model to estimate N leaching and runoff was validated using site-specific or representative data or site-specific primary measurement data is available, leaching fractions ($Frac_{LEACH}$) for humid regions or in drylands which receive irrigation other than drip irrigation, shall be used. In this case, N leaching is calculated according to the IPCC (2006) methodology by multiplying the leaching fraction by the total N input in the various N sources. Conversely, in dry areas where soil water-holding capacity is not exceeded by rainfall, the default values for leaching and runoff are zero for rain-fed cultivation or drip irrigation. In areas characterized by marked differences between rainy and dry seasons, $Frac_{LEACH}$ and N or P leaching should be calculated for each season separately and the quantity of N or P leached added. A suitable country-specific leaching fraction might be available from national greenhouse gas inventories that can be downloaded from the UNFCCC website[4].

Note that the IPCC methodology provides estimates for N leaching plus runoff. Care has been taken to avoid double-counting of losses if losses from water-erosion are estimated according to section 4.2.3.5.

The factors affecting P losses by runoff and leaching are (i) type of P compounds, (ii) soil physical and chemical properties (rock type, hydrology, porosity, depth to

[4] http://unfccc.int/national_reports/items/1408.php

groundwater etc.), (iii) management practices (fertilization program, tillage), and (iv) climatic and environmental conditions (rainfall, drought, erosion, etc.). Dissolved (soluble) and particulate P (eroded soil particles) are the forms of P most susceptible to be lost from soils.

The P index system, which combines source and transport factors, is a tool commonly used to assess P losses to waterways, including from grazed livestock systems (section 4.3.3.6; Appendix 8). It includes P from erosion as well as soluble P losses via runoff + leaching from added sources. It is recommended to follow a country/region/area specific methodology or protocol. Overall, P leaching is considered a minor flow compared to runoff and erosion, and there are no Tier 1 or Tier 2 methods available for it.

When P is intentionally added to excess-fertility soils, soil P accumulation can be more than plant needs. Therefore, this fraction of P may increase the risk of P leaching/runoff; thus, for P accounting, it is considered as lost. An approach for estimating the 'unsustainable' P is described in Uwizeye *et al.* (2016).

4.2.4 Internal flows

4.2.4.1 Nitrogen and phosphorus in crop residues and green manure

After harvest, a part of the crop biomass is left in the field and will partially decompose, releasing N by mineralisation that becomes available to subsequent crops. In the case of forage crops, the stubble can be grazed, and thus a part of the plant will be taken from the field instead of being left to mineralise. Crop residues, i.e. all the plant material left on an agricultural field after harvest, serve several purposes: (i) protection of soils against erosion; (ii) improvement of water retention; (iii) increase of soil organic matter content; and (iv) nutrient recycling.

The rate of mineralisation of crop residues, and thus the availability of its nutrients for subsequent crops, depends on the quality of the residue, such as its lignin content and C:N ratio, soil properties, meteorological conditions and crop management related factors. Straw-based stubble can promote immobilization, thus reducing N availability to plants.

If no primary data on N and P input from crop residues is available, it shall be calculated according to the IPCC (2006) methodology. This method considers the harvested yield [kg dry biomass ha^{-1}] and the fraction of the field area that is not burned and renewed. The nutrient release from above- and below-ground crop residues are obtained from the fraction of above-ground residue to harvested crop and fraction of below-ground residues to above-ground biomass and the respective nutrient contents. Required default values are given in Table 11.2 of Volume 4 for several crops (IPCC, 2006). These default values should be replaced with country-specific data (e.g. Björnsson *et al.*, 2013; Hay, 1995)[5]. Country-specific data might also be available from national greenhouse gas inventories for some countries and can be downloaded from the UNFCCC website[6].

In the case of green manure, no plant biomass is usually removed from the field. Total plant biomass is either mulched or tilled into the soil, instead. To calculate total plant biomass of green manure the same approach may be used provided that the yield of green manure is known (see section 4.2.2.8).

[5] Note that additional factors might be provided in the upcoming IPCC 2019 refinement of the IPCC 2006 guidelines

[6] http://unfccc.int/national_reports/items/1408.php

It is important to consider that for nutrient assessment of livestock supply chains, the nutrient input occurs with crop residues and green manure grown before the feed crop is sown, which could be a different crop. The cut-off date to determine crop reference periods is the start of land preparation. Thus, the period between land preparation for the previous crop to land preparation for the feed crop will determine the emissions from residues that should be allocated to the previous crop, while emissions from land preparation for the feed crop to the land preparation of the following crop should be related to the feed crop. Emissions from green manure shall be entirely allocated to subsequent crops. Nutrient input is, therefore, the content of nutrients in the residue minus the emissions occurring before the cut-off date (see 4.2.5.1).

Emissions from residues of green manure or the previous crop after the cut-off date are accounted as nutrient losses for the current feed crop, as are emissions of crop residues from the feed crop occurring before the next cut-off date. Remaining nutrients in the crop residues at that date are considered as adding to soil nutrient stocks.

4.2.4.2 Soil nitrogen stock changes

N in soil organic matter, residual organic matter from the application of organic fertilizers in previous years, and crop residues that have not been removed from the system occur in different pools to decrease plant availability (Cordovil, 2004):

- inorganic compounds (NO_3^- and NH_4^+);
- readily mineralizable compounds, such as urea and uric acid, which are quickly converted into NH_4^+;
- simple organic compounds mineralizable by soil microorganisms;
- recalcitrant organic compounds, resistant to microbial attack.

The quantity of N in soil organic matter increases or decreases as a balance between input from external sources and immobilization of mineral N, and decomposition/mineralisation of present organic matter. The rates of these microbiological processes depend strongly on soil and meteorological conditions, characteristic of some climatic types, and of agricultural practices. Soil organic matter might also decrease as a consequence of direct land use change, with high rates of soil organic matter mineralisation in the first years and decreasing rates in subsequent years until a new "equilibrium" level of soil organic matter is reached.

Default data or methods to determine the change of N stocks in soils are not available. If no site-specific and peer-reviewed model to estimate soil N changes or site-specific measurement data is available, an initial estimate can be obtained with a soil-balance method (Uwizeye *et al.*, 2016). However, this method provides uncertain results, as it is based on several terms which are highly uncertain (such as N_2 emissions). Özbek and Leip (2015) and Özbek *et al.* (2016) propose a methodology to estimate soil nutrient stock changes from available data where the assumption of zero soil nutrient stock changes (Leip *et al.*, 2014b; Velthof *et al.*, 2009) seemed to be plausible. As a criterion, the authors used a minimum and maximums value of NUE.

While the method described above requires a large quantity of data, a method for a "data-poor situation" was proposed by Hutton *et al.* (2017), comparing fertilized and unfertilized plots, where nutrients are drawn from the mineralisation of soil organic matter stocks, often as a consequence of land use change. Based on observed differences in yield in conjunction with fertilization rates, a minimum level of soil mining occurring for different crops could be derived.

4.2.4.3 Soil phosphorus stock changes

The stock of P in the soils of the feed production system includes soluble P, P contained in living microbes and organisms, dead organic matter, and sorbed P, i.e. inorganic forms of P bound to surfaces, precipitated, or complexed with other materials. Soluble P concentrations in soils are typically low (< 0.01 to 1 mg L^{-1} in fertile soils) (Jones and Oburger, 2011) largely, due to inorganic P sorption and precipitation processes buffering the soil solution concentration, and the contribution of organic P transformations (Dodd and Sharpely, 2015). Microbial P constitutes between 0.5 percent and 26 percent of total soil P (Oberson *et al.*, 2005), while total organic P forms represent 30 percent to 80 percent of total P (Jones and Oburger, 2011). Given the low concentrations and total masses of soluble inorganic P in soils, it is evident that this mass of P is rapidly replenished by soil biogeochemical processes. Indeed, it is suggested that the replenishment of total soluble P required to meet plant growth requirements is likely to be around 10 to 20 times the magnitude of the soluble fraction each day (Rengel, 2012). Figure 4 gives a conceptual view of the forms of inorganic P.

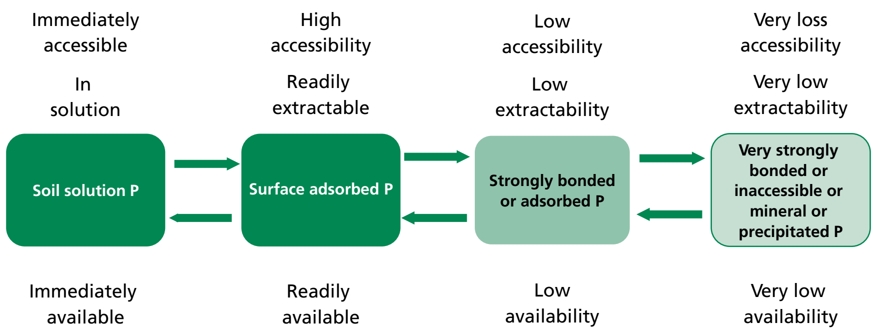
While it is arguable that all sorbed and precipitated P forms can theoretically become agronomically available again (Barrow, 1986), observations that the residual value of previously applied P declines with time after application (Bolland and Gilkes, 1998) suggest that sorption processes dominate and net sorption rates (sorption minus desorption) are generally positive. The portion of P_{sorbed} which is not readily accessed is called $P_{recalcitrant}$, and is represented by the right-most pool in Figure 4. P stock changes can therefore be estimated based on a P soil balance or by estimating the fraction of P input that undergoes strong sorption P_{sorbed} (Equation 4):

Equation 4

$$\Delta P_{stock} = \Delta Pavailable_{sorbed} + \Delta P_{recalcitrant} + \Delta P_{solution} = P_{inputs} - P_{erosion\ loss} - P_{leaching\ loss} - P_{uptake}$$

Figure 4

Conceptual diagram of the forms of inorganic P in soil categorised in terms of accessibility, extractability and plant availability



Note: The relatively unavailable right-hand pool is represented in further discussions as $P_{recalcitrant}$, while P_{sorbed} is represented by all three pools on the right.
Source: Syers *et al.*, 2008.

No default methodology is available to quantify $P_{recalcitrant}$. Models of P sorption based on Langmuir or Freundlich kinetics (McGechan and Lewis, 2002) are prominent in the literature. However P_{sorbed} and these models do not directly predict $P_{recalcitrant}$ (Figure 4).

In dominantly sandy soils (> 90 percent sand), no effective long-term pool of $P_{recalcitrant}$ exists. In other soils, an upper limit of this "internal loss" of strongly sorbed P, $P_{recalcitrant}$ [kg ha^{-1}], can be estimated using Equation 5 based on a conservative estimate of P sorption at the soil solution's eutrophic trigger concentration and the soil bulk density (BD [kg m^{-3}]). P from manure or fertilizers not used (taken up by plants) and not transported off site (leaching, overland flow etc.) after three seasons should be assumed to move into this internal loss pool $P_{recalcitrant}$ (Redding *et al.*, 2016, 2006), until the point when this pool is full. Subsequent additions will then remain not only available for plant uptake but also vulnerable to external losses (leaching, overland flow). Based on a value of 50 mg kg^{-1} for P_{sorbed} at the eutrophic trigger level, the limit of the recalcitrant P storage capacity for $P_{recalcitrant}$ is conservatively (i.e. tending to overestimate this pool) assumed to be (kg ha^{-1}):

Equation 5

$$P_{recalcitrant} < 50 \cdot BD \cdot \frac{(Depth \cdot 10000)}{1000^2}$$

where BD is the bulk density of the soil (kg m^{-3}) and Depth is the storage depth (m) assumed to be 1 m (or where the soil depth is less than 1 m, use the soil depth). The addition of manure-based P sources has been observed to extend the agronomic availability of the nutrient relative to that of an inorganic application (Redding *et al.*, 2016). When better soil data is available, less conservative calculation approaches can be followed (Appendix 4).

4.2.5 Attributing emissions and resource use to single production units

4.2.5.1 Allocation between multiple crops in crop sequences

N and P inputs from organic biomass sources, including residues and green manures, can contribute to the production of several crops grown in sequence. A biophysical allocation approach shall be used according to the number of crops over which their benefits/effects can be attributed. Ideally, this accounts for the temporal pattern of nutrient availability and the relative uptake by the different crops over time. However, where emissions can be specifically related to a single crop (e.g. NO_x from fuel use for specific crop activities), they shall be fully attributed to that crop (Goglio *et al.*, 2017).

If a different crop is grown in a field after another field crop, the calculation of the emissions using the methodologies given in section 4.2.3.2 through 4.2.3.6 need to be allocated between the two crops. This concerns in particular emissions from crop residues and organic fertilizers including manure. Some of the emissions clearly related to cultivation specific crop occur as a consequence of and therefore after harvest of the crop (e.g. nitrate leaching). Emissions shall be allocated in proportion to the share of nutrient remaining in the soil at a defined *cut-off date*. The cut-off date is defined as the start of land preparation for a crop. Thus, the temporal boundaries for the allocation of emissions to a feed crop are from land preparation for the feed crop to land preparation for the following crop.

Remaining nutrients at the cut-off date are considered to add to soil nutrient stocks (see sections 4.2.4.2 and 4.2.4.3). It is important to consider this historic addition of organic biomass when calculating the quantity of nutrient mineralisation for the models used for instance to estimate losses of N_2O and N leaching.

This approach can be applied independently of whether the crop sequence includes or excludes leguminous crops whose nutrients are considered a co-product for the following crop.

Cultivation of legumes as catch crops have the sole purpose of delivering nutrients to the next crop and are considered as part of the "preparation" for that crop. Hence, emissions occurring in the period between land preparation for the catch crop, defined as a fast-growing crop, which is grown between successive plantings of a main crop, and land preparation for the next crop, are allocated to the next crop.

If a catch crop is grown with the purpose of avoiding emissions from the previous crop or if the soil is left bare during a part of the year, emissions occurring until the preparation of the land for the subsequent crop are allocated to the previous crop.

4.2.5.1 Emissions from direct land use change

Land use change, such as the clearing of forests to establish cropland or pasture land, leads to nutrient release following the mineralisation of soil organic matter. These nutrients contribute to input flows similar to releases of residual nutrients from previous applications of fertilizers or crop residues and are discussed in section 4.2.4.

Emissions of nutrients that are caused by land use change and occur before land preparation for the first crop or grassland should be allocated to the crops grown until a new equilibrium is reached (using a default period of 20 years), allocating 1/20 of the emissions to the crops grown each year. The detailed approach to estimate emissions from land use change is provided in the LEAP guidelines for feed supply chains.

4.2.6 Field-to-Gate assessment

The field-to-gate concept estimates harvest and storage losses before the feed is sent to the livestock production unit. These losses could mainly be related to the handling of feed crops at harvest by the feed production unit before it is handed over to the livestock production unit, which defines the 'field-to-gate' and the 'gate-to-mouth' compartments, respectively. In field-to-gate, when there are delays in transporting of feedstocks, losses may occur because of factors such as moisture, temperature, insect and fungal damage, disease, harvesting methods, threshing methods, drying methods, storage conditions, bird and animal damage to the feed crop, and transportation (Appendix 5).

These factors can make the use of the product unsuitable as animal feed. In some cases, food losses may be recycled in the field (residual flows), but in other cases, they are to be considered waste flows.

These flows shall be quantified by using an estimate of total biomass flows in kg DM and the N and P contents in kg N and P $(\text{kg DM})^{-1}$, respectively. Nutrient content shall be obtained by using primary data or if unavailable it may be derived from relevant secondary data.

Feed processing can also occur in the feed-to-gate stage, and associated emissions shall be accounted for (section 4.5 covers background emissions associated with feed processing).

In gate-to-mouth, there can also be feed losses associated with intermediate storage after transfer to animals (Appendix 5) and from feed wastage due to uneaten supplementary feeds. This latter aspect is covered in section 4.3.3.3.

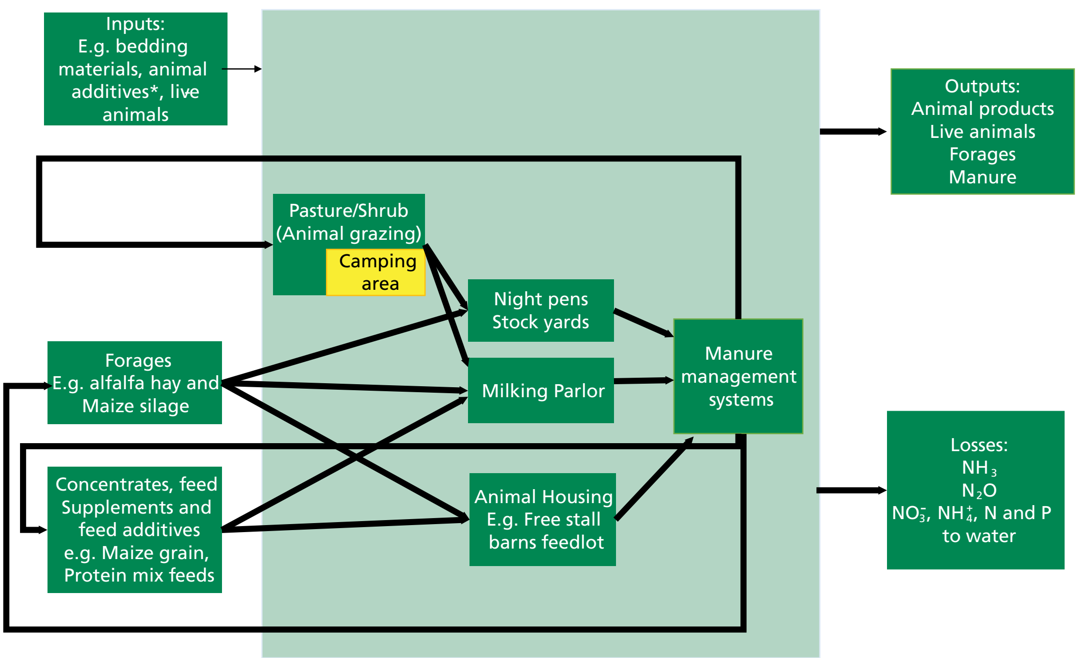
4.3 ANIMAL HUSBANDRY: CONFINED OR HOUSED, GRAZING AND MIXED ANIMAL SYSTEMS

4.3.1 Introduction

This section provides recommendations to estimate nutrient input and output flows in confined or housed livestock systems, grazing systems and mixed housing and grazing systems (Figure 5). The boundary for these systems was drawn to include feed storage and processing on the farm (avoiding double-counting, which is covered in section 4.2.6), animal housing (or confinement lots), manure processing and storage. Depending on the individual farm or region being analysed, some of these sub-systems may not be included. All related feed production components have been covered in section 4.2.

Estimates of nutrient flows of the animal production systems account for all breeding animals associated with the production of the animal output products. In practice, the final production of finished animals for meat processing may involve more than one farm or production system (e.g. separate breeding and finishing systems) and the analysis shall cover N and P flows associated with all components of breeding and finishing systems. In addition, many farms have a mixture of animal species (e.g. sheep, cattle, water buffalo, poultry or pigs), which are often farmed

Figure 5
Generalised diagram showing the components covered in confined, grazed and mixed livestock systems



* animal additives refer to additives directly supplemented to animals e.g. urea blocks for cattle grazing poor quality grass.

together. In these cases, it is recommended to separate activities of the farm system for different animal species where specific uses can be defined, to account for the entire amount of inputs and outputs of the system.

During the transition from the soil-plant system to the animal, the major biogeochemical change is the uncoupling of C from P and N, leading to dung rich in C and P, urine rich in N and K, or a mixture in the form of manures (slurry, solid manure, compost, etc.). In all these animal excreta, C, N, P, and K are more or less labile (from organic to mineral forms) and have the potential to contribute to nutrient losses, recycling or storage in plant or soil compartments.

The wide variability in livestock production systems that exist for all types of animals have been described in the LEAP livestock supply chain guidelines (FAO 2016b, c, d). These livestock production systems cover a range of agro-ecological zones, production intensities, and animal management systems. Animals may be fully housed with brought-in feeds, confined on farms where they graze or browse on feed resources ranging from grassland to mixed grassland/crop/silvo-pastoral systems, or involved in nomadic or transhumance systems with regular movement of animals depending on the feed resources. Analyses need to account for nutrient flows associated with all feed and animal transfers that contribute to production of the animal products.

Most animal production systems have animal collection areas, ranging from little use (e.g. to treat animals for intestinal parasites or for collection before sending off for processing) to regular use (e.g. night corrals or milking parlours) or to continuously confined use (e.g. in housed or feedlot systems). Manure management is an important determinant of nutrient flows associated with the animal collection areas, and all related nutrient flows and losses shall be accounted for. Figure 5 shows some of the main components of confined, grazed and mixed livestock systems.

For grazing systems, excreted nutrients are largely redistributed in the landscape by direct deposition by animals. Excreta deposition is, therefore, often uneven, with high nutrient loads in some areas which may pose a high risk of nutrient loss and environmental contamination, depending on how intense the livestock header is.

Most nutrient flows depend on animal population densities. Accurate estimates of animal populations are essential to accurately estimate nutrient inputs and outputs. Many circumstances contribute to an average animal population that varies from an animal feeding operations maximum capacity or lead to animal housing being occupied less than 365 days per year.

4.3.2 Quantification of input flows

A first step in estimating nutrient flows into the livestock stage is to estimate the nutrient input in the different types of feed, imported bedding materials, additives provided directly to animals, and live animals entering the farm.

4.3.2.1 Quantification of animal N and P intake and bedding materials

Previous LEAP guidelines for the livestock supply chain have covered the methodology for calculating animal dietary intake and some aspects of excretion. When the amount and types of different feeds consumed are not measured, the use of energy-based feed intake models are recommended to determine energy requirements, and this is then linked with data on energy and nutrient composition of the feeds to calculate N and P intake in feeds. This shall be based on primary data to account for the animal population (herd and flock size), productivity and timing through the year (FAO, 2016b). Similarly, primary data on the composition of feeds (including N and P concentrations) shall be used based on farm-specific or regionally-valid feed composition data. When this data is not available, national databases should be preferred over continental/global feed composition data. N and P contents of individual feed ingredients can be derived from feed databases such as FAO's Feedipedia[7] and the National Animal Nutrition Program for the the United States of America[8]. When additives containing N and/or P are mixed with feeds during compound feed production or at the time of feeding to animals, this extra input shall be accounted for based on primary data on the quantity and nutrient concentrations of the compound feeds or of the direct additives.

In grazing systems, there is substantial variation in nutrient concentrations in forage-based diets. For each feed source utilized by grazing animals, there is a need to have an accurate average estimate of the chemical composition (concentrations of dry matter, metabolizable energy, digestibility, N and P content) based on either a weighted annual average or on a monthly basis accounting for feed quality differences and changes in profile of energy demand throughout the year. When possible, primary forage composition data should be obtained at least on a seasonal basis. However, in grass-based systems, most feeds are not routinely analysed for nutrient concentration prior to consumption. When primary data is unavailable, the most accurate secondary data available for the specific regional system should be used (i.e. data from existing feed databases or published statistics of relevance to the study system, location, and feed type). If data on feed types consumed and nutrient concentrations have very high uncertainty, an option for estimating N and P intake is also to do a sensitivity analysis based on the use of an animal protein or P requirements model (NRC, 2000). Note, however, that the latter would provide data on the minimum N and P requirements and therefore is likely to underestimate actual N and P intake (and consequently also underestimate N and P excretion calculations based on that data).

Nutrient imports in the bedding material depend on the amounts used per livestock unit, the number of animal units on the farm, and also on the type and quality of the bedding material. As many bedding materials can serve as (low-energy) feed, their nutrient composition is frequently included in feed databases (e.g. FAO Feedipedia; NRC 2001).

In extensive grazing systems, N and/or P may be provided directly to animals, for example, via direct dosing, within salt blocks, in water systems or in trays in the field. Where this occurs, primary data on the amount of supplement and its N and P concentration shall be determined.

4.3.2.2 Animal inputs

Animal inputs from outside of the system under study (e.g. live animals from other farms, such as weaned animals to finishing farms) should be estimated. Procedures for estimating animal nutrient outflows are discussed in Section 4.3.3. These same procedures can be used to represent inputs as replacement animals and grazing animals.

[7] http://www.feedipedia.org/

[8] https://nanp-nrsp-9.org/

4.3.3 Quantification of output flows

In grazing systems, the main N and P output flows are in animal products or as live or dead animals, and the various losses are from excreta deposited directly on the grazed area and from the manure management system from the animal collection area (e.g. from uncovered yards and housing). In housed livestock systems, outputs of N and P in manure to crop or pasture land (section 4.3.3.3) or other endpoints (e.g. sold as a fertilizer or soil amendment or to waste) represents the difference between the various inputs to the manure system (excreta, wasted feeds, bedding) and losses from collection and storage.

4.3.3.1 Mass of N and P in live weight

The mass of N or P in the animal body entering or exiting the system is estimated from data on animal numbers, live weight (LW), nutrient concentration and live weight correction (LWC) factor for gut-fill (equations 6 and 7). Estimates of nutrient concentration (NC_{EBW}) and LWC factor are given in Appendix 6. Estimates of live weight and number of animals entering the system shall be determined for the studied system (e.g. an individual farm or for the region or country based on available production statistics).

Most nutrient concentrations are reported on an empty body wet weight (EBW) basis. If LW is commonly available, an LWC factor from LW to EBW will need to be applied. It is important to also apply an animal body nutrient concentration value (NC_{EBW}) representative of both the animal species and its weight. Nutrient concentrations in animals typically change with body mass.

For dead animals transferred to off-farm uses (e.g. rendering), one may choose to use the average of weight in and weight out as LW. This assumes that mortality occurs at a constant rate over time. In reality, more deaths typically occur among the youngest animals shortly after arrival at the farm.

If weight is reported as LW, the mass of N and P represented by animals is calculated according to Equation 6; if weight is reported as EBW, Equation 7 shall be used.

Equation 6

$$NUTR_{BM} = NC_{EBW} \cdot LW \cdot LWC \cdot A$$

Equation 7

$$NUTR_{BM} = NC_{EBW} \cdot EBW \cdot A$$

$NUTR_{BM}$	Mass of N or P represented by the animal body mass (kg unit of time^{-1})
NC_{EBW}	Nutrient concentration (kg of nutrient kg EBW or percent^{-1})
LW	Live weight of animal (kg)
EBW	Empty Body Weight of animal (kg)
LWC	Live weight correction factor or ratio of EBW to LW. The difference between LW and EBW is the weight of gut contents.
A	Number of animals entering (nutrient input) or exiting (nutrient output) the system per unit of time.

4.3.3.2 Mass of N and P in animal products

The mass of N or P represented by animal products (milk, eggs, wool) is estimated based on the mass and nutrient concentration of the products (Equation 8). Estimates of the nutrient concentration of products are in Appendix 6.

Equation 8

$$NUTR_{AP} = NC_{AP} \cdot AP$$

$NUTR_{AP}$	Mass of N or P in animal products such as milk or eggs (kg of nutrient unit of time^{-1})
NC_{AP}	Nutrient concentration in animal product (kg of nutrient kg of animal product^{-1} – e.g. milk or eggs)
AP	Mass of the animal products produced (kg of animal product unit of time^{-1})

Values for N and P concentrations of animal body mass and animal products should be based on primary data. When unavailable, secondary data shall be obtained from relevant databases. This should be representative of animal factors including animal type, weight, productivity, and breed.

4.3.3.3 N and P in excreta and manure

A Tier 2 method is recommended to estimate the amount of N and P excreted by animals, which is based on the difference between estimates of N and P intake in feeds (as outlined in section 4.3.2.1) and of N and P incorporated into animal tissues and products (as outlined in sections 4.3.3.1 and 4.3.3.2) (ASAE, 2014; IPCC, 2006), which may vary between sex, age and production stage.

In grazing systems, urine and dung depositions often occur spatially disconnected, and the relative amounts of N and P excreted in urine compared to that in dung influences N and P flows. Generally, 50 and 90 percent of the N and P consumed by pigs and ruminants is excreted. As the concentration of N in an animal's diet is increased, the amount of N excreted in urine increases sharply, while the amount of N in the dung remains relatively constant (Peyraud *et al.*, 1995). In contrast, most P is in dung, and urinary P excretion can be considered negligible, at least for ruminants (Alvarez-Fuentes *et al.*, 2016). A summary of research using an analysis of published data for dairy cattle, beef cattle and sheep resulted in the following equation (Luo and Kelliher, 2010; $r^2 = 0.67$, $P < 0.01$):

Equation 9

$$f_{N,urine} = 10.5 \cdot N_{diet} + 34.4$$

Where $f_{N,urine}$ is the percentage of total excreted N in urine (percent) and N_{diet} the N content in the diet (percent). The difference from 100 is the percentage of N excreted in dung.

For ruminants, it can be assumed that 100 percent of the P excreted is in dung.

In confinement or housed livestock systems, the dung and urine are generally deposited together onto surfaces that may range from the bare soil through to fully sealed systems (e.g. concreted). All or a proportion of this excreta is collected in a manure storage system. Thus, recognizing differences between excreted and collected manure in housing systems is important when defining manure flow.

Additionally, inputs into the manure system can include wasted uneaten feed during and following feeding and shall be accounted for. Pigs and poultry in many systems are fed within the animal housing, and any wasted feed is immediately incorporated into the manure or litter. Wasted feed from some dairy and beef systems may be separate from the animal housing and not added to the manure. In most cases, the wasted feed does not leave the farm, or it may be transferred to the cropping system.

The collected manure may be managed as a slurry or as solid. Slurry consists of excreta, some bedding material, spilt animal feed and drinking water, and water added during cleaning or to assist in handling. Solid manure consists of excreta, spilt animal feed, and drinking water, and it may also include bedding material. These forms are equivalent to the liquid/slurry or solid manure category in IPCC (2006).

The manure management systems (MMS) of the supply chain should be obtained from primary data. If these are not available, the distribution of manure over the various MMS present in a country (including the share of manure excreted by grazing animals) is available from the CRF Table 3B(b) of the national GHG inventory[9]. The national GHG inventory reports should also contain information on any other use of manure and/or import or export.

4.3.3.4 Gaseous N flows and sources of emissions from manure

During grazing and manure management, emissions of NH_3, N_2O, nitric oxide (NO), and molecular N (N_2) can occur. The amounts of the losses depend on the type of MMS.

Guidance for the manure pool and grazing emissions builds entirely on existing guidelines relevant for emissions and N flows in grazing and manure management and storage systems:

- IPCC guidelines (IPCC, 2006), Volume 4 (Agriculture, Forestry, and Other Land Uses, AFOLU). For confined system manure management, Chapter 10 (Emissions from livestock and manure management). Section 10.5 (N_2O emissions from manure management, pages 52-70) explains the methodology for calculating direct and indirect N_2O emissions from manure management as well as the coordination with emissions from manure applied to soils. The IPCC guidelines also give default factors of total N losses from manure management including losses of N_2. For grazing systems – Chapter 11 (N_2O emissions from managed soils, and CO_2 emissions from lime and urea application). Section 11.2 (N_2O emissions from managed soils, pages 5-27) outlines a methodology for calculating direct and indirect N_2O emissions from urine and dung directly deposited on soils. Where possible, emission factors should be derived from country-specific data, and consideration should be made of recent peer-reviewed studies.
- EMEP/EEA air pollutant emission inventory guidebook 2016 (EEA, 2016), Tier 2 techniques for NH_3 and NO emissions when detailed information on manure management and composition from confined systems is available.

[9] See examples of CRF Tables submitted to the UNFCCC here http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/9492.php

Ammonia volatilisation losses

The estimation of ammonia volatilisation losses should be based on emission factors (EFs). Country-specific EFs should be prioritised. For example, for UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP; Gothenburg Protocol) members, a Framework Code for "good agricultural practices for reducing ammonia emissions" provided EFs for several countries. When country-specific data is not available, ammonia emissions can be estimated using IPCC (2006) (Tier 1) equations and EFs. However, consideration should be made of the body of recent relevant peer-reviewed studies. Another alternative is the use of ammonia emission models such as Bouwman *et al.* (1997) or Beusen *et al.* (2008).

Housed and confined livestock

A Tier 2 methodology consists of applying a specific ammonia EF required for each MMS, and any manure treatment applied. EFs should preferentially be based on country-specific data (potentially derived from the Informative Inventory Report [IIR] or National Inventory Report [NIR]) more recent published and validated data (e.g. with regard to beef feedlot pen surfaces; Denmead *et al.* (2008); Flesch *et al.* (2007); Loh *et al.* (2008); McGinn *et al.* (2007); McGinn *et al.* (2016). In the absence of relevant country-specific material, EFs from Table 3.7 in the EEA guidebook may be applied. The effect of some abatement measures can be adequately described using a reduction factor, i.e. proportional reduction in emission compared with the unabated situation. Abatement may be achieved by manure treatment and by covering of manure stores. For each MMS, an integrated EF may be calculated with implementation factors of the applied emission reduction measure.

Grazing livestock systems

For a Tier 2 methodology, country-specific EFs, based on representative measurements made in that country or region shall be used when they exist. This could include separate EFs for urine and dung N because the proportion of losses are generally higher from urine than from dung (e.g. Kelliher *et al.*, 2014).

N_2O emission

Direct N_2O emissions from animal excreta and manure shall be estimated (see also section 4.2.3.4). The latter depends on the fraction of manure that is managed in each type of MMS. For each MMS, an N_2O EF is needed. If no country-specific data is available in the IIR or NIR, EFs from Table 10.21 of the IPCC 2006 guidebook can be used. However, consideration should be given to the body of relevant peer reviewed studies subsequently available (post 2006).

4.3.3.5 N and P runoff and leaching from confined manure management

P flows from manure management are restricted to dissolved and particulate forms in outdoor systems, largely via water transport (Larney *et al.*, 2014; Vadas and Powell, 2013). While water transport of N is also likely (Larney *et al.*, 2014), the magnitude of this pathway in adequately managed systems may be small relative to gaseous emissions. Management approaches can be applied to minimize water-borne losses (Skerman, 2000; Tucker *et al.*, 2004). It is also conceivable that wind-blown dust from manure management areas may contain P and N (Miller and Berry, 2005), though the magnitude of this export is not known, but likely to be small relative to other pathways.

However, appropriate management approaches are available to reduce these flows for example construction techniques that prevent leaching by compacting underlying soil and bonding the storage area to collect all runoff. When such management approaches are in place to limit these flows, the flows should be accounted as zero.

Pond over-topping losses are probably more difficult to manage, but in locations with good meteorological data and given appropriate production data, design approaches that decrease this risk to a negligible level are also available (1 in 10-year over-topping frequency; Skerman 2000). These same design criteria could be applied to estimate direct N and P over-topping losses from pond systems, or direct data should be used where available.

Limited research exists on P and N runoff from solid manure stored outdoors (e.g. in windrows) and therefore a Tier 2 method based on Larney *et al.* (2014) is proposed in Appendix 8, requiring data on manure storage area, mean annual duration of precipitation events generating runoff, and water-soluble P concentration of manure. It is recommended that this Tier 2 method is used when no primary data is available. P loss in runoff from cattle yards and feedlots can be estimated using Vadas *et al.* (2015).

4.3.3.6 N and P runoff and leaching from grazing systems

N: Grazing systems result in a concentration of N in discrete urine and dung patches at very high N rates and can lead to significant N leaching (particularly from urine; Ledgard *et al.*, 2009). Excreted N partitioned into urine and dung (Equation 9) can be used to estimate N leaching using Tier 2 country-specific EFs where available. Section 4.2.3.6 describes the basic Tier 2 method using a single FracLEACH value for the different N input sources. However, various countries have specific Tier 2 or 3 models that account for urine and dung N and can include greater site differentiation based on soil and climate properties and temporal differences throughout the year. The use of such models should be based on them having been validated, published and accepted as recognized country-specific models.

P: Dung is the dominant source of excreted P in grazing systems, and it can be the main source of P runoff from grazed pasture systems other than P loss from erosion (e.g. Vadas *et al.,* 2014). The specific annual dissolved P loss in runoff from dung in grazed pastures can be calculated based on Vadas *et al.* (2014) using the Equation 10:

Equation 10

$$\text{Dung P runoff} = (\text{dung WEP}) \cdot (\text{annual runoff / precipitation}) \cdot (\text{P distribution factor}) \cdot (\text{dung cover reduction factor})$$

Where dung WEP is dung water extractable P and the P distribution factor = (annual runoff/precipitation)$^{0.225}$. As dung does not cover the entire soil surface, the estimation of dung P loss for cattle is corrected by an annual dung cover reduction factor:

Equation 11

$$\text{Dung cover reduction factor} = 1.2 \cdot (250 \cdot \text{annual cover}) / ((250 \cdot \text{annual cover}) + 73.1))$$

Where annual cover is expressed as a decimal with a maximum of 1.0 and is calculated assuming each 250 g (dry weight basis) of excreted cattle dung covers 659 cm^2.

However, it is necessary to account for all sources of P runoff, and the commonly used approach across a range of countries is the use of a P Index system (section 4.2.3.6, Appendix 8).

A P index system for grazed pasture systems was adopted in 47 the United States of America (Sharpley *et al.*, 2003) and in the United Kingdom, Finland and New Zealand (Heathwaite *et al.*, 2003 and McDowell *et al.*, 2007). This index represents site vulnerability to P loss and is determined by multiplying source and transport factors (Sharpley *et al.*, 2003). An important characteristic of P loss from grazed pastoral soils is the spatial variability and link between the site of P sources for loss and the site-specific vulnerability. Thus, when available, a country-specific validated P Index system that is locally-calibrated is recommended because it can potentially account for spatial variability within a landscape in source/site factors and in the pattern of animal grazing and any heterogeneous deposition of excreta (Appendix 8).

4.3.4 Allocation of emissions to manure

Manure represents a valuable source of nutrients that can have multiple uses:

a) Manure can be used for its fertilizer value and be applied or deposited to land (crops and grassland). In this case, manure is used instead of or to partially replace mineral fertilizers which would possibly need to be purchased. Benefits from manure are its content of nutrients (including N, P), but manure also returns organic matter to the land and might lead to positive structural characteristics of the soil.
b) Manure can be used for its energy value and (upon drying) used as a feedstock or small-scale stoves. In these cases, manure replaces other fuels (gas, coal, etc).
c) Manure can be sold on the market for further processing and/or application to land. Manure can be treated in biogas installations producing both energy and a digestate with fertilizer value.

In all three cases, manure generates a benefit for the farmer, even though only in case (c) does it generate a direct revenue to the farmer. In cases (a) and (b) the value for the farmer and his/her household is the saved expenses for fertilizers and/or fuel. However, even if the manure is sold, in some cases it is difficult to relate the revenue of the manure to its value, as policies in many countries limit the application of manure to land, and thus the fertilizer price reflects also costs avoided for alternative 'waste' treatment options. In some other cases, manure can be regarded as an important or even the most important co-product of a livestock production system, with the aim to transfer nutrients from grassland systems to (cash) crops (Rufino *et al.*, 2007, 2006; Weiler *et al.*, 2014).

Therefore, manure shall be considered as a co-product, with some exceptions. These exceptions include landfilling or "dumping", including discharge to water courses, application in excess of crop needs, and incineration without energy recovery. This also holds for other organic fertilizers applied to crops. "Excess of crop needs" can be assessed with crop-response curves if available or crop nutrient requirements and are quantified based on mineral fertilizer equivalents of the applied nutrients. Excess application of nutrients occurs when a crop receives more nutrients than the physiological optimum for potential yield, beyond which no further yield increase is achieved. When the land receives nutrient inputs from various sources, the order of nutrient sources for determining which is in excess of crop needs should be as follows: nutrients mineralised from soil stocks (and crop residues and residual mineral and organic fertilizers applied in previous growing seasons) > nutrients from biological fixation and atmospheric deposition > nutrients from recycling of organic material (manure and other organic fertilizers) > nutrients from mineral fertilizers. Thus, if the total input of nutrients exceeds the physiological optimum, mineral fertilizers applied are considered as "wasteful" application first, before any other nutrient source (such as manure) is to be considered as waste. There are two possible options to allocate upstream emissions of livestock production systems between manure that leaves the production system and animal co-products:

- Method 1: Bio-physical allocation using the heat energy, as explained in Appendix 3 of the FAO LEAP poultry guidelines;
- Method 2: Economic allocation based on the fertilizer value. Details of a possible implementation of such an approach are provided in Appendix 7. The method consists of quantifying the fertilizer value of the manure based on crop-nutrient response curves, relative nutrient efficiencies, and mineral fertilizer nutrient prices.

Method 1 is much easier to apply as it does not require additional data, gives an allocation factor as a function of feed intake, independent of the animal type, and links with the fraction of metabolizable energy intake that is required for digestion. In contrast, method 2 requires more data, in particular also on the system the manure is applied to (which could be outside the system boundaries of livestock supply chain under consideration). On the other hand, it gives an allocation factor as a function of the benefits that are derived from the use of manure.

In most cases, method 1 (biophysical) will be preferable due to its robustness and simplicity. However, it is recommended that when sufficient data is available, method 2 (economic) is evaluated.

4.4 ANIMAL PROCESSING

Different animal parts re-enter the production system through different pathways, such as organic fertilizers or animal feed. A key challenge is therefore to identify these N and P flows and the downstream processing technologies that recover part of these nutrient flows. Particularly for P, the by-products, for example, bones, contribute a significant share of the flow for which the statistical data sources of end use are lacking. This section gives an overview of the different possible flows and recovery options and the emissions generated when they are not recovered. The amount and the type of recovery differ greatly depending on the supply chain and the legal requirements imposed on it.

Quantifying flows in a Tier 1 approach can be based on the mass balance method. Tier 2 requires gathering primary data on the partitioning of animals into products and their respective nutrient contents and the subsequent processing steps applied to the generated products and waste. Principles of allocation of emissions between co-products, residuals, and wastes were described in the LEAP Livestock Guideline documents (FAO, 2016b, 2016c, 2016d). Recycling of nutrients from residues and waste from animal processing or later life cycle stages (e.g. in sewage from consumed products) onto land, such as for crop production, will be accounted for when an LCA covers cradle-to-grave stages. For a cradle-to-primary-processing LCA, these nutrients will be accounted for as inputs, as described in sections 4.2.2.5 and 4.2.2.7.

4.4.1 Nitrogen and phosphorus output in products

Appendix 6 provides detailed information on the typical N and P concentrations in a range of animal products.

4.4.2 Residues, waste and wastewater treatment

4.4.2.1 Nitrogen and phosphorus in residues and waste

The residues (occasionally these might be co-products) and solid waste at the animal processing level include hooves, feathers, hair, skin, bones, skull, brains, intestinal contents, and animals that died before slaughtering or for disease prevention. All these sources of solid waste or residues are rich in N and P, and their treatment and fate should be considered in assessing the nutrient flow of a livestock production chain. The relative share of the different type of residues compared with a main product depends on the type or even the breed of the animal. Therefore, if detailed data is missing, the simplest approach to quantify the N and P losses is to compare the LW of the animal and the total mass of the end products sold, while assuming that the relative share of N and P will be similar. However, when there are inedible co-products used for other purposes, then primary data or published secondary data on their N and P concentrations should be obtained, because they can be highly variable, for example tallow used for various purposes including biofuel can be considered as having no N or P. Dairy processing facilities are not considered to produce solid waste originating from livestock production.

4.4.2.2 Treatments and fate of residues and waste

The fate of the nutrients, emissions, and losses during processing of animal products depend on the degree of recycling and the processing options of residues and waste.

Animal fat and sometimes protein fractions that are not used in feed or pet food may be treated using anaerobic digestion to produce biogas. The nutrient losses during this treatment are very low. All P remains in the digestate, and small (less than 5 percent) ammonia volatilization losses can occur. The efficiency of the nutrient fraction that goes to the anaerobic digestion depends on the further treatment or application of the digestate. Digestate can be directly applied to land or undergo separation into solid and liquid parts. A relatively higher share of the P ends up in the solid fraction and a higher share of the N in the liquid fraction. The solid fraction can be incinerated, composted or again applied to land. The liquid fraction can be applied to land as a fertilizer or treated in a wastewater treatment (see following section).

Composting is another treatment option more often applied in developing countries directly on the solid waste or residues and sometimes on the solid fraction of digestate. All P can be accounted for as fertilizer if appropriately applied to land. Volatile N losses that occur during composting can only be prevented in controlled composting units using air scrubbers.

Biochar production is mainly applied to animal bones, which consist of 65–70 percent inorganic substances, mainly calcium hydroxyapatite ($Ca_{10}(PO_4)_6(OH)_2$). Bone char is a P fertilizer and soil improver and is produced by high temperature pyrolysis to more than 500 degrees Celsius in the absence of oxygen. The N present in tissues attached to the bones is volatilized and lost during the process.

4.4.2.3 P and N in wastewater

Wastewater is generated by the processing unit through cleaning of the equipment and facilities. For animal meat processing plants, wastewater contains residues of urine, faeces and blood and can contain both N and P. The biggest obstacle for untreated recovery or reuse is bacterial contamination.

Wastewater is also produced in households and restaurants from the consumption of animal products, and this can be processed in a wastewater treatment plant (WWTP). In some cases, it is collected and applied to land or may enter soil via septic tank systems.

Biological contamination will be overcome mostly by secondary treatment at the WWTP and finalized by tertiary treatment for pathogens.

4.4.2.4 Wastewater treatment and P and N removal efficiency

Wastewater treatment consists of three treatment phases. The primary treatment typically starts with sedimentation and complementary flocculation, in which a part of the waste in the water can be recovered in the solid fraction. The N or P recovered during flocculation can be further treated using anaerobic digestion or composting and later be applied as fertilizer.

Depending on the composition of the wastewater, precipitation chemicals can be used to flocculate P. Another technology for P-rich wastewater is the precipitation of struvite (magnesium ammonium phosphate: $NH_4MgPO_4{\cdot}6H_2O$). However, this method is not as effective in binding P as chemical precipitation. Struvite is a phosphate mineral that can later be used as input for the phosphate industry or be applied directly as fertilizer. The P-removal stage is often combined with N-removal in gaseous form, which means a loss of N. The sludge produced is often dewatered. A large part of the N is dissolved in the liquid fraction, and a great deal is lost with the effluent. Sludge from municipal wastewater treatment is rich in P, especially if chemical precipitation is used. Depending on other contributors to the WWTP, such as industries, different amounts of unwanted substances can be found in the sewage sludge. If these contributors are restricted and the sewage sludge not contaminated (with microrganisms [namely pathogens], and /or heavy metals), it can be used on farmland for irrigation and fertilization purposes, but this is highly regulated in some countries.

The secondary, or biological treatment, will be up to 95 percent of microbial biomass and allow the discharge of most of the treated water into natural receptors and safe use for irrigation. Finally, the tertiary treatment will remove pathogens and recover a significant amount of nutrients such as N and P.

The sludge remaining after the above-mentioned anaerobic digestion can also be incinerated. All N present in the waste is lost, while the P can be recovered in regions where the ashes are allowed to be used as fertilizer. The presence of excess metals generally precludes their use on farmland. Primary data on nutrient output from wastewater processing should be used. When these are not available, the latter should be estimated from secondary data according to the type of wastewater processing system used. A default option for gaseous N emission factors is to use those for manure from section 4.3.3.4 according to the type of storage and treatment system used.

4.4.2.5 Feed and food residues and waste

Feed and food losses occur throughout the entire feed/food supply chain and potentially generate nutrient losses into the environment, besides the social and economic implications. The Food and Agriculture Organization of the United Nations (FAO) estimates that about one-third of food produced worldwide is annually "wasted" (equivalent to $1.3 \cdot 10^9$ tonnes) (FAO, 2011). Nutrients contained in the food not eaten by humans from unsold or unsaleable fresh produce from farms, supermarkets and other sources of matter from urban centres have been used as animal feed, added to bio-digesters or applied to agricultural land. The latter residues frequently enter municipal solid waste streams and are applied to soil after composting. According to Kantor *et al.* (1997), 32 percent and 25 percent of the total grain products and vegetables, respectively, that are supplied by the retailer, food service and consumer end of the supply chain are uneaten by humans. In practice, it can be difficult to obtain accurate estimates of the extent of food losses or wastes for a studied system. When this is the case, it is recommended that a sensitivity analysis be used in LCAs that extends to the retailer/consumer level to illustrate the effects on nutrient flows from food residues or wastes.

4.5 UPSTREAM PROCESSES AND TRANSPORTATION

4.5.1 Fertilizer production

A review of global fertilizer production, energy use, and GHG emissions was given by Kool *et al.* (2002). Limited specific data on N and P emissions during manufacturing of some fertilizers from this review and industry sources are given in Appendix 9. Examples of some N and P emissions from manufacturing of some N and P fertilizers are also given in Appendix 10.

During manufacturing of fertilizers, there may be more than fertilizer products produced. One example is during manufacturing of superphosphate from phosphate rock and elemental sulphur. The elemental sulphur is used to produce sulphuric acid, which is reacted with the phosphate rock. This process is exothermic, and the heat generated can be used to generate electricity that can be fed back into the national grid. Thus, co-products of superphosphate are superphosphate and electricity. Because these co-products have different functions, the method of allocating emissions between co-products would be economic allocation according to the value of the two co-products. However, some electricity is also used in the process of manufacturing superphosphate. In the case of the average superphosphate produced in New Zealand (Ledgard *et al.,* 2011), the electricity use almost exactly matches the electricity generation and, in that case, it can be assumed that there is no net electricity use/generation and that no allocation was required.

4.5.2 Production and use of cleaning chemicals, refrigerants and other consumables

The production and use of any input contributing more than 1 percent to the nutrient cycle impact assessment of the entire supply chain should be accounted for. Such inputs can include, among others:

- Alkaline builders (e.g. sodium hydroxide)
- Acid builders (organic and inorganic acids)
- Water conditioners (e.g. sodium tripolyphosphate)
- Oxidizing Agents (e.g. hypochlorite)

- Refrigerants (e.g. ammonia, R404A, R410A)
- Packaging materials (e.g. glass, HDPE, aluminium)

N and P emissions and depletion due to the production of the above-mentioned compounds can be retrieved from databases (e.g. ecoinvent) or literature studies (e.g. Kapur *et al.*, 2012). Nutrient-related emissions during the production of these products are mostly the reactive N emissions during combustion processes needed for energy and transport during production.

P-related emissions from the use of products are mainly the P inputs in surface waters from P-containing detergents. P from detergents may account for up to 28 percent of P in human wastewater to surface waters in countries where wastewater treatment is poor and P-containing detergents are dominant (Wind, 2007).

N emissions related to the use of inputs consists mainly of ammonia used as a refrigerant.

Because of its high energy efficiency and low cost, ammonia is extensively used in industrial refrigeration applications, warehouses, and regional distribution centres. DEFRA (2008) estimates its annual leakage to be 15 percent.

4.5.3 Generation and use of energy

To calculate the emissions associated with the use of energy in a livestock supply chain, the energy use shall be carefully determined or retrieved from the literature or databases (e.g. ecoinvent) if direct measurements are not available.

For example, at most abattoirs, the refrigeration plant is the major contributor to electricity use. It constitutes 45 - 90 percent of the total requirements during the working day and almost 100 percent during non-generation periods. The cooling energy supplies chillers, freezers and refrigerated storage rooms (EC, 2005). An indication of energy use in abattoirs and dairy processing plants is given in Appendix 12. Primary data on fuel use from transportation should be collected or estimated based on the type of vehicles used and distances covered (see details on transportation calculations in the main animal guidelines, e.g. for large ruminants [FAO, 2016b]).

Once the electricity and fuel use is defined, the N (NO_x and NH_3) emissions associated with their generation and use shall be calculated.

The generation of conventional fuels is associated with the release into the atmosphere of NO_x emissions. Biofuels can also be responsible for the generation of N_2O and NH_3 emissions to the air and of nitrate and phosphate discharge into water (through leaching and runoff). To quantify such emissions, data can be sourced from databases (e.g. ecoinvent) or from literature studies.

The relevant N pollutant originating from fuel combustion is NO_x, while small amounts of NH_3 may be emitted as a result of incomplete combustion of all solid fuels containing N. This occurs when the combustion temperatures are very low (fireplaces, stoves, old-design boilers) (EEA, 2016). Emissions associated with fuel burning depend on the type of fuel used (e.g. petrol, diesel, LPG) and the type of machinery/plant where the combustion of fuel takes place. Such emissions can be sourced from most widespread databases available for LCA studies (e.g. ecoinvent). Alternatively, they can be calculated using the EFs available in the literature, such as the ones provided by the European Environment Agency (Combustion in the manufacturing industry, EEA, 2016).

Electricity generation is a key contributor to global emissions of NO_x and related impacts. Direct emissions from plant operation represent most of the life cycle emissions for fossil fuel technologies, while fuel supply represents the largest contribution to biomass technologies (54 percent) and nuclear power (82 percent); infrastructures are the main contributor for renewable energy sources (Turconi *et al.*, 2013).

The starting point for calculating the emissions of NO_x associated with electricity generation is the definition of the electrical mix in the country where the electricity is produced. The ecoinvent database provides NO_x emissions for several country mixes.

5. Life Cycle Impact Assessment (LCIA)

Life Cycle Impact Assessment (LCIA) evaluates the magnitude and significance of potential environmental impacts of a product system throughout the life cycle of the product or service (ISO 14040:2006). The selection of environmental impacts is a mandatory step of LCIA, and this selection shall be justified and consistent with the goal and scope of the study (ISO 14040:2006). For the environmental impact assessment of nutrient emissions from livestock supply chains in an LCA context, all impact categories that are qualified as relevant and operational, according to the selection and classification steps of the impact assessment phase (ISO 14044:2006), should be covered.

Impacts can be modelled at different levels in the environmental cause-effect chain, which links elementary flows of the LCI (emissions and consumptions) to impact categories. In LCIA, the cause-effect relationship between emissions and impact is quantified using characterization factors (CFs), which have units of impact per emission. Figure 6 provides an overview of some potential impacts arising from the full supply chain of livestock production.

It is essential to distinguish midpoint impacts, which characterize impacts located anywhere between emission and areas of protection, representing the values society aims to protect in the environmental cause-effect chain, and endpoint impacts, which characterize impacts at the end of the environmental cause-effect chain. Impacts may be aggregated to provide indicators at, or close to, the areas of protection. Usually, three areas of protection are recognized: human health, ecosystems quality, and resources. Aggregation at the endpoint level and at the areas of protection level is an optional phase of LCA according to ISO 14044:2006.

Aquatic eutrophication potential is an example of a midpoint impact category. The results of the LCI are the contributing substances covering the total loads of N and P compounds emitted, per functional unit, to aquatic systems. Based on the eutrophic activity and CFs specific to each compound of N and P, eutrophication potential can be used to aggregate all nutrient losses to the same midpoint impact category indicator, for example kilograms of phosphate (PO_4^{3-}) equivalents per functional unit. Extending the cause-effect chain, the contributing substances' impacts are modelled as effects on ecosystems (e.g. a fraction of species affected), which results in an endpoint impact.

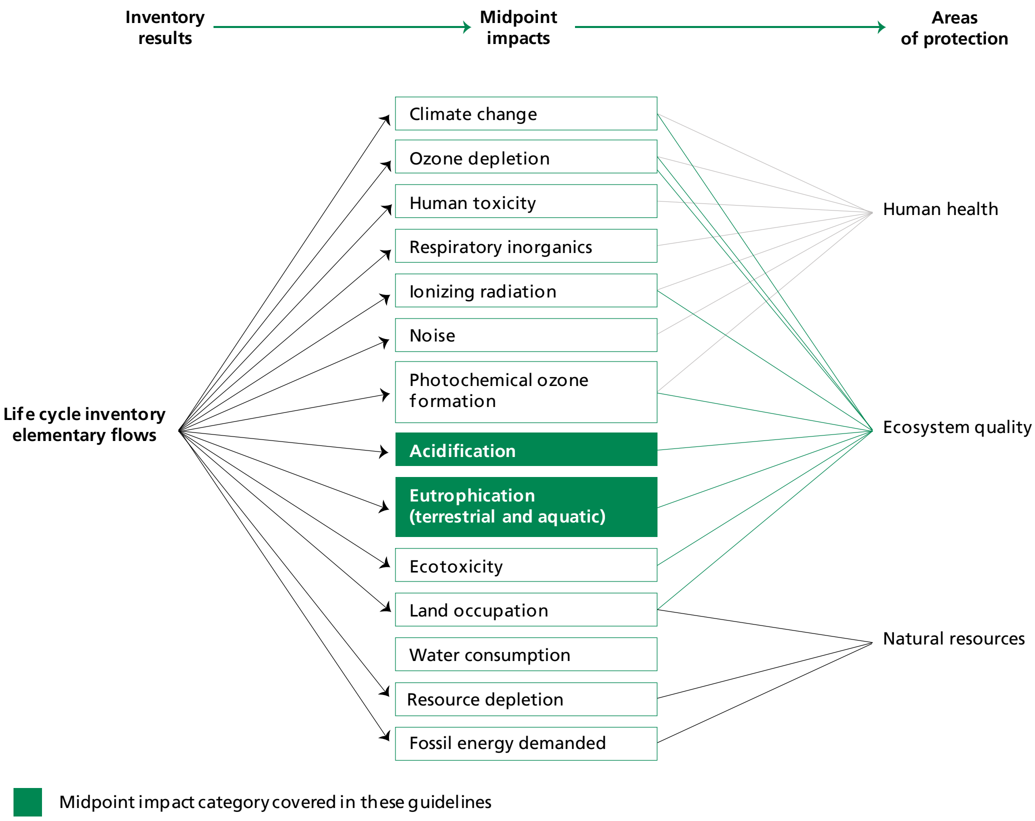
The following sections describe in detail the two impact categories likely affected by nutrient emissions to the environment that are covered in these guidelines: eutrophication and acidification.

5.1 IMPACT CATEGORIES

The following sections describe the processes and substances, related to agriculture, that contribute to acidification and eutrophication. While the nature of the effects of the two impacts is different, acidification and eutrophication share some fate and transport processes in the environmental cause-effect chain, largely because N compounds can contribute to both.

Reactive N compounds may contribute to several LCIA impact categories. Waterborne dissolved inorganic N (DIN) forms include nitrate (NO_3^-), nitrite (NO_2),

Figure 6
Environmental cause-and-effect chain and categories of impact



Note: Arrows in the figure represent CFs in the application of LCIA.
Source: adapted from (Pant *et al.*, 2010).

and ammonium (NH_4^+) and contribute to aquatic eutrophication. Atmospheric deposition of NH_3 and NO_x can contribute to ecosystem acidification and eutrophication (terrestrial and aquatic), N_2O contributes to climate change and to stratospheric ozone depletion, NO_x is a precursor of tropospheric ozone (photochemical oxidant formation), and both NO_x and NH_3 contribute to fine particulate matter formation. For the indicators of photochemical ozone formation potential and particulate matter, the N sources are readily defined, but the methodology for estimating volatile organic compounds and fine particulate matter (PM <2.5μ diameter), respectively, in livestock supply chains is not well defined.

P and PO_4^{3-} contribute mainly to aquatic (freshwater) eutrophication. The N and P impacts to eutrophication and acidification, and respective impact assessment pathways, are covered in Appendix 11. P sources (especially from fertilizers) can also contribute to the indicator of resource depletion, but accurate quantification of some other compounds that can be important for the resource depletion indicator (e.g. indium and nickel) in livestock supply chains can be difficult.

5.1.1 Eutrophication: environnemental cause-effet chain

5.1.1.1 Terrestrial eutrophication

Terrestrial eutrophication originates from the deposition on the land of airborne-N compounds (NO_x from combustion processes, and NH_3 volatilized from agricultural activities). In this case, airborne-N is deposited on soils that have either low N contents or susceptible plant species unable to compete well with species better adapted to take advantage of additional nutrients (Bobbink *et al.*, 1998).

5.1.1.2 Aquatic eutrophication

Nutrients from the various stages of livestock production can be lost to the aquatic environment. This process can provide limiting nutrients to algae and aquatic vegetation in excess of natural rates, which may drive a cascade of changes, including alterations in aquatic species composition, biomass, or productivity in freshwater and marine ecosystems (Henderson, 2015). Fate processes in the environment can also attenuate the impact and contribute to the mitigation of their eutrophication potential (freshwater and marine).

5.1.1.3 Freshwater eutrophication

P is generally the limiting nutrient in freshwater ecosystems, and its emission to these systems often causes freshwater eutrophication (Correll, 1998; Smith *et al.*, 2006). While LCIA typically models freshwater as impacted only by P, either N or P can be limiting (or co-limiting), which will vary with the specific ecosystem characteristics (see Appendix 11).

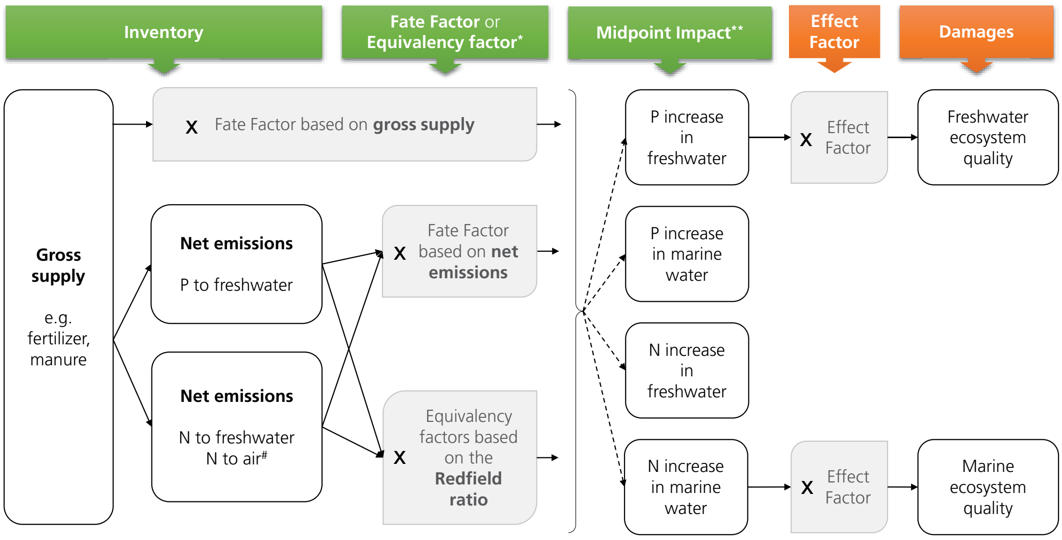
5.1.1.4 Marine eutrophication

N emissions to water, either directly or via atmospheric deposition, generally contribute to marine eutrophication, and any attenuation of the N-content of these emissions associated with fate and transport will mitigate the marine eutrophication potential (Cosme *et al.*, 2017; Nixon *et al.*, 1996).

5.1.2 From the inventory of nutrient emissions to impact assessment for eutrophication

The procedure to apply inventory data from land, animals, processing and upstream stages (calculated using methods in sections 4.2.3, 4.3.3, 4.4.2 and 4.5, respectively for LCIA involves several stages (Figure 7, for eutrophication). The first is to identify the relevant emissions for the impact category being assessed, as described in section 5.1. The estimated emissions of N or P will then need to be multiplied by characterisation factors to define the amount of N or P that has a potential impact. Depending on the LCIA method (described in section 5.2), this characterisation factor can have different components that account for fewer or more environmental mechanisms (corresponding to midpoint and endpoint modelling, respectively). It can be a simple "conversion factor" expressing N compounds in phosphate equivalent (kg PO_4^{3-}e kg N^{-1}; e.g. for CML 2002 method; Huijbregts *et al.*, 2001), or it can include a **fate factor** for midpoint modelling. The fate factor represents the exported fraction of nutrient persisting in the receiving compartment (e.g. freshwater or marine water). For example, some N leaching models estimate the amount of N leached below the root zone of a crop, but some of it may be attenuated (e.g. denitrified) between the zone of leaching and entry to a freshwater body.

Figure 7
Nutrient inventory flow requirements, Fate Factor, Equivalency factor and Effect Factor modelling throughout the eutrophication cause and effect chain in LCA that vary with different LCIA methods



Source: Payen and Ledgard (2017)
* Equivalency factor is only valid for the CML method. ** Depending on the LCIA method, the eutrophication midpoint impact category will include the increased concentration of N and/or P in marine and/or freshwater. # Redeposition of airborne N emissions should be part of the fate modelling.

Potentially, LCIA can be extended to include endpoint impacts. Once the potential amounts of the contributing substances entering the appropriate terrestrial, freshwater and/or marine bodies are defined (can be considered as a midpoint impact), calculating the corresponding damages on ecosystems requires multiplication by an **effect factor**. The effect factor represents the effect of the nutrient concentration increase on the corresponding type of ecosystem (terrestrial or aquatic). In practice, the CFs available in an LCIA method combines the fate and effect components. For example, for freshwater eutrophication (end-point), P emissions to freshwater (sections 4.2.3.5 and 4.2.3.6) could be multiplied by relevant site-specific fate factors (to account for attenuation of P within freshwater) and by a relevant site-specific effect factor (e.g. P concentration increase in species occurrence in freshwater ecosystems; Azevedo *et al.*, 2013b).

The final choice of the LCIA method determines any requirement for inventory (i.e. before applying the characterisation factor, in case the LCIA method does or does not include N leaching in the root zone), because fate modelling choices are embedded within some methods. A nutrient flow accounting summary should be carried out to ensure that all relevant nutrient flows and the appropriate fate, equivalency and effect factors are recognised. For instance, regarding acidification, equivalence factors are used to convert one unit of SO_2 or NH_3 emissions to one unit of SO_2 equivalent whereas the effect factor is based on the decline in richness of vascular plants (Azevedo *et al.*, 2013c).

5.1.3 Acidification

A wide variety of sources (including field-applied synthetic fertilizers and manure, energy and fertilizer production, combustion, etc.) can emit NO_x, NH_3, and SO_x leading to hydrogen ions (H^+) release and/or forming acids which contribute to the potential acidification of soils and water; when the receiving environment's buffering capacity is exceeded by these inputs, it results in soil and lake acidification.

5.2 GENERIC VERSUS SITE-SPECIFIC ASSESSMENT

Eutrophication and acidification can show high spatial variation. The basis for estimation of the spatial differentiation of impacts and characterization models arises from modelling both the locations of given emissions and the relevant conditions that influence the environmental fate and transport of the substances emitted, the resulting ecosystem exposure to these, and the potential effect they have on sensitive receptors.

Efforts to model this spatial variation are reflected in the evolution of LCIA methods from site-generic methods not accounting for the fate of nutrients (e.g. CML 2002 method; Huijbregts *et al.*, 2001) to site-generic methods accounting for regional fate (e.g. ReCiPe 2008; Struijs *et al.*, 2011) and more recently to site-specific methods with a global geographic validity (e.g. Helmes *et al.*, 2012). Several recent impact assessment methods have included spatial differentiation in the modelling work of the terrestrial and freshwater acidification and eutrophication, for example Eco-indicator 99 (Goedkoop and Spriensma, 2000), EDIP2003 (Hauschild and Potting, 2005), ReCiPe (Goedkoop *et al.*, 2013; Huijbregts *et al.*, 2017), LUCAS (Toffoletto *et al.*, 2007) and TRACI (Norris 2003) impact methods. A consistent spatial resolution and geographic scope, where the scale reflects the nature of the impact, is generally lacking among these methods. The UNEP/SETAC (2016) provided guidance on spatio-temporal aspects and related modelling. Methods at an ideal global coverage and spatial differentiation at country scale (at least) are still not available at a necessary maturity level for international recommendation and application.

Recent methods such as Helmes *et al.* (2012) for freshwater eutrophication and Cosme *et al.* (2017) for marine eutrophication are highly relevant due to their global geographical validity and environmental relevance because they include spatially-explicit nutrient fate modelling. However, they cover only N or P (not both N and P) as contributing sources, they have had limited previous application, and their applicability is hampered by the lack of support of regionalization in commercial software. Only OpenLCA and Brightway software enable a regionalized impact assessment, which is currently not available in commercial software such as Gabi or Simapro.

Regionalized and site-specific assessment may help increase the relevance of LCA results (Mutel *et al.,* 2009), but it comes at a price in greater data and modelling requirements. The potential discriminatory power and local environmental relevance offered by spatially differentiated models and impact results may give useful information to LCA studies enabling recommendations for improvement that may be relevant to a site in question (de Haes *et al.*, 2002; Hauschild, 2006). When this approach is followed, it is important that only the impacts are summed across the supply chain, and that inventory remains spatially differentiated in any reporting. This enables the interpretation of the results to properly identify supply chain hotspots through contribution analysis of the full supply chain impacts.

However, when spatial inventory information is lacking, the practitioner must use a method at a lower spatial resolution. This could include the use of aggregated site-specific factors at a global scale (e.g. ReCiPe 2016), or by using site-generic factors from simplified models (e.g. CML).

Site-generic, or global, CFs can be used for those assessments for which spatial information of emission location may be lacking, difficult to obtain, or not relevant in some cases. For "upstream" emissions (such as fertilizer or electricity production), the location of emissions may not be known, and average or "generic" LCI datasets and LCIA CFs may be used.

For freshwater eutrophication, a simplified/composite version has been implemented in commercial software (ReCiPe 2016 based on Helmes *et al.,* [2012], Azevedo *et al.,* [2013a, 2013b and 2014]), but it is not site-specific because only global-scale CFs are currently available.

The CML 2002 (Huijbregts *et al.,* 2001) eutrophication potential indicator represents both terrestrial and aquatic eutrophication. In this single indicator, all emissions of N and P to air, water, and soil and organic matter to water are aggregated according to the Redfield ratio relating to algal growth, providing "equivalency factors".

5.3 RECOMMENDATIONS FOR IMPACT ASSESSMENT IN LCA

Recommendations for the impact assessment indicators outlined in the following sections 5.3.1-5.3.4 were based on reviewing a range of current approaches (including via ILCD 2011 and specific methods noted in section 5.2), which considered global geographic validity, coverage of all contributing sources (e.g. N and P for eutrophication), spatial scale/resolution, extent of modelling of the environmental mechanisms, previous applications, applicability and availability in commercial software.

5.3.1 Eutrophication

Because of the global applicability of the CML method, we recommend its use for the generic midpoint assessment of eutrophication potential (aquatic + terrestrial). However, the limitations due to the absence of fate and effects modelling of nutrient emissions mean that it should be considered as a "worst-case" Tier 1 scoping method. If this impact category appears as a hotspot in the supply chain, then additional effort to more fully characterize the impacts for the geographic region or regions receiving the emissions shall be undertaken. When available, other CFs should be applied for eutrophication if: a) they have greater local relevance (geographic coverage and spatial differentiation of impacts); b) they have been published as peer-reviewed scientific literature, and c) they are publicly available to other users. In this respect, the impact category eutrophication can be differentiated into freshwater, marine and terrestrial. This differentiation into freshwater and/or marine eutrophication requires additional information related to the geographic location of the production system and major inputs, especially feed production. Figure 7 illustrates nutrient inventory flow requirements, fate factor, equivalency factor and effect factor modelling throughout the eutrophication cause and effect chain.

5.3.2 Freshwater Eutrophication

The practitioner should consider whether the specific regions of interest are known to be P- or N-limited. A large majority of freshwater bodies are P-limited, and thus CFs account for P only. This is the baseline assumption in most LCIA methods, where the effect of N is not considered for freshwater eutrophication. If the practitioner is uncertain about which nutrient is limiting in the study region, then both N and P CFs of the CML method (midpoint indicator) should be retained. In cases where a freshwater system is known to be N-limited, the CFs for P compounds can be considered as zero. Unfortunately, no method is currently available to assess the effect of N on freshwater. Where recognized published data is available on attenuation of N and/or P before entry to freshwater bodies, then the relevant fate factors should be used. Future development of LCIA methodology should address the complex interactions between mid-point assessment from increasing N and P, as well as the end-point effects (i.e. increasing primary production).

For practitioners in North America, the robustness of conclusions based on the CML methodology should be assessed against the TRACI methodology, which is developed for North American conditions but uses modelling approaches similar to CML. Practitioners in Europe should adopt the ILCD recommendation to use the ReCiPe model with its associated European P fate factors and should assess it against the CML method. The practitioner shall explain the basis for selection of the final choice of LCIA method(s) used, according to the various points noted above.

5.3.3 Marine Eutrophication

The CML method does not include assessment of marine eutrophication, and therefore we adopt the recommendation of the ILCD to evaluate marine eutrophication (midpoint indicator) with the ReCiPe 2008 model (the ReCiPe 2016 method was not considered because it does not address marine eutrophication). Because this methodology is only validated within the European context, it shall be considered as a Tier 1 screening methodology. For situations in which marine eutrophication is identified as a hotspot, additional evaluation of N emissions to the marine ecosystem are required. Furthermore, practitioners should make a qualitative assessment regarding the likelihood that the fate and effect factors which have been incorporated into this methodology for European conditions are similar to those for the region under study.

5.3.4 Acidification

Again, due to the global applicability of the CML method, we recommend its use for the midpoint assessment of acidification potential (aquatic + terrestrial). Methodologies for acidification all focus on terrestrial acidification. For practitioners in North America, the robustness of conclusions based on the CML methodology should be assessed against the TRACI methodology, which is developed specifically for North American conditions. Practitioners in Europe should adopt the ILCD recommendation, which is the method of Accumulated Exceedance (AE; Seppälä *et al.*, 2006) and should assess it against the CML method.

5.3.5 Sensitivity analysis and current developments

Depending on the goals and scope of the LCA study, reporting of results should include a sensitivity analysis of the methods applied, often achieved through comparison with the alternative method(s).

The limitations of the recommended methods for eutrophication and acidification are the topic of current research – methodology relating to eutrophication and acidification is developing rapidly. It is recommended that the UNEP/SETAC Life Cycle Initiative (http://www.lifecycleinitiative.org/) on eutrophication and acidification be

Table 1: Emerging impact assessment methods for endpoint characterization of emissions with eutrophying and acidifying impacts (with global coverage and spatially differentiated)

Impact category	Substances	Endpoint	Geographic scope	Spatial resolution	Reference
Freshwater eutrophication	P	Plant and animal species richness	Global	Grid cells (0.5°×0.5°)	Helmes *et al.* (2012); Azevedo *et al.* (2013a, 2013b) [implemented in ReCiPe 2016]
Marine eutrophication	N	Animal species richness (6 taxonomic groups)	Global	5,772 river basins	Cosme *et al.* (2017, 2015); Cosme and Hauschild (2017, 2016)
Terrestrial acidification	NO_x, SO_2, NH_3	Plant species richness	Global	Grid cells (2°×2.5°)	Azevedo *et al.* (2013c), Roy *et al.* (2014a, 2012a, 2012b)
Freshwater acidification	NO_x, SO_2, NH_3	Fish species richness	Global	Grid cells (2°×2.5°)	Roy *et al.* (2014b)

Source: adapted from Henderson 2015; Van Zelm *et al.*, 2015

consulted to follow up on new method assessments and recommendations. It is anticipated that within the next 2-4 years, spatially explicit methods, with global coverage, will become more widely available and incorporated in commercially-available LCA software. Table 1 provides a summary of the categories and scales currently under development for some emerging methods. For situations where the recommended methods identify hotspots for specific nutrient related impacts, the practitioner is also encouraged to consider evaluating one of those methods. The interpretation phase of the report should provide the rationale and justification for the selection of the specific model used.

6. Resource use assessment

Nutrients are part of natural biogeochemical cycles, which distribute the nutrients and make them available for plant and animal growth, including where there are no direct sources of them. For example, only some plants are capable of fixing N from the abundant supply of N_2 in the atmosphere through symbiosis with N-fixing microorganisms, thus receiving a competitive advantage over plants without this capability. Reactive N (Nr) forms can also be transformed to inert N_2. In pre-industrial times, microbial N-fixation and denitrification process were approximately equal, and Nr did not accumulate in environmental reservoirs (Galloway *et al.*, 2003). However, currently the N cycle has exceeded a defined planetary boundary (Steffen *et al.*, 2015). Therefore, it is important that assessment of the environmental sustainability of livestock supply chains, incorporates measurement of the efficiency with which nutrients are used (Gerber *et al.*, 2014).

Resource use shall be assessed based on the Life-Cycle Material Use Efficiency concept developed by Suh and Yee (2011). This assessment gives an indication of the efficiency with which nutrients are converted into useful products in a supply chain, without distinguishing between residuals and (co)products, or differentiating by the value of co-products, but considering losses of nutrients and recycling of nutrients within the supply chain. The analysis is purely based on the share of nutrients being used within the supply chain, being used outside the supply chain, wasted or lost to the environment. The analysis allows quantification of nutrient use efficiency at the process level for each life cycle stage individually or in the entire supply chain. Resource use efficiency builds on the concepts of "inputs" and "useful outputs".

Total inputs into the system are input flows as described in Section 4, and include also Nr, which is released from indirect land use changes and Nr releases from the use of energy sources. To distinguish this total "input" from studies looking at the farm scale or supply chain that exclude these emissions (many soil budgets, farm budget papers, etc.), the term "total embodied Nr" (Erb *et al.*, 2009; Leip *et al.*, 2014a) can be used.

Useful outputs include all flows which are considered as co-products or residuals in LCA, while non-useful outputs are identical to all waste and loss flows. Useful outputs include:

- Food and fibre products, which are not considered "waste";
- Accumulation of nutrients in soil reservoirs (soil stock changes), as long as they remain potentially available for future plant uptake, but soil P should be monitored based on P index to avoid the risk of further leaching and runoff;
- Food losses in the post-processing gate food supply chain, as long as they are gainfully reused for agricultural or forestry production (there is no requirement that the nutrients be re-used in the same supply chain to which they were originally input);
- Household food wastes, under the same conditions as those outlined for food losses;
- Sewage sludge, which is gainfully used for agricultural of forestry production (directly or following bio-refinery treatment);

- Emissions of Nr, as long as they are removed from the environment and piped back into agricultural or forestry supply chains before causing any adverse effect. Examples include N that is recovered in animal housing systems with air scrubbers and converted into fertilizers; emissions of NH_3 and NO_x which are deposited on agricultural land or forest ecosystems stimulating plant growth without negatively altering plant and soil health and biodiversity; losses of Nr to aquatic systems which are recovered in (artificial) wetland, algae farms or similar and gainfully used for agricultural production or used as food without negatively altering ecosystem biodiversity;
- Nutrient deposited by grazing animals to natural ecosystems (herbaceous vegetation, marginal areas, shrub lands, sparse vegetation) if it can be shown that the addition of nutrients contributes to maintaining these ecosystems in a healthy state.

Excluded as useful outputs are

- Emissions of nutrients to the environment, which are causing health (e.g. particulate matter, nitrate in drinking water) or ecosystem (e.g. acidification, eutrophication) impacts, even if they are recovered further down the nutrient cascade and gainfully used in agricultural or forestry production.
- Nutrients dispersed in the environment or accumulating in environmental compartments without any positive or negative effect, which cannot be/are not recovered within the time horizon of the assessment[10], including denitrification to N_2, sedimentation in lakes and oceans, P accumulation to soils, etc.
- Food losses and wastes and human excreta dispersed in the environment, landfilled or used in agricultural or forestry production beyond requirements (see Section 4.3.4).

6.1 NUTRIENT USE EFFICIENCY AT EACH PRODUCTION STAGE

Nutrient use efficiency at each stage or process *p* of a supply chain is defined as the total of N or P ($NUE_{nut\ p}$, here nut refers to N or P) in useful outputs (products, recycled nutrients, and stock changes) divided by the total of N or P in external or recycled inputs (Equation 12):

Equation 12

$$NUE_{nut,p} = \frac{F_{prd,p} + \Sigma_q F_{int,p,q} + SC_p}{F_{i,p} + \Sigma_q F_{int,q,p}}$$

where

- $F_{prd,p} = F_{res,p} + F_{cp,p}$ is the sum of the relevant nutrient in products produced in the life cycle stage (or process) *p*;
- $F_{i,p}$ is the sum of the relevant nutrient in all "external" input flows entering the supply chain in process p from either nature (e.g. biological N fixation), industrial process (e.g. synthetic fertilizer) or other agricultural activities (e.g. recycled manure from other livestock species);

[10] Landfills could be mined, or forests could grow on some of the nutrients released; one could define "landfills" as waste flow generally or define a cut-off period beyond which recovery is not considered to be "linked" to the waste flow any more.

- $\Sigma_q F_{int,p,q}$ and $\Sigma_q F_{int,q,p}$ are the sums of all internal flows of the nutrient recycled in the supply chain produced in process p and consumed in any process q or produced in any process q and consumed in process p, respectively. This includes both flows $F_{rec,p,q}$ that are recycled in the supply chain, either in the same process or in another upstream process, and flows that carry nutrients along the supply chain;
- SC_p the nutrient stock changes induced by process p. Stock changes are accounted as positive if there is accumulation of nutrients in the process in pools which can be used to substitute inputs in future process cycles (Leip *et al.*, 2011b; Uwizeye *et al.*, 2016).

or based on matrix calculation, (see Appendix 13 for the matrix construction based on Uwizeye *et al.*, 2016) according to Equation 13:

Equation 13

$$NUE_{nutp} = \frac{F_{PROD,p} + SC'_p}{F'_{INP,p} + F'_{RES,p}}$$

where

' – denotes the transposed matrix

$F_{PROD,p}$ denotes the product output of nutrient from each process of supply chain p;

$F_{INP,p}$ denotes the internal amount of product input of nutrient to each process of supply chain p;

$F_{RES,p}$ denotes the amount of "new" nutrient (resources) input to each process of supply chain p from either nature (e.g. biological N fixation), industrial process (e.g. synthetic fertilizer) or other agricultural activities (e.g. recycled manure from other livestock species).

6.2 LIFE CYCLE NUTRIENT USE EFFICIENCY

The entire supply chain NUE is here called "Life Cycle NUE" (LCNUE) and is expressed as one unit of nutrient in the sum of products of the 'last' stage of a supply chain that produced the end-products of interest, divided by the amount of external nutrient additions to the supply chain to produce it. The quantification of external nutrient additions along the supply chain is based on material flow analysis.

The intensity of "new" nutrient mobilised (F^*_{RES}) at each process, expressed as the amount of nutrients mobilised to produce 1 kg of nutrient in the end-products, is estimated as follows (Suh and Yee, 2011):

Equation 14

$$F^*_{RES,p} = F'_{RES,p} \cdot (F'_{PROD,p} - F_{INP,p} + \widehat{SC}_p)^{-1}$$

Here, $\widehat{SC}_p$ stands for the diagonalized vector of stock changes induced by each process.

For a supply chain covering p stages, LCNUE is therefore calculated as the inverse of the p^{th} element of the vector F^*_{RES}, indicating the quantity of nutrients in the products that are produced in the last stage of the supply chain as a fraction of the total amount of new nutrients mobilised (Suh and Yee, 2011).

Equation 15

$$LCNUE_{nut} = 1 / F^{*}_{RESp}$$

Case studies 1, 2, and 4 in the appendices illustrate contrasting examples of nutrient flows in livestock supply chains with their associated impacts in New Zealand, Uruguay and Rwanda.

7. Interpretation of results

The interpretation of results requires careful identification of significant issues, evaluation, conclusions, limitations and recommendations. In this section, we evaluate the completeness and consistency of life cycle stages and elementary flows in relation to the goals and scope of the assessment, whereas uncertainty analyses and sensitivity analyses provide measures on the accuracy and precision of the assessments. This section is based on ISO 14044:2006 (ISO, 2006b) and EC-JRC, 2010.

7.1 DATA QUALITY

Comprehensive assessment of nutrient flows in LCA involves the collection and integration of data regarding the products, process or activity under study. This data is gathered from different sources; as such, the management of data quality shall be an integral part of the overall process. The data quality requirement is detailed in LEAP feed and animal guidelines (e.g. FAO, 2016a), which are based on ISO 14044:2006 (ISO, 2006b).

7.2 SIGNIFICANT ISSUES

Through this stage, the results of inventory and impact assessment phases are structured to help determine the significant issues in accordance with the goal and scope definition. First, the main contributors to the inventory and impact assessment vary according to the life cycle stage and the relevant impact category. The contribution of each contributor can be assessed through contribution analysis, which separates the aggregated results of the inventory analysis or impact assessment into a number of constituting elements (Heijungs and Kleijn, 2001). Second, the methodological choices can significantly influence the results. They include the allocation rules, system boundary, assumptions, foreground and background data used and impact assessment approach ISO 14044:2006 (ISO, 2006b).

7.3 EVALUATION

Evaluation shall be performed to establish and enhance the confidence in, and the reliability of, the results of the inventory and LCA, including the significant issues identified in section 7.2. The evaluation involves a completeness check, sensitivity check in combination with scenario analysis and uncertainty analysis and consistency check.

7.3.1 Completeness check

The completeness check enables practitioners to ensure that all relevant information such as flows, stage of a supply chain, data, and interactions are available and complete as well as aligned with the goals and scope. If any relevant information is missing or incomplete, the necessity of such information to satisfy the goal and scope shall be considered. In case of cut-off, as described in LEAP feed and animal guidelines (e.g. FAO, 2016b), it shall be recorded and justified. For these guidelines, it is recommended to include as many nutrient flows as possible in the inventory to enable answering potential questions about missing flows. The mandatory steps and the choice of indicators for nutrient accounting for LCA and resource use assessment are illustrated in Table 2. All elementary flows that are relevant for the impact assessment for eutrophication and acidification should be included. A more comprehensive way of estimating the impact of missing flows, methodological choices, and assumptions is to conduct a sensitivity analysis.

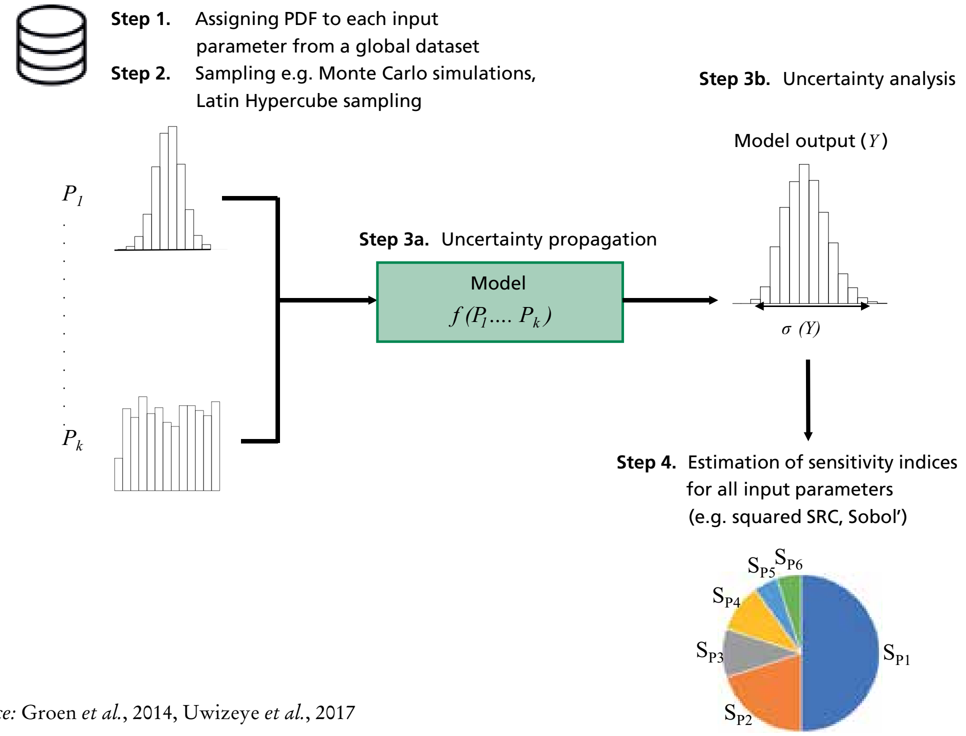
7.3.2 Sensitivity check

The effect of uncertainties of input parameters is evaluated through sensitivity analysis, which is recommended to assess the reliability of the final results and will support the conclusions and recommendations of the nutrient assessment and LCA results. Two sensitivity analysis methods are used. Local sensitivity analysis is based on changing of input parameters around a reference value and ranking the magnitude of the effect of each parameter (Campolongo *et al.*, 2007). An example of such an approach modifying parameters one by one is provided by Tittonell *et al.* (2006). Global sensitivity analysis is based on the variation of input parameters according to their distribution function, and subsequently determines how much each parameter explains the model output variance (Groen *et al.*, 2014a; Pianosi *et al.*, 2016; Saltelli *et al.*, 2008; Uwizeye *et al.*, 2017). A practical example of a global sensitivity analysis is presented by Uwizeye *et al.* (2017) for N use assessment in mixed dairy systems. Here, the practitioner shall use one of these approaches. The local sensitivity analysis is simple and easy to conduct. However, its results are less reliable because it does not consider the entire dimension of the variability of the input parameters or the interactions between them. The global sensitivity analysis is more robust, even though it can be time consuming in case of detailed input data. It consists of four main steps illustrated in Figure 8.

Table 2: Mandatory steps and the choice of indicators for nutrient accounting for LCA and resource use assessment

Step of the assessment	LCA	Resource use efficiency
Goal and Scope definition	Mandatory	Mandatory
Inventory	Tier 1: Recommended for Scoping analysis Tier 2: Recommended for supply chain and regional assessment Tier 3: Complex model specific to a given production system	Tier 1: Recommended for Scoping analysis (input-output methods) Tier 2: Recommended for supply chain and regional assessment Tier 3: Detailed and specific models
Data	Primary and secondary data Data quality assessment is mandatory	Primary and secondary data Data quality assessment is mandatory
Choice of Pressure indicators	Expressed per functional unit (FU) N_2O emissions NH_3 emissions NO_x emissions N run-off and leaching losses P run-off and leaching losses	Pressure indicators N losses ha^{-1} P losses ha^{-1} Example of footprint indicators N losses FU^{-1} P losses FU^{-1}
Efficiency indicators	None	NUE (N or P) for each stage of the supply chain Life cycle NUE (N or P) N or P circularity (see section 7.4.3)
Impact assessment indicators	CML, ReCiPe, TRACI, Accumulated Exceedance Eutrophication potential Acidification potential	None

Figure 8
Stepwise global sensitivity analysis



Source: Groen *et al.*, 2014, Uwizeye *et al.*, 2017

7.3.2.1 Uncertainty analysis steps

Step 1. Selection of the probability density functions (PDFs) for each input parameter based on survey data. Practitioners shall select PDFs that give the best goodness-of-fit. If literature data is used without any information about their variance, IPCC (2006) recommends using a coefficient of variation of 10 percent or 20 percent. Uwizeye *et al.* (2017), for example, assigned triangular distribution for the emission factors described by fixed minimum and maximum and a specific likely value, normal distribution for the data defined by an average and a standard deviation and uniform distribution for the data described by minimum and maximum values.

Step 2. Sampling. Groen *et al.* (2014b) provide different sampling techniques including Monte Carlo Simulations (MCS), Latin hypercube sampling (LHS), Quasi Monte Carlo Sampling, Analytical uncertainty propagation, Fuzzy interval arithmetic or Bootstrapping. Here we describe several options for uncertainty analysis, their application, and advantages and disadvantages for practitioners to choose which one is suitable based on the goals and scope (Table 3). MCS can be used to estimate uncertainty in stocks and flows of N and P by drawing numbers from a probability distribution for each variable. This process can produce thousands of outcomes, combining numerous random estimates for each of the variables and for all the variables selected and considered uncertain. Ortiz-Gonzalo *et al.* (2017) present an example of the use of MCS to identify sources of uncertainty in farm-scale analyses of GHG emissions due to management of crops and livestock. The analysis was also useful for identifying manure management as one the most important hotspots driving GHG fluxes in a mixed farm. LHS is in principle a technique

Table 3: Example of methodological options for uncertainty analysis

Method	Advantages	Disadvantages	Further reading
Monte Carlo simulations	Relatively simple to apply. Accounts for large and small uncertainties. Accounts for non-linearity and correlations.	It assumes input variables are not correlated. Because sampling is random, samples can be clustered around low probability ranges	Gilks, W.R., S. Richardson, D.J. Spiegelhalter (1996) Markov chain Monte Carlo in practise, Chapman and Hall, London, United Kingdom
Latin hypercube sampling	Produces similar robust uncertainty analyses as MCS, using fewer simulations	Cannot handle a large number of variables. Because it samples intervals for each variable, it has large computing requirements.	Helton, J.C., F.J. Davis (2003) Latin hypercube sampling and the propagation of uncertainty in analyses of complex systems. Reliab. Eng. Syst. Safety 81,23–69
Bootstrapping	Simple and independent of the distribution of the population. Small samples can be used. It works with non-linearity in the variables	It cannot be used when the populations are heavily tailed (skewed)	Efron B., R. J. Tibshirani, (1998) An introduction to the bootstrap, Chapman & Hall, CRC

similar to MCS. However, it stratifies the probability distribution of input parameters into intervals, and samples from that interval instead of completely randomly like in MCS. This reduces the number of interactions or simulations to achieve robust uncertainty analysis. Van Wijk *et al.* (2009) use LHS to estimate uncertainties in N and P fluxes at the farm level, and how they influence overall farm performance. Bootstrapping is a simple technique to estimate statistics from unknown populations (variables) using re-sampling with the replacement of relatively small samples. This technique is useful for conducting farm analysis with incomplete data and for handling uncertainties. For example, Schrade *et al.* (2012) use bootstrapping to estimate ammonia emission factors from dairy farms, addressing uncertainties in model parameter estimates.

Step 3. Uncertainty propagation and uncertainty analysis of the results. The uncertainties of all input parameters are propagated through the inventory model based on sampling techniques from PDF. Uncertainty analysis is designed to estimate the overall robustness of the analysis and the contribution of individual categories and components to this robustness. By identifying uncertainties, practitioners can take different actions. For example, uncertain estimates can lead to follow up and in-depth studies, and to cautious recommendations of practices that may require further testing. Uncertainty analyses are critical for assessing complex systems performance, where implementation of interventions requires an understanding of relative effects. Once uncertainties are identified, additional techniques such as bootstrapping can be used to deal with uncertain data. The statistical results of the uncertainty propagation describes the uncertainty of the outcomes. However, this information is not complete because it does not give the contribution of each input parameter to the outcome variance. Table 3 shows examples of uncertainty analysis methods.

7.3.2.2 Sensitivity analysis steps

Step 4. Sensitivity analysis. There are several methods for the sensitivity analysis. Squared standardized regression coefficients (see Uwizeye *et al.*, 2017) and the Sobol method (Groen *et al.*, 2016) are mainly used to estimate the contribution of

Table 4: Examples of methods for completeness and sensitivity check

Interpretation domains	Concerns	Recommended method	Back-up method
Life cycle stages, nutrient flows	Completeness: system definitions, missing categories, and stages, or missing components and flows	Sensitivity analysis	Contribution analysis (based on expert knowledge)
	Consistency: Allocation rules and system boundaries	Scoping analysis (with secondary data)	Checklist, Best practice, Peer review
Uncertainty	Data quality	Uncertainty analysis	Qualitative description of limitations
	Knowledge gaps	Uncertainty analysis	Qualitative description of limitations
	Identification of hotspots	Sensitivity analysis and uncertainty analysis	Expert knowledge

each input parameter to the variance of the results. The parameters are classified as important or non-important parameters. Only the important parameters need to be established with high-quality data to reduce the uncertainty and increase the robustness of the study. Regardless of the sensitivity analyses that are used, the range in possible values within the models shall be appropriate for that particular measurement and the scenario to which is it applied.

7.3.3 Consistency check

To better interpret the results of LCA and nutrient flow analysis, it is recommended to perform a consistency check. It consists of determining whether the assumptions, methods and data is consistent with the goals and scope. This consistency is evaluated for data quality, regional and/or temporal differences in the data, allocation rules, system boundary and impact assessment method. Table 4 shows examples of methods for completeness, consistency and sensitivity checks.

7.4 ADDITIONAL INDICATORS TO SUPPORT THE INTERPRETATION OF NUTRIENT BUDGET ANALYSIS

This section addresses indicators which are specific for assessment of nutrient flows. It gives guidance on which indicators should be included in a report to enable wide comparability, for example comparing with "agri-environmental" databases.
Three indicators are proposed:

- N and P footprints
- Nutrient surplus
- Circularity indicator

7.4.1 Nitrogen and phosphorus footprints

N and P footprints are the sum of emissions that are caused by the production of one unit of final product. To calculate the total emissions, all processes need to be scaled so that the quantity of intermediate products produced equals the quantity required if subsequent supply chain stages (Heijungs and Suh, 2002) and emissions are allocated to different co-products along the supply chain according to the rules defined in sections 4 and 5 and in previous guidelines.

The N footprint of a livestock supply chain includes emissions of molecular N (N_2), which does not contribute to any environmental impact but represents a "waste" of resources. N footprints of food products are frequently used as a tool for communication of the overall pressure on the environment with respect to N (Galloway *et al.*, 2014; Hutton *et al.*, 2017; Leach *et al.*, 2012; Leip *et al.*, 2014b; Pelletier and Leip, 2014; Pierer *et al.*, 2014; Shibata *et al.*, 2014; Stevens *et al.*, 2014).

For these guidelines, N and P footprints could be calculated to the farm gate, the primary processing gate of the animal products or for the entire life cycle.

7.4.2 Gross nutrient surplus

The gross nutrient surplus (GNS) indicator is an agri-environmental indicator used as a proxy for agricultural pressure on the environment from agricultural production. It is calculated as the difference between total nutrient inputs and total nutrient outputs at a land or production unit level (Leip *et al.*, 2011b). It thus includes all nutrient losses occurring from soil management during crop cultivation (until harvest) and all nutrient losses from manure in livestock housing and manure storage systems.

The GNS is expressed in kilograms of nutrients per hectare of agricultural land (kg N or P ha^{-1}), commonly reported over a one-year timeframe.

Equation 16

$$GNS = \frac{F_{i,farm} - F_{o,farm}}{A}$$

Inputs (F_i) and outputs (F_o) to be considered are listed in Eurostat (2013) and Özbek and Leip (Özbek and Leip, 2015), whereby input and output flows and area (A) are quantified with respect to the boundaries of farms for the supply chain in question. This may not necessarily be "a farm" but could include several farms that are supplying feed for a livestock supply chain (Leip *et al.*, 2014b). Thus, all land used for feed and animal production shall be accounted for, but there can also be value in assessing component farms or areas to identify hotspots.

As for the quantification of the resource use efficiency indicators, soil stock changes that are recoverable in future cropping seasons are considered as being included in the outputs. Case study 3 in the appendices illustrates gross nutrient surplus in the egg production systems in Sweden.

7.4.3 Circularity indicator

In livestock supply chains, not all nutrients that are required in the processes can be used directly in the final products, but are available to be recycled and used as input in a different process of the supply chain. This is referred to as "recycling". When nutrients are recycled instead of being used in a product, the recycling of nutrients ensures that they are not wasted or lost to the environment and can be used as an input in the same or another supply chain. If this happens, input of "new" nutrients from external sources can be avoided. Circularity is therefore a measure of the degree to which nutrients that are not used in the final product(s) are re-used in the processes, substituting input of new/external nutrient inputs. Even though recycling of nutrients increases the life-cycle nutrient use efficiency, a separate indicator of the degree of circularity enables separating such "logistic" effects on the efficiency from process formulation effects. Therefore, circularity indicators from a perspective of either input (*ICirc*) or output (*OCirc*) flows may be used.

For inputs, the circularity analysis distinguishes between 'new' inputs $F_{i,new}$ (which include mineral fertilizer and biological fixation, as well as Nr losses from energy use) and 'recycled' inputs, regardless of whether they originate from the same or another supply chain (atmospheric deposition, organic fertilizers, animal excreta, feeding food processing by-products or food waste). Hence, they could originate either from external sources ($F_{i,rec}$) or by being recycled in the supply chain itself (F_{rec}).

For outputs, the circularity analysis distinguishes between products intended for "consumption" (co-products F_{cp}) versus those which are recycled (residues F_{res} and recycling F_{rec} flows).

There are two possible circularity indicators, i.e. from the perspective of input flows (*ICirc*) and from the perspective of output flows (*OCirc*). They are defined as given in Equation 17 and Equation 18. The circularity indicators can be quantified for individual life cycle stages or for partial or entire supply chains.

Equation 17

$$ICirc = \frac{F_{i,rec} + F_{rec}}{F_{i,new} + F_{i,rec} + F_{rec}}$$

Equation 18

$$OCirc = \frac{F_{res} + F_{rec}}{F_{cp} + F_{res} + F_{rec}}$$

7.5 CONCLUSIONS, RECOMMENDATIONS AND LIMITATIONS

The final part of interpretation is to draw conclusions derived from the results, provide answers to the questions raised in the goals and scope definition stage, and recommend appropriate actions to the intended audience, within the context of the goal and scope, explicitly accounting for limitations to robustness, uncertainty and applicability.

Conclusions derived from the study should summarize supply chain "hotspots" derived from the contribution analysis and the improvement potential associated with possible management interventions. Conclusions should be given in the strict context of the stated goals and scope of the study, and any limitation of the goals and scope can be discussed a posteriori in the conclusions.

As required under ISO 14044:2006, if the study is intended to support comparative assertions (i.e. claims asserting difference in the merits of products based on the study results), then it is necessary to fully consider whether differences in method or data quality used in the models of the compared products impair the comparison. Any inconsistencies in functional units, system boundaries, data quality, or impact assessment shall be evaluated and communicated. Additional guidance for comparability between studies are provided in LEAP feed and animal guidelines (e.g. FAO, 2016b).

Recommendations are based on the final conclusion of the LCA or nutrient use assessment study. They shall be logical, reasonable, plausibly founded and strictly related to the goal of the study. Recommendations shall be given along with limitations to avoid their misinterpretation beyond the scope of the study.

7.5.1 Good practice in reporting LCA results

The results and interpretation shall be fully reported, without bias and consistent with the goals and scope of the study. The type and format of the report should be appropriate to the scale and objectives of the study, and the language should be accurate and understandable to the intended user to minimise misinterpretation.

The description of the data and method shall be included in the report in sufficient detail and transparency to clearly show the scope, limitations and complexity of the analysis. The selected allocation method used shall be documented, and any variation from the recommendations in these guidelines shall be justified.

The report should include an extensive discussion of the limitations related to accounting for a small numbers of impact categories and outputs. This discussion should address:

- Negative impacts on other environmental criteria;
- Environmental impacts;
- Multifunctional outputs other than production (e.g. economic, social, nutrition).

If intended for the public domain, a communication plan shall be developed to establish accurate communication that is adapted to the target audience and is defensible.

7.5.2 Report elements and structure

The following elements should be included in the LCA report (see ISO 14044:2006 [ISO, 2006b]):

- Executive summary typically targeting a non-technical audience (e.g. decision-makers), including key elements of goals and scope of the system studied and the main results and recommendations, while clearly giving assumptions and limitations;
- Identification of the study, including name, date, responsible organization or researchers, objectives of/reasons for the study and intended users;
- Goal of the study: intended applications and targeted audience, methodology (including consistency with these guidelines);
- Functional unit and reference flows, including overview of species, geographic location and regional relevance of the study;
- System boundary and unit stages (e.g. farm gate to primary processing gate);
- Materiality criteria and cut-off thresholds;
- Allocation method(s) and justification if different from the recommendations in these guidelines;
- Description of inventory data: representativeness, averaging periods (if used), and assessment of quality of data;
- Description of assumptions or value choices made for the production and processing systems, with justification;
- LCI modelling and calculated LCI results;
- Results and interpretation of the study and conclusions;
- Description of the limitations and any trade-offs;
- If intended for the public domain, the report should also state whether the study was subject to independent third-party verification.

7.5.3 Critical review

Internal review and iterative improvement should be carried out for any LCA study. In addition, if the results are intended to be released to the public, third-party verification and/or external critical review shall be undertaken (ISO 14025, ISO 2006c) to ensure that:

- Methods used to carry out the LCA are consistent with these guidelines and are scientifically and technically valid;
- Data and assumptions used are appropriate and reasonable;
- Interpretations take into account the complexities and limitations inherent in LCA studies for on-farm and primary processing;
- The report is transparent, free from bias and sufficient for the intended user(s).

The critical review shall be undertaken by an individual or panel with appropriate expertise, for example suitably qualified reviewers from the agricultural industry or government or non-government officers with experience in the assessed supply chains and LCA. Independent reviewers are highly preferable.

The panel report and critical review statement and recommendations shall be included in the study report if publicly available.

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APPENDICES

Appendix 1

Tiered approaches

The methodology to use for specific supply chains and regional assessment are principally the same, even though generic (representative) data might be used for regional scale assessment whereas measured data might be used for specific supply chain assessments. Also, more simple methods can be used in regional scale assessments if data availability is insufficient for applying more accurate methods, but the choice of methods could be stricter in the case of specific supply chain assessments. For example, the quantification of total N excretion from dairy cattle should in both cases ideally be based on an "animal-budget" model, accounting for total nutrient intake in the feed, total nutrient retention in livestock and their products and total nutrients excreted. If representative feed rations for dairy cattle are not known, the use of typical N-excretion rates listed, for example region-specific values from the IPCC guidelines, could be adequate for national assessments, but this is not adequate in assessments of specific supply chains.

In other cases, different methodologies might be recommended. For example, available measurements of soil stock changes are scarce in many countries, and the models proposed in these guidelines for regional scale assessment can only deliver approximations. However, for accurate assessments at a supply chain level, measurements of soil stock changes may be necessary.

Once all relevant N and P flows for the supply chain have been identified, the methods for their quantification must be selected. A cut-off of flows can be applied where minor flows are unable to be quantified and where the contribution of the flow to the total nutrient input is less than 1 percent (FAO, 2016b, section 8.4.3). The more data is available, the more detailed disaggregation of the methods can be applied in the assessment. In analogy to IPCC definitions (IPCC, 2006), three Tier levels are distinguished:

Tier 1: Tier 1 refers to generic methods or default emission factors per unit of product or activity

- For regional assessments, Tier 1 method should be used only in data poor situations or if the flow is not significant of the nutrient cycle assessment. For example, N_2O emissions are amongst the most important for comprehensive LCIAs when the climate change impacts need to be quantified, but if the focus is on resource efficiency, eutrophication and acidification, then flows of N_2O represent only a small fraction of loss flows and it is usually sufficient to apply IPCC default emission factors.
- For the assessment of specific supply chains, Tier 1 methods should only be applied for flows which amount to a maximum 1 percent of the total embedded input flows at the specific stage where the flows "starts" from. The total flows assessed with a Tier 1 method at a specific stage should not be more than 5 percent of total embedded input flows.

Tier 2: Tier 2 methods provide more detailed calculation that better reflect the national or specific circumstances where the flow occurs.

- For regional assessments, this means often that the activity data is split into sub-groups which differ significantly in their characteristics (relevant for the estimation of the flow strength, e.g. different N content in different plant compartments) or directly on their "flow factor" (e.g. different manure management systems; or differentiation between crops on mineral or organic soils). In other cases, Tier 2 methods require the estimation of additional parameters used in the methodology, such as the digestibility of feed to estimate total energy and nutrient intake.

- For specific supply chain assessments, the difference between Tier 1 and Tier 2 is minor - instead of generic flow factors, they are estimated based on additional activity data that need to be surveyed (for foreground processes) or estimated (for background processes) to allow the use of disaggregated flow factors, and/or additional parameters that need to be estimated.

For nutrient assessments, Tier 2 methods are recommended. If not all data is available to use Tier 2 methodologies, effort needs to be undertaken to collect all necessary data. Only in case this is not possible or in case a scoping study has established that a flow is smaller than 1 percent of the total input flows of a pool, compilers can use a Tier 1 methodology.

Tier 3: Tier 3 approaches are the most detailed methodologies and provide potentially the most accurate estimates.

- For regional assessments, Tier 3 methods are often mechanistic models. These models need to be rigorously calibrated and validated for national circumstances. Generally, mechanistic models require a large amount of input data, including soil and climatic data and run at high spatial and temporal resolution. Leip *et al.* (2011) have shown in the example of N_2O fluxes from agricultural soils, process-based models do not outperform more simple methodologies due to the lack of experimental observations and risk of producing outliers at the margin or outside the domain spanned by the experimental observations. Despite the theoretical power of mechanistic models to interpolate to conditions not actually monitored, care must be taken. Generally, they do not necessarily require fewer experimental observations than empirical models which would lead to stratified flow factors (Tier 2).
- For specific supply chains, Tier 3 methods are either mechanistic models or actual measurements. Applying mechanistic models to specific supply chains does not suffer from the aggregation error and need "only" to be calibrated and validated for the specific farm conditions. Measurements should follow sampling and measurement protocols according to current state-of-the-art.

Tier 3 methods are very data intensive. If such methods are available to the practitioner and have been published and validated for the relevant region or supply chain, then Tier 3 methods are suitable to reduce uncertainty and/or provide the means for specific investigations (e.g. assessment of scenarios, mitigation options etc.). These methods are optional where high quality data is available and accepted methodology exists.

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Appendix 2

Nutrient assessment – relevant guidelines

For most of the nutrient flows that need to be quantified in feed supply chains, existing guidelines have defined relevant methods. The LEAP Feeds Guidelines (FAO, 2016) cover all aspects of feed production and material flows associated with production of a wide range of crop and pasture systems through to the animal's mouth. The LEAP animal supply chains guidelines cover animal-related flows. However, these guidelines provided limited guidance on N and P flows and losses, which are the focus of this adjunct Guidelines. Other useful information sources are:

- Annex Agriculture to the UNECE Guidance document on national N budgets (Leip *et al.*, 2016);
- Eurostat/OECD Nutrient Budgets Handbook (Eurostat 2013);
- IPCC (2006) guidelines for national GHG emissions inventories, in particular Volume 4 (Agriculture, Forestry and Other Land Use, AFOLU), Chapter 10 (Emissions from Livestock and Manure Management) and Chapter 11 (N_2O emissions from soils, and CO_2 emissions from lime and urea application);
- EMEP/EEA air pollutant emission inventory guidebook (EEA, 2016), in particular Part B.3.D (Crop production and agricultural soils).

These guidelines serve different reporting obligations at a country level: annual greenhouse gas inventories need to be reported to the UNFCCC and the Kyoto Protocol, parties to the UNECE must report air pollutant inventories to EMEP under the Convention on Long-Range Transboundary Air Pollution, and member countries of the OECD and Eurostat are requested to report agricultural Gross Nutrient Balances. Reporting of national N budgets is recommended in Annex IX of the revised Gothenburg Protocol and the EU NEC Directive (EU, 2016).

These guidelines are not independent, but rather build together a consistent framework for the quantification of N and P flows in agriculture. While EEA (2016) focuses on air pollutants (NH_3 and NO_x), IPCC (2006) provides guidance for the quantification of greenhouse gases (N_2O). As indirect N_2O emissions are a consequence of agricultural losses of reactive N to the atmosphere and to the hydrosphere, for example, in Europe, it is good practice to use national GHG inventory methods to estimate indirect N_2O emission through volatilization (e.g. as identified in EEA, 2016).

Eurostat (2013) builds on the previous two guidelines, and provides methods for additional flows, i.e. N inputs via N fixation, atmospheric deposition, seeds and planting materials, and crop residues, and N outputs via crop and fodder production and crop residues. GHG inventories require only the estimation of net crop residues removal, however - in the ideal case described in Eurostat (2013) - total crop residues production needs to be accounted for to properly derive farm nutrient efficiency indicators.

Similarly, Leip *et al.* (2016) give guidance on obtaining the best possible available data estimated for one of the reporting obligations or improve N estimates in cooperation with the reporting agencies.

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Appendix 3
Biological nitrogen fixation

In livestock production systems, inputs of N from biological fixation of atmospheric N_2 can occur mainly via symbiotic association between legumes and rhizobia. However, there can also be small-moderate amounts of N_2 fixation via free-living microorganisms in soils.

LEGUME N_2 FIXATION

Section 4.2.2.2 of the main Guidelines described the principles for estimating legume N_2 fixation based on estimation of legume yield, N concentration, proportion of total N derived from atmospheric N_2 fixation (Ndfa; remaining N is from soil or added N and Ndfa is generally assumed to be the same for above and below-ground tissues) and a whole plant factor (to account for fixed N in roots and non-harvested plant material). This Appendix gives further information on these components and some tier 1 estimates of legume N_2 fixation.

The average N concentration for a legume species is relatively constant and is best based on primary data. However, it can vary with stage of growth, season, climatic conditions (particularly for pasture legumes) and these factors should be recognised when obtaining relevant data on average N concentration. Where primary data on N concentration of legumes is not available then it should be based on published data for the relevant legume species for the region of production.

For a given species, Ndfa varies with N availability in soils (soil N mineralisation, N inputs from fertilizers, animal deposition) and biophysical parameters such as soil pH and moisture (Peoples *et al.,* 1995). Average values for Ndfa are summarised in Table A3.1, as well as typical values for the amount of N fixed per tonne of dry matter (DM) “harvested” and a factor for conversion to whole-plant N_2 fixation.

The Ndfa value of 90 percent for grassland legumes is typical for cutting systems. However, in grazed pastures without added N fertilizer, the average Ndfa is lower at

Table A3.1: Mean N_2 fixation rates for some nodulated legumes cultivated for animal feed, and example coefficients to include whole-plant N_2 fixation

Species	Mean proportion of N fixed (Ndfa %)	N fixed kg N t DM^{-1} in aboveground biomass	Coefficient for whole-plant N_2 fixation (f_{yield})
Alfalfa, sainfoin, vetches, lotus, birdsfoot trefoil	70-80 %	20	1.7 (for white clover, due to stolons) 1.5 (all other species)
Red clover	80-90 %	26	
White clover (in mixture with grasses)	80-95 %	31	
Féverole, lupin	70-80 %	20	1.2 - 1.4
Soyabean, peanuts	65-70 %	18	
Peas, chickpea, lentils	60-65 %	18	
Beans	40%	15	

Source: Anglade *et al.*, 2015; Peoples *et al.*, 2009; Voisin and Gastal 2015; Jørgensen and Ledgard 1997

75-80 percent due to effects of N return in animal excreta (Ledgard 2001). When associated with grasses or cereals and not fertilized, the fixation rates of legumes are higher compared to monocultures, as associated grasses are competitive for mineral N in soils. Studies in legume/grass pastures receiving N fertilizer indicate that the amount of N fixed decreases by an average of approximately 0.3 kg N kg fertilizer-N^{-1} (e.g. Ledgard *et al.*, 2001).

A whole-plant-factor can be used to account for the amount of N fixed below the usual harvest height (c. 5 cm). Additional fixed N below cutting or grazing height (including in stolons and roots) typically adds 1.2-1.7 times the amount of fixed N estimated in harvested legumes (Table A3.1). A factor of 1.7 is relevant for legumes with stolons or rhizomes (e.g. clovers), while 1.5 is appropriate for other legumes (e.g. review from Anglade *et al.*, 2015).

For legumes in grazing pasture systems, the legume yield can be estimated from the calculated pasture intake by animals, a utilization factor $f_{utilization}$ and an estimated proportion of legumes in the pasture $f_{legumes}$ (section 4.2.2.2). The $f_{utilization}$ multiplication factor varies between about 1.25 and 2.0 for typical utilisation levels of 50-80 percent depending on grazing intensity.

The $f_{legumes}$ factor in pasture varies seasonally and can fluctuate over time, and therefore a weighted average value should be used to represent a longer-term average. For example, the average $f_{legumes}$ in grazed pastures in temperate systems receiving no N fertilizer in the review by Ledgard (2001) was 16 percent (dry weight basis). Higher values (30-35 percent) are targeted in intensive grass-clover based dairy systems of Western Europe (e.g. Rasmussen *et al.*, 2012) to ensure high yields without N inputs and avoid a strong increase of N leaching losses.

N_2-FIXATION FROM FREE LIVING ORGANISMS

The contribution of free-living organisms to global N-fixation rates is generally considered to be minor because of the scarcity of suitable carbon and energy sources (Wagner, 2012). Heterotrophic free-living N_2 fixers that use plant residues such as straw and leaf litter appear to contribute only small amounts of N to dry-land agriculture, mostly <5 kg N ha^{-1} yr^{-1} (Unkovich *et al.*, 2008). However, some measurements of N_2 fixation by free-living organisms have exceeded 5 kg N ha^{-1} per year and been up to 20 kg N ha^{-1} during the growing season in cereal fields in humid environments (Neyra and Dobereiner, 1977). A study in Australia of an intensive wheat rotation farming system demonstrated that free-living microorganisms contributed 20 kg N ha^{-1} per year to the long-term N needs of this crop system (30-50 percent of the total needs; Vadakattu and Peterson, 2006). Also, free-living N_2 fixation in flooded rice production systems has been shown to be up to 30 kg N ha^{-1} (Firth *et al.*, 1973), and in tropical crops such as sugarcane in the order of 10–65 kg N ha^{-1} per year (e.g. Boddey *et al.*, 1995), and up to 160 kg N ha^{-1} (Bohlool *et al.*, 1992).

Thus, the amount of N fixed by free-living soil bacteria is generally small, i.e. < 5 kg N ha^{-1} yr^{-1} (Paul and Clark 1996; Unkovich *et al.*, 2008; Vitousek *et al.*, 2002), with the exception of some high values found mainly in humid tropical regions. However, some methodology used has been questioned and the data is variable and inadequate to obtain regional average values. It is recommended that N_2 fixation from free-living organisms should not be included in accounting for N flows unless published local data is available. In humid and tropical conditions, a literature search should be done for the region being studied.

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Appendix 4

Estimating the soil non-labile phosphorus pool

The residual value of previously applied conventional P fertilizers is indicated to decline with time after application (Burkitt *et al.*, 2002; Bolland and Gilkes, 1998; immobilisation flows). This is due to the rapid conversion of soluble forms to more stable less soluble forms, through microbial processes, sorption and precipitation.

The prevalence of insoluble P forms in the soil pool, and their subsequent availability is dependent on a range of factors, including soil characteristics, and the form of additions of P made to the soil. Adsorption and precipitation into relatively unavailable pools is decreased where carbon is available to drive microbial P uptake (Drinkwater and Snapp, 2007; Kouno *et al.*, 2002). Addition of manure-based P sources has been observed to extend the agronomic availability of the nutrient relative to an inorganic application (Redding *et al.*, 2016). While it is arguable that all sorbed and precipitated P forms can theoretically again become agronomically available (Barrow, 1986), observations that the residual value of previously applied P declines with time after application (Bolland and Gilkes, 1998) suggest that sorption processes may dominate the processes that release P in such systems.

Conceptually the soil P stock could be considered to be made up of the following pools:

Equation A4.1

$$P_{stock} = P_{sorbed} + P_{actively\ cycling\ pool} + P_{solution}$$

where, $P_{actively\ cycling\ pool}$ represents relatively labile P (which could be organic or inorganic). Likewise, P_{sorbed} is an aggregation of less available P in inorganic, organic, and precipitated forms. A proportion of P_{sorbed} is considered to be effectively unavailable on the time scale of seasonal agricultural production and is termed **recalcitrant** here ($P_{recalcitrant}$).

Tier 1 approach: as described in the main text, the limit to the recalcitrant P storage capacity is conservatively assumed to be (kg ha^{-1}):

Equation A4.2

$$P_{recalcitrant} < 50 \cdot BD \cdot \frac{Depth \cdot 10000}{1000^2}$$

where BD is the bulk density of the soil (kg m^{-3}) and 50 mg [kg of soil]$^{-1}$ conservatively estimates the sorption at the eutrophic trigger concentration. Residual P retained in the soil is assumed to move to this pool after three seasons.

Movement to $P_{non\text{-}labile}$ forms is assumed to decrease the potential for plant utilisation and the vulnerability to transport by water in dissolved forms.

Table A4.1: Example soil P storage behaviour based on sorption for up to 196 days

Soil	n[1]	k[2]
1. Quality agricultural soil with high iron content. Red clay soil.	229	0.374
2. Arable cracking clay	142	0.431
3. Arable black cracking clay	140	0.293
4. Sandy soil	7.64	0.771

[1] Empirical constant related to the bonding strength; determined via 18 hours 1:10 soil to solution batch sorption isotherm (method 9J in Rayment and Lyons. 2010).

[2] Empirical constant related to the sorption index; determined via 18 hours 1:10 soil to solution batch sorption isotherm (method 9J in Rayment and Lyons. 2010).

Source: Redding *et al.*, 2006

Tier 2: In summary, this method involves the simplified use of sorption curve data representative of an area's soils in a modification of Equation A4.1, where residual P is assumed to move into the $P_{recalcitrant}$ pool after three seasons, with a limit to the capacity of this pool:

Equation A4.3

$$P_{recalcitrant} < S_{eutrophic\ trigger} \cdot BD \cdot D_{rooting} \cdot 10000 / 1000^2$$

where $S_{eutrophic\ trigger}$ represents a justifiable trigger concentration (mg kg of soil^{-1}) for eutrophication of waterbodies that may be contaminated by leachate or lateral/interflow water from an area. The effective depth of rooting of the crops or plants growing in this environment is referred to as $D_{rooting}$ (m), which will be controlled by the plant species and factors such as the depth and character of the soil profile. The acceptable water concentration can be used to define $S_{eutrophic\ trigger}$. A water concentration value of 0.01 mg L^{-1} appears to be conservative relative to the range of data available[11]. While the time-scale of in-field sorption processes, is measured in years, a standard laboratory 8-hour equilibration is recommended here (e.g. method 9J in Rayment and Lyons 2010). This introduces conservatism in the estimation of the proportion of P_{sorbed} that is considered to be $P_{recalcitrant}$. Four example soils (Redding *et al.*, 2006; Table A4.1) are provided, though region specific data is required to apply the Tier 2 method. This data uses the Freundlich form equation, determined for an equilibration period of 8 hours:

Equation A4.4

$$Psorbed = kC^n$$

where the units of are mg kg of soil^{-1}, *k and n* are fitted parameters, and *C* represents the solution concentration (mg L^{-1}). Using an acceptable water concentration of 0.01 mg L^{-1} and applying Equation A4.4:

Equation A4.5

$$Seutrophic\ trigger = k\ 0.01^n$$

[11] https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria

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Appendix 5

Nitrogen and phosphorus losses from feed storage

Feeds are purchased and stored on animal farms for various periods of time, during which substantial losses of dry matter (DM) and nutrients may occur. Losses, also referred to as "feed shrink", can be caused by many factors including delivery weight errors, wind, birds, rodents, tires and tracked feed, silage losses due to anaerobic and aerobic fermentation, heating, and spoilage, mixing errors, scale accuracy, and feed refusals and feed waste at the feed bunk (Brouk, 2009). Depending on the type of feed and storage facility, losses may reach 30 percent or higher of the feed purchased by the farm (Table A5.1). Typically, shrink losses from concentrate feeds are around 10 to 15 percent (as-is basis). Well-managed farms may have 5 percent or less storage losses for their concentrate feedstuffs and less than 10 percent losses from stored forages.

To reduce feed losses, producers should have a good handle on the actual amount of feed delivered to the farm. On large farms, incoming truckloads should be weighed, and feed ingredients sampled and analysed, at least for DM, so accurate feed inventories are maintained. When individual feeds are mixed on the farm, proper mixing protocols must be developed and implemented. Feed intake must be closely monitored and, if forages are fed, forage DM must be analysed weekly and necessary corrections to the animal diet should be made. Expensive feed ingredients (cereal grains, soybean meal, premixes, for example) should be stored in

Table A5.1: Example of typical losses due to shrink and spoilage during bulk storage and handling of selected dairy feedstuffs (percent losses on dry matter basis)

Feed Ingredient	Open, uncovered piles	Covered, three-sided bays	Closed bulk tanks
Alfalfa meal	7-15	5-10	2-5
Alfalfa, chopped	10-20	5-10	-
Bakery waste	8-16	4-7	-
Barley grain, meal	5-10	3-8	2-5
Barley grain, whole	5-8	4-7	2-3
Beet pulp, dried	12-20	5-10	3-5
Bran, wheat	15-28	6-12	2-5
Brewers grain, dry	12-20	5-10	3-5
Brewers grain, wet	15-30	15-30	-
Concentrate supplements	4-5	4-5	-
Cottonseed, whole	10-20	5-15	-
Distillers grains, dry	15-22	7-10	3-6
Distillers grains, wet	15-40	15-40	-
Dry grains, typical	5-8	4-7	2-4
Middlings, wheat	14-22	4-9	3-5
Soybean hulls	12-20	5-10	2-5

Source: adapted from Kertz, 1998

enclosed facilities, such as upright bins, instead of commodity sheds to minimize losses. Except for feeds with low flowability, storage of feed ingredients in upright silos can reduce losses to 1 to 2 percent, compared with 5 to 15 percent in open commodity bays (Kertz, 1998).

HAY AND SILAGE LOSSES

Hay DM losses can occur during all stages of hay-making. Plants continue to respire after cutting, which results in net losses of nutrients (mostly sugars and proteins). Respiration losses are reduced by decreasing forage moisture as quickly as possible. Depending on ambient temperature, respiration losses can be 1 percent (at 50 percent moisture) to 3 percent (at 80 percent moisture) of the forage DM in 12 h, at temperature of 27-28°C (Van Soest, 1994). Mechanical losses during hay harvest or baling can be large, particularly with leafy forages such as alfalfa. Loss of alfalfa leaves also decreases the nutritive value of the hay because leaves have higher protein content than stems. The dryer the hay is at baling - the larger are the leaf losses. Leaves are lost during hay curing on the field (tedding, raking) and during baling. As a result, the relative feeding value of alfalfa can decrease by 30 percent due to extensive leaf losses.

Hay can be successfully baled when moisture is below 20 percent, but losses can increase depending on the type of bale. Hay baled in smaller, rectangular bales, for example, can have moisture up to 20 percent, but hay baled in denser, large round or rectangular bales should have moisture below 18 and even 16 percent because these bales lose less moisture during storage and losses from heating and moulding can be higher. Once baled, hay will continue to lose moisture and DM. Even barn-stored hay will lose 5-10 percent (about 5 percent as DM and the remaining as moisture) of its weight over several months (Shewmaker and Thaemert, 2005). Hay stored outside, on the ground, and without cover may lose up to 15-20 percent of its DM due to respiration, physical losses, and microbial activities. Dry matter losses from bales stored directly on the ground can be as high as 50 percent. Hay quality, specifically protein digestibility, can dramatically deteriorate due to heating, if bale moisture is too high. Some heating will take place even in hay with 15 percent moisture, but protein losses (i.e. undigested protein losses in faeces) will be significant above bale temperatures of around 48-50°C and spontaneous combustion may occur, if hay temperature reaches 70°C.

Silage losses are usually the largest feed storage losses on cattle farms that make silage and can exceed 30 to 40 percent of harvested forage DM. Losses occur at all segments of the production chain: from harvest (field losses), through filling the silo and storage (fermentation losses), to feeding the silage (aerobic fermentation losses). Harvesting the forage too wet or too dry will increase either harvest or fermentation (or both) losses. Typically, well-preserved and managed silage should lose less than 10 percent, and close to 5 percent, of its DM during storage. Extremely poorly-managed silages, for example, silage that is not packed well and not covered, can have 40 percent and even higher DM losses. On most farms, silage losses will likely be around 15 percent or less of DM entering the silo.

It should be noted that silage fermentation losses are primarily carbon losses (as CO_2). P is not lost with fermentation gases and there are little losses of N as ammonia or N_2O. In fact, concentration N and P can increase in fermented silage, compared with the original forage. Nutrients in silage are lost at equal rate with silage effluent when forages are ensiled too wet. Therefore, for accurate estimation of silage N and P losses, it is important that actual forage analyses data is used.

To avoid effluent losses, forages should be ensiled at DM content of ≥25 percent. Typical effluent production is 0-100 L t^{-1} for corn silage (25-30 percent DM), 180-290 L t^{-1} for fresh grass or clover silage (17 to 22 percent DM), with no effluent losses for grasses wilted to >22 percent DM (Jones and Jones, 1995). Various equations have been developed to predict effluent losses from silage. One example is the equation of Bastiman and Altman (1985): $y = 767.0 - 5.34x + 0.00936x^2$; where y is effluent production (L t $silage^{-1}$) and x is DM content of the silage (g kg^{-1}). Concentration of P and N in silage effluent will depend on the type of forage and can vary from 37 to 563 mg L^{-1} soluble reactive P and from 2.8 to 4.9 g L^{-1} total N (Gebrehanna *et al.*, 2014).

If forages are ensiled too dry or are wilted before ensiling and not well packed, N losses in manure may increase due to decreased protein degradability as a result of heating. A good indicator of heating in forages, hay or silage, is acid-detergent fiber-bound crude protein (ADF-CP) or acid-detergent fiber-bound N (ADF-N). When excessive binding of forage protein takes place, manure N losses will increase. Thus, if manure N excretion is calculated using dietary protein digestibility, correction for decreased forage protein digestibility should be applied, as described below. Increased silage temperature may, particularly with legume forages, also increase ammonia formation and potentially N volatilization losses during feedout (Muck and Dickerson, 1988). Below is an example of calculating protein indigestibility based on ADF-CP/crude protein (CP) ratios (Cumberland Valley Analytical Services, Maugansville, MD; methods are available at: http://www.foragelab.com/Resources/Lab-Procedures, accessed February 6, 2017):

Ratio ADF-CP/total CP = ADF-CP/total CP × 100 (units are percent on DM basis)

1. If the ratio is <14, all ADF-CP is considered digestible (adjusted CP = CP, i.e. no adjustment is necessary)
2. If the ratio is >14 but <20, only ADF-CP above 7 percent is considered indigestible (i.e. adjusted CP = CP – {[(Ratio-7)/100] · CP}
3. If the ratio is >20, all ADF-CP is considered indigestible (adjusted CP = CP – ADF-CP)

Another important point to consider when it comes to silage losses is accurate determination of silage DM. It has been suggested that silage is a significant source of volatile organic compounds (VOC; i.e. alcohols, volatile fatty acids, aldehydes; Hafner *et al.*, 2013), which are lost during silage manipulation and feedout. Recent studies, however, have emphasized the importance of correcting DM losses for volatile compounds lost during silage DM determination and have placed silage DM losses at less than 9 percent and as low as 3 percent (Köhler *et al.*, 2013; Robinson *et al.*, 2016). These findings were considered in the recommendations provided below.

CALCULATING FEED LOSSES

Feed losses can be calculated if initial feed weight, current inventory, and amount fed are known. Losses should be calculated on DM basis taking into consideration nutrient concentrations whenever possible. The following inputs are needed: (1) initial feed inventory, (2) current feed inventory, (3a) initial and current DM content of the feed (Tier 1) or (3b) initial and current nutrient concentration (Tier 2 and 3), and (3) amount of feed, as DM, fed to the animals on the farm. For example, feed losses can be calculated as follows:

Tier 1

When actual feed DM losses are not known and cannot be reliably calculated, losses of feed due to shrinkage on the farm can be estimated based on data in Table A5.2 (on DM losses) and Equation A5.1 below. In this approach, possible changes in nutrient concentration in the feed are ignored.

Equation A5.1
(calculating N and P losses; also applicable to any feed nutrient):

$$\text{Losses of N (or P), kg or t} = \text{Dry matter losses, kg or t} \cdot \text{N (or P) concentration in feed, as fraction on DM basis}$$

When feeds are not analysed, N and P concentrations can be taken from country-specific feed composition tables (recommended) or sources such as Feedipedia (http://www.feedipedia.org/), NRC (2001), the U.S. National Animal Nutrition Program (https://nanp-nrsp-9.org/) and others.

Tier 2

The assumption of proportionality of DM loss and nutrient losses will in most cases lead to a likely overestimation of nutrient losses. In the Tier 2 methodology, an estimate of the changed nutrient concentration is taken into consideration based on an estimate for the share of loss-processes that go ahead with the loss of both nutrient and DM:

Equation A5.2
(calculating N and P losses; also applicable to any feed nutrient):

$$\text{Losses of N (or P), kg or t} = \text{Dry matter losses, kg or t} \cdot \text{N (or P) concentration in feed, as fraction on DM basis} \cdot \text{Share of processes with losses of both DM and nutrients}$$

Table A5.2: Suggested storage feed dry matter losses (as percent; Tier 1 and 2 approach)

	Level of Farm Management[1]		
Feed category & storage facility	**Poor**	**Medium**	**High**
Hay[2]	≥25	10-20	≤10
Concentrate feeds	≥15	5-10	≤5
Silages[2]			
Trench or bunker silo	≥30	10-20	≤10
Upright silo	≥15	10-15	≤7
Silage bags[3]	≥30	≤15	≤10
Balage[3]	≥25	≤15	≤10

[1] Examples of *poor level of farm/feed management*: high field losses during hay harvest, including from rain, hay stored on the ground and without cover, high silage fermentation losses due to poor packing and lack of cover, or poor silage face management, concentrate feeds left in the open without cover; Examples of *medium level of farm/feed management*: moderate losses of hay DM during harvest, hay and silage covered but not well-packed, concentrate feeds stored in a bay; Examples of *high level of farm/feed management*: minimal field losses during harvest of hay, silage well-packed, covered with plastic and weighted, use of silage preservatives and silage defacer, concentrate feeds stored in feed bins.

[2] Silage N (and DM) losses may be higher for legume hay or silages compared with whole-crop corn, small-grain, or grass silages.

[3] Silage bags and bales: *poor management* = bales not wrapped in plastic, stored outdoor, bags with high-DM silage not well packed; *medium management* = low density bales or poor packing of bagged silage, poor control of bag integrity; *high management* = plastic wrap for bailage, well-packed bagged silage, control of bag integrity.

Tier 3

On-farm losses of N and P are most accurately estimated when feeds are weighed when entering the farm and when fed to the animals and nutrient composition is monitored by sampling and analysis of representative feed samples (Equations 3).

Equation A5.3
(calculating feed DM loss when feed inventories and feed intake are known):

Feed loss, percent = {[(Initial feed inventory, kg N or P – Current feed inventory, kg N or P) – Feed fed to the animals on the farm, kg N or P] / (Initial feed inventory, kg N or P – Current feed inventory, kg N or P)} · 100.

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Appendix 6

Estimation of the N and P content of animals and animal products

Many nations or regions will have access to tools to estimate nutrient concentrations of animals and animal products, where primary data is unavailable. Table A6.1 represents an example often used in the Unites States of America for entire farm nutrient balance calculations. Such tools may provide a starting point to estimate nutrient concentrations (NC_{EBW} and NC_{AP}). A tier 1 approach would be to utilize simple factors for nutrient concentration such as those used by the Cornell University Whole Farm Nutrient Balance calculator (see Table A6.1) or comparable tools locally available.

Milk Sold: Milk protein reported to the producer as true protein is converted to crude protein by multiplying by 1.075 (Cornell Animal Science Dept. Mimeo 213). The N content of milk crude protein is calculated dividing by 6.38, as follows:

Equation A6.1

$$\text{N (tonnes N yr}^{-1}) = (\text{kg of milk sold}) \cdot \\ (\text{milk true protein (percent)} \cdot 1.075) / 6.38) \cdot 1000$$

Equation A6.2

$$\text{P (tonnes P yr}^{-1}) = (\text{kg of milk sold} \cdot 0.0009) \cdot 1000$$

Literature values are typically reported as crude protein. Crude protein is converted to N dividing by 6.25 for eggs and meat and by 6.38 for milk (FAO, 2003).

Such tools may lack specificity of nutrient concentration by age, animal body, breed, or genetics. To confirm accuracy of these values or refine their estimates, a

Table A6.1: Nutrient composition of livestock (N, P) as percent of live bodyweight and milk used by Cornell University Whole Farm Nutrient Balance calculator

Species	N as % of bodyweight	P as % of bodyweight
Dairy	2.9	0.70
Beef < 454 kg	2.7	0.73
Beef>= 454 kg	2.4	0.65
Pigs < 45.4 kg	2.5	0.56
Pigs => 45.4 kg	2.4	0.47
Poultry	2.8	0.58
Goats	2.4	0.60
Sheep	2.5	0.60
Horses	2.9	0.70

Source: Rasmussen, *et al.*, 2011

regional or international literature review is suggested. The following discussion will share literature review examples of estimates of animal product concentrations that may further refine these estimates.

BEEF AND DAIRY CATTLE SYSTEMS

A partial review of literature estimates of beef and dairy nutrient concentrations (NC_{EBW} and NC_{AP}) is shared in Table A6.2. The literature contains multiple research studies defining whole body N concentration (see Table A6.2). A less extensive database exists for whole body P concentration. Ellenberger *et al.* (1950) is a classic reference that continues to be quoted for P retention in dairy cattle. Because comparable quality references have not been identified for beef animals, this reference for P content would be our recommended resource for beef cattle.

For beef, the N estimates used by the Cornell University Whole Farm Nutrient Balance calculator appear to be slightly high compared to the literature values in Table A6.2. This review would also suggest some need for adjustments in the values used by Cornell University's Whole Farm Nutrient Balance calculator for P (lower for cattle under 454 kg except for calves, higher for cattle over 454 kg).

Some data is reported based on live body weight (LW) while most data is estimated based upon empty body weight (EBW). Data from several references in Table A6.2 would suggest that EBW is approximately 90 percent of live weight for beef animals of 500 kg or larger and 85 percent for animals of 300 to 400 kg. National

Table A6.2: N and P concentrations of beef cattle (percent of EBW) based upon sample literature citations

		Average	Min.	Max.	Reference
Calves at birth	N	-	-	-	
	P	0.76	-	-	4
Calves	N	3.3	3.2	3.4	3
	P	0.78	-	-	4
	EBW LW^{-1}	95%	93%	96%	3
200 to 500 kg cattle	N	2.6	2.3	2.9	1, 2, 5, 6
	P	0.8	0.78	0.834	4, 6
Cattle over 500 kg	N	2.5	2.4	2.7	2
	P	0.9	0.86	0.93	4

[1] Carstens *et al.* (1991); [2] Coleman *et al.* (1993); [3] Diaz *et al.* (2001); [4] Ellenberger *et al.* (1950); [5] Ferrell *et al.* (1976); [6] Maarcondes *et al.* (2012)

Table A6.3: Nutrient content of fluid milk as reported by the USDA Food Composition Databases (USDA, 2015)

Description	Protein (g 100 g $milk^{-1}$)	P (mg 100 g $milk^{-1}$)	N (%)[1]	P (%)
Milk, goat, fluid, with added vitamin D	3.56	111	0.558%	0.111%
Milk, Indian buffalo, fluid	3.75	117	0.588%	0.117%
Milk, dairy cow, fluid, 3.7% milkfat	3.28	93	0.514%	0.093%
Milk, sheep, fluid	5.98	158	0.937%	0.158%

[1] Conversion of 6.38 used to estimate N based upon reported protein content.

Academies (2016) estimates empty body weight to be 89.1 percent of shrunk body weight or 85.5 percent of full body weight for finished beef cattle.

Nutrient concentration in milk is reported by a variety of food nutrient content databases. One example is the U.S. Department of Agriculture National Nutrient Database for Standard Reference for which whole milk nutrient concentrations are illustrated in Table A6.3.

SHEEP

Sheep LW has been estimated to have N and P concentrations of 2.5 percent and 0.74 percent, respectively (e.g. from a VERA Swedish Board of Agriculture programme). Corresponding N and P concentrations in shorn greasy wool are 9.1-11.2 percent (depending on level of plant and soil contamination; 16 percent in clean scoured wool) and 0.01 percent, respectively (Wiedemann *et al.*, 2015).

PORK PRODUCTION SYSTEMS

The literature contains multiple research studies defining whole body N concentration (see Table A6.4). A less extensive database exists for whole body P concentration. Mudd *et al.* (1969) reported LW P concentrations of 5.54 mg/kg for 23 kg pigs and 5.52 percent for 41 kg pigs. These values are close to estimates assembled by Fernandez *et al.* (1999) illustrated in Table A6.4. Without completing a more extensive literature review, the values used by the Cornell University Whole Farm Nutrient Balance calculator would appear to be reasonable estimates of nutrient concentration for use in Equation 6 and 7, Section 4.3.3.1.

Most data is reported on the basis of EBW, which represents approximately 95 percent of whole body weight.

Table A6.4: The content of N and P in the body of piglets and in the body weight gain of sows and weaners

		Content	Min.	Max.	Reference
Sows[a], g kg BW-gain^{-1}	N	25	20	30	3, 4, 5, 7, 8[bd]
	P	5	4.7	5.1	4[b]
Piglets (7.5 kg), g kg BW^{-1}	N	24	23	24	1, 2, 4, 6[d]
	P	5	4.7	5.0	1, 2, 4, 6[d]
Weaners (7.5-30 kg BW), g kg EBW-gain^{-1}	N	29	-	-	9[c]
	P	5.7	-	-	9[c]
Weaners (21.4 kg LW / 19.0 kg EBW), g kg LW^{-1}	N	24.9			9[c]
	P	5.1			9[c]
Growing Pigs (88.2 kg LW / 83.3 kg EBW), g kg LW^{-1}	N	27.2			9[c]
	P	5.5			9[c]

[a] BW-gain was estimated based on experimentally determined average weight gain of sows over several parities and added the contribution under practical conditions of boars, replacement gilts and dead piglets (<2-kg LW) to 60 kg/sow/yr.

[b] Combined with unpublished Danish results.

[c] Calculated based on the body content of piglets and the body content of weaners. EBW, empty body weight.

[d] Becker *et al.* (1979); Berge and Indrebo (1954); De Wilde (1980); Everts and Dekker (1991); Everts and Dekker (1994); Nielsen (1973); Walach-Janial *et al.* (1986); Whittemore and Yang (1989); Fernandez, *et al.* (1991)

Source: Fernandez *et al.*, 1991

Table A6.5: Nutrient content of eggs as reported by the USDA Food Composition Databases (USDA, 2015) with an adjustment for shell nutrient content

	Egg Weight (grams)		N & P Concentration of Whole Egg (minus shell)[1]		Shell Nutrient Content from two references			Estimated Nutrient Content for Combined Whole Egg and Shell[2]	
Egg Description	**Whole Egg only[1]**	**Whole Egg and Shell[2]**	**N ($g\ egg^{-1}$)**	**P ($mg\ egg^{-1}$)**	**% P [3]**	**% N [4]**	**% P [4]**	**N Content ($g\ egg^{-1}$)**	**P Content ($mg\ egg^{-1}$)**
Duck	70	78	1.44	154	0.085%	0.40%	0.10%	1.47	161
Goose	144	160	3.20	300	0.085%	0.40%	0.10%	3.26	315
Quail	9	10	0.19	20	0.085%	0.40%	0.10%	0.19	21
Turkey	79	88	1.73	134	0.085%	0.40%	0.10%	1.76	142
Chicken	50	56	1.00	99	0.085%	0.40%	0.10%	1.03	104

[1] USDA, 2016.

[2] Assumes weight of shell is 10 percent of weight of whole egg plus shell. Based upon literature values for chicken eggs only.

[3] Atteh and Leeson (1983) reported eggshell P content ranging from 0.08 to 0.09 percent for 7 dietary treatments (percent of eggshell weight).

[4] Schaafsma *et al.* (2000) reported N and P composition of eggshell powder ranging from 3.90 to 4.02 mg N/g of powder and 0.2 to 1.9 mg P/g powder for four genetic strains of chicken layers.

Note: Column 3 and columns 9/10 provide potential values for the mass of eggs produced and the nutrient concentration of eggs

POULTRY – EGG PRODUCTION

The nutrient output of layer facilities include both eggs and "spent" hens (hens that are ready for slaughter when no longer producing eggs economically). A discussion of nutrient flows represented by bird body mass is presented in the "Poultry – Meat Bird Production" and provides an approximation of nutrient flows as spent hens. In addition, layer facilities will receive pullets that should be characterized as a nutrient inflow. This nutrient flow can be estimated following procedures discussed in the meat bird production section.

Estimating the nutrient out-flow in eggs can use Equation 8, section 4.3.3.2 and requires estimation of the nutrient concentrations of eggs and the mass of the eggs produced. Nutrient concentrations for whole fresh eggs are commonly reported in food nutrient concentration databases such as the USDA Food Composition Database (USDA, 2015). These databases report nutrient concentrations typically as crude protein (adjustable to N by dividing by 6.25) and P for the fluid part of the egg. A literature review suggests that egg shells represent about 8 to 11 percent of the total eggs weight and contain both N and P. An adjustment for N and P in the egg shell, summarized in Table A6.5, suggests that a 3 to 5 percent increase in N and P content per egg and an increase in egg weight of roughly 10 percent. Adjusting food database values for egg shell weight and nutrient content would result in a 13 to 15 percent greater (and more accurate) estimate of nutrient output.

POULTRY – MEAT BIRD PRODUCTION

The literature contains multiple research studies defining whole body N concentration, but fewer studies with P concentration (Table A6.6).

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Table A6.6: Example of N and P concentrations of poultry based on sample literature citations

Source	Live Weight (kg)	Crude Protein (% of LW)	N (% of LW)	P (% of LW)	Notes
Aletor *et al.*, 2000	2.19	18.8%	3.01%		6-week-old broiler chickens
Bregendahl *et al.*, 2002	0.76		2.60%		Three experiments involving chicks harvested at 3 weeks.
Bregendahl *et al.*, 2002	0.85		2.59%		
Bregendahl *et al.*, 2002	0.72		2.57%		
Donaldson *et al.*, 1956	0.44	19.4%	3.10%		16 diet trials of crude protein and energy
Olukosi *et al.*, 2008 a	0.13	13.9%	2.22%	0.38%	5 diet treatments, 7-day old broiler chicks
Olukosi *et al.*, 2008 a	0.34	18.2%	2.91%	0.41%	5 diet treatments, 14-day old broiler chicks
Olukosi *et al.*, 2008 a	0.69	20.2%	3.23%	0.39%	5 diet treatments, 21-day old broiler chicks
Olukosi *et al.*, 2008 b	0.51			0.45%	5 diet treatments, 21-day old broiler chicks
Mavromichalis *et al.*, 2000	0.42	19.30%	3.09%		22 day-old chicks fed standard diet
Hemme *et al.*, 2005	2.01			0.44%	2 trials of broiler chicks harvested at 36 days

LW: Live weight is typically measured at end of trial after 24 hour fasting period.

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Appendix 7

Approach to allocate upstream livestock emissions to manure and livestock products*

DIFFERENTIATING MANURE APPLICATION AND DEPOSITION BETWEEN "PRODUCT" AND "WASTE"

Losses of nutrients from the soil-plant-atmosphere continuum are unavoidable to an extent that is dependent on environmental conditions, available technologies and farm practices. The magnitude of the losses – and thus the response of crops to the incremental addition of fertilizers – depends on the absolute fertilization level. This is usually expressed in crop-growth curves showing the yield that is obtained at a certain fertilization level. Such curves often have an exponential shape with physical optimum corresponding to the fertilizer application level which gives the maximum obtainable yield (under other given environmental and management conditions).

A possible formalization of a crop nutrient response curve is developed in Godard *et al.* (2008) but others are possible.

Equation A.7.1

$$Y = Y_{mx} - (Y_{mx} - Y_{mn}) \cdot exp\{-x \cdot f\}$$

The first derivative of the curve gives the nutrient uptake efficiency [t harvested kg nutrient^{-1}] or fertilizer recovery F_R [kg nutrient harvested kg nutrient applied^{-1}]. It is calculated using the nutrient content N_Y [kg nutrient in harvested product kg harvested product^{-1}].

Equation A.7.2

$$\frac{\partial Y}{\partial f} = (Y_{mx} - Y_{mn}) \cdot x \cdot exp\{-x \cdot f\}$$

Equation A.7.3

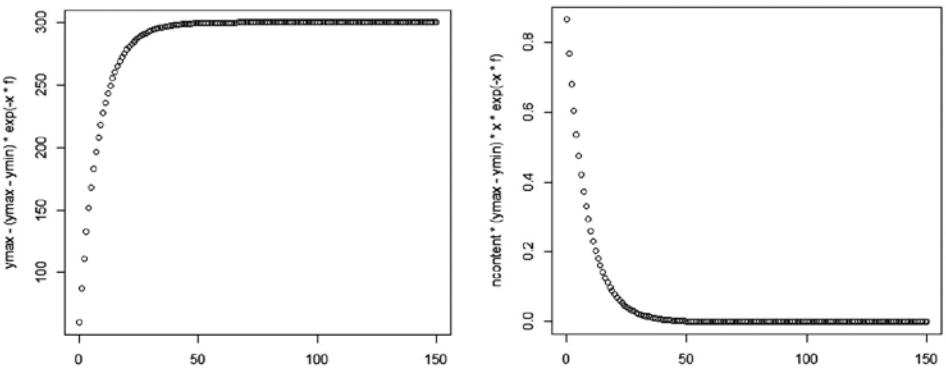
$$F_R = N_Y \cdot (Y_{mx} - Y_{mn}) \cdot x \cdot exp\{-x \cdot f\}$$

Figure A.7.1 shows an example of nutrient response curve with x=0.04 (left) and first derivative (nutrient use efficiency, right). Unit y-axis: kg nutrient harvested ha^{-1} yr^{-1} (left), kg nutrient harvested/kg nutrient applied (right). Unit x-axis: kg nutrient ha^{-1} yr^{-1}.

The economic optimum gives the fertilization level at which the added value of harvested crop equals the cost of the additional fertilizer ∂C_f (Godard *et al.,* 2008),

* A scientific paper describing the approach proposed has been prepared and is currently under review at the Journal on Environmental Management. If accepted, it will be published in a special issue of the journal "Nitrogen Solutions".

Figure A.7.1
Example of nutrient response curve with x=0.04 (left) and first derivative (nutrient use efficiency, right). Unit y-axis: kg nutrient harvested ha^{-1} yr^{-1} (left), kg nutrient harvested/kg nutrient applied (right). Unit x-axis: kg nutrient ha^{-1} yr^{-1}



including other costs linked to the production level $C_Y \partial Y$. F. For the farmer an increase in fertilization up to the economic optimum is rational.

The yield increment is obtained from F_R (kg nutrient in product kg nutrients in fertilizer^{-1}) at the fertilizer level and the nutrient content in the product N_Y (kg nutrients kg product^{-1})

Equation A.7.4

$$\partial Y = \frac{\partial f \cdot F_{R,f}}{N_Y} = (Y_{mx} - Y_{mn}) \cdot x \cdot exp\{-x \cdot f\} \, \partial f$$

The economic optimum is reached when the additional income $\partial I = P_Y \cdot \partial Y$ with P_Y being the revenue for the product [Euro/kg product] equals the additional cost of fertilizer equivalent $\partial C_f = \partial f \cdot C_f$, with C_f being the cost per unit of fertilizer [Euro/kg nutrients], plus any other variable costs that are linked proportionally to the yield increment $C_Y \cdot \partial Y$

Equation A.7.5

$$P_Y \cdot \partial Y \,|_{econopt} = (\partial f \cdot C_f + \partial Y \cdot C_Y)_{econopt}$$

Thus, at the economic optimum, the following holds:

Equation A.7.6

$$(Y_{mx} - Y_{mn}) \cdot x \cdot exp\{-x \cdot f_{econopt}\} \cdot (P_Y - C_Y) = C_f$$

Equation A.7.7

$$f_{econopt} = -\frac{1}{x} \cdot ln\left\{\frac{C_f}{(P_Y - C_Y) \cdot (Y_{mx} - Y_{mn}) \cdot x}\right\}$$

Equation A.7.8

$$F_{R,econopt} = \frac{N_Y \cdot C_f}{(P_Y - C_Y)}$$

Manure added to a field to the point of the economic optimum in mineral fertilizer equivalent would be replaced by synthetic fertilizers if the manure were not available. In this case, the value of the manure equals the value of the equivalent amount of synthetic fertilizer and the manure is to be regarded as co-product.

Equation A.7.9

$$M_{full} = min\ (Q_{m,} \frac{f_{econopt}}{f_{eq}})$$

With the fertilizer equivalent f_{eq} being calculated from the quantity of mineral fertilizer nutrient Q_f and manure nutrient Q_m application as explained below.

Equation A.7.10

$$f_{eq} = \frac{Q_f}{Q_m}$$

If the farmer applies manure at a level that is beyond the economic optimum but below the physical optimum, the farmer generates value only because the manure is freely available (or cheaper than mineral fertilizer) and external costs caused by the (high) losses are not internalized. This share of manure is to be regarded also as co-product but using a lower value corresponding to half FR at economic optimum. The fertilizer equivalent value is the same as used below the economic optimum at the economic optimum point, and zero at the physical optimum, as no further yield increase results from the application. This default method suggests using the average equivalent value in this range.

Equation A.7.11

$$M_{half} = max\ (0,min\ (Q_{m,} f_{mx} / f_{eq}) - M_{full})$$

Here we define f_m as the physical optimum fertilizer application rate that is required for achieving a yield of 95 percent of the maximum yield. [Note: the value of 95 percent is arbitrary – also a higher share of e.g. 99 percent could be used]

Any application of nutrients in manure beyond f_{mx} is to be considered as waste (M_{waste}).

CALCULATING FERTILIZER EQUIVALENTS

For a farmer, the value of manure can be obtained by the quantity of mineral fertilizer she/he would purchase in case the manure was not available. A good approximation to this quantity is the amount of mineral fertilizer that would be needed to provide the same amount of nutrients for plant uptake. Thus, the quantities of mineral fertilizer Q_f and manure Q_m multiplied by their nutrient use efficiencies (NUE_f and NUE_m, respectively) must be identical.

Equation A.7.12

$$Q_f \cdot NUE_f = Q_m \cdot NUE_m$$

Note the difference between the NUE used here and the fertilizer recovery F_R used above. NUE is the share of nutrient input taken up by the plant as a whole, including crop residues.

Equation A.7.13

$$NUE = N_{output} / N_{input}$$

Whereby N_{output} refers to total nutrients in plant biomass plus nutrients stored in soils (soil stock changes, N_{ssc}). The difference N_{input} - $N_{surplus}$ gives the which equals the sum of all losses to atmosphere and hydrosphere. The nutrient balance equation is

Equation A.7.14

$$N_{input} = N_{plant} + N_{ssc} + N_{surplus}$$

F_R, on the other hand, refers to nutrients in harvested material only, therefore

Equation A.7.15

$$NUE = F_R + ((N_{cres} + N_{ssc}) / N_{input})$$

Assuming equal distribution of nutrients across crop compartments, the only difference in the N output is the soil stock change; for the N inputs only, N in manure or mineral fertilizer is different. Thus Equation A7.15 becomes

Equation A.7.16

$$f_{eq} = Q_f / Q_m = NUE_m / NUE_f = NUE_m \cdot N_{input,f} / N_{output,f}$$

Equation A.7.17

$$f_{eq} = NUE_m \cdot (N_{input,m} + \Delta N_{ssc,m} + \Delta N_{surplus,m}) / (N_{output,m} + \Delta N_{ssc,m})$$

Equation A.7.18

$$f_{eq} = NUE_m \cdot (1 + (2 \cdot \Delta N_{surplus,m} / (N_{output,m} + \Delta N_{ssc,m})))$$

With

Equation A.7.19

$$N_{input,f} = N_{input,m} + N_{ssc,f} - N_{ssc.m} + N_{surplus,f} - N_{surplus,m}$$

Hence the fertilizer equivalent can be calculated based on the nutrient use efficiency for the total nutrient input level if manure is used, the yield at this point, and the differences in soil stock changes and nutrient losses if mineral fertilizer were used in a quantity that yields the same total nutrient plant uptake.

SUMMARY

Assuming a farmer applies X kg ha^{-1} of mineral fertilizer and Y kg ha^{-1} of manure. Considering N and P, the crop receives $X \cdot C_{N,x} + Y \cdot C_{N,y}$ of N and $X \cdot C_{P,x} + Y \cdot C_{P,y}$ of P, with $C_{nut,fer}$ as the nutrient content in the fertilizers.

a) Based on Equation A7.18, the fertilizer equivalents for N and P can be calculated, using the N and P models to quantify soil stock changes and loss flows: and

b) The economic optimum and is determined using Equation A7.7 or any analogue equation, depending on the crop nutrient response curves that are being used.

c) Other sources of nutrients might be present which are independent from fertilizer addition, such as atmospheric deposition, biological N-fixation, or decomposing crop residues, need to be accounted for. Equation A7.9 quantifying manure as co-product with full-fertilizer equivalents changes thus to:

Equation A.7.20

$$M_{full,nut} = min\ \{M_{nut}' \cdot (f_{econopt,nut} - f_{other}) / f_{eq}\ \}$$

d) The value of the nutrient in manure to be used to allocate emissions of livestock supply chain is obtained from using fertilizer price and the difference between total manure applied and the manure that is accounted fully as fertilizer equivalents is accounted for with half fertilizer price.

Equation A.7.21

$$P_{nut} = (M_{full,nut} + {}^{1}/_{2}\ M_{half,nut)} \cdot P_{min,nut}$$

e) Total manure value is the sum of the value for the individual nutrients in manure, using separate crop response curves under the assumption that only one of nutrients is limiting at a time.

Equation A.7.22

$$P_{manure} = P_N + P_P$$

f) For sustainable agriculture it is assumed that available manure is used as much as possible. If this minimum share plus the amount of nutrient in manure is equal or more than the economic optimum, then additional mineral fertilizer is assumed to be applied unsustainably and has no impact on the allocation problem of livestock prechain emissions between products and manure.

ILLUSTRATIVE EXAMPLE

We refer to the example given in the LEAP guidelines on poultry supply chains (FAO, 2016, Appendix 3).

Three co-products were considered for a laying operation with 1000 layers, whereof 350 were sent to slaughter annually: eggs, poultry meat, and manure.

In contrast we do not consider that the manure is sold to a nearby power plant for electricity production but is used as fertilizer on a cereal field.

Appendix 3 of the poultry guidelines calculates allocation of burden to eggs, meat and manure using the partitioning of the metabolizable energy (ME) into ME requirements for maintenance, growth, and production. The following information is obtained for the example:

- The allocation results in 46.5 percent for eggs, 27.4 percent for meat and 26.1 percent for manure.
- This gives an allocation between eggs and meat of only 63 percent for eggs and 37 percent for meat.
- The average spent hen weight is 3.3 kg
- The eggs mass produced in 100 weeks is 23.3 kg.

The economic allocation requires farm gate prices of cereals, mineral fertilizers, eggs, and poultry meat. Table A7.1 gives an overview of prices available in the CAPRI database (for the year 2008) for EU-28. All data is in Euro per t of product. Other data required to obtain the value of manure versus the value of eggs and poultry meat are the N and P contents in each co-product, the edible fraction of the poultry body mass as given in Table A7-2.

Using the above values, the calculation allocates 6 percent of emissions to manure, and 94 percent to eggs and meat. The allocation takes all the value that manure gives to the farmer for crop production into consideration, in the example this is the sum of N and P, but other values could be considered as well (carbon, soil structural benefit), as long as the benefit can be expressed as monetary value. The allocation among eggs and meat vary depending on whether the physical allocation factors developed in the LEAP guidelines on poultry supply chains example are used, or all allocation factors calculated based on economic allocation.

Thus, where manure is considered as a co-product, 6 percent of the upstream burden is allocated to the crop it is applied to (when it is applied to land). For the check if the application of manure is to be considered as waste, additional information is required:

- The quantity of manure-nutrients.
- The sources of other inputs to the land including atmospheric deposition, biological fixation, and mineralisation of soil organic matter or use of inputs from previous years (e.g. crop residues), but NOT the input of mineral fertilizers.
- The maximum amount of nutrients that should be applied at the economic and physical optima.

Table A7.1: Producer prices of cereals, mineral fertilizer (N, P, K), eggs and poultry meat in EU-28. *Unit: Euro t product^{-1}*

Soft wheat	Mineral fertilizer: N	Mineral fertilizer: P_2O_5	Mineral fertilizer: K_2O	Eggs	Poultry meat
150	1037	1452	641	1182	1379

Source: CAPRI database for base year 2008, revision 228, July 2015

Table A7.2: Summary table for the calculation of the value of the co-products for the illustrative examples for eggs, poultry meat and manure

Item	Value	N	P	Unit	Note
a) Eggs					
Weight produced	23.3			kg	
Nutrient content		0.018	0.002	kg kg egg^{-1}	Appendix 6, Table A6.5, considering whole egg incl. shell
Nutrient in egg		0.43	0.04	kg	
Price	1182.0			Euro t^{-1}	CAPRI
Value	**27.5**			**Euro**	
b) Poultry meat					
Weight	3.3			kg	
Carcass fraction	0.6				After Ramirez, 2012
Nutrient content		0.028	0.004	kg kg body mass^{-1}	Appendix 6, Table A6.5, average of reported values
Nutrient in body mass		0.09	0.01	kg	
Price	1379.0			Euro t^{-1}	
Value	**2.6**			**Euro**	
c) Manure					
Weight	12.8			kg manure	
Total Nutrient produced		2.56	2.29	kg	
Total Nutrient in manure		2.04	2.24	kg	
Nutrient content		0.159	0.174	kg kg manure^{-1}	
Fertilizer equivalent		44%	100%		Assuming loss of N in MMS of 50% (based on values indicated in IPCC 2006) and a higher volatilization rate upon application of 20% of manure versus 10% for mineral fertilizer. 100% fertilizer equivalent assumed for P.
Fertilizer price		1037	409	Euro t^{-1}	
Manure value		**0.9**	**0.9**	**Euro**	

Table A7.3: Allocation factors of the poultry system in the example over eggs, poultry meat and manure based on economic allocation between manure and food product and physical allocation amongst the food products (column Allocation mixed) or overall economic allocation (column Allocation economic)

	Value [Euro]	Allocation bio-physical	Allocation mixed	Allocation economic
Eggs	27.5	0.63	0.59	0.86
Poultry meat	2.6	0.37	0.35	0.08
Manure	1.9	0	0.06	0.06

Table A7.4: Soft wheat production in EU-28: area, production, yield and nutrient application with mineral fertilizers and manure

Area	Production	Yield
1000 ha yr^{-1}	1000 t yr^{-1}	kg ha^{-1} yr^{-1}
23028	132548	5756

N uptake by crop	N in mineral fertilizers	N in manure applied	N in crop res.+ atm. dep	P_2O_5 uptake by crop	P_2O_5 in mineral fertilizers	P_2O_5 in manure applied	P_2O_5 in crop res.+ atm. dep
kg N ha^{-1} yr^{-1}				kg P_2O_5 ha^{-1} yr^{-1}			
138	**125**	**26**	**39**	**60**	**20**	**24**	**24**

Source: CAPRI database for base year 2008, revision 228, July 2015

Table A7.5: Allocation factors of the poultry system in the example over eggs, poultry meat and manure for Cyprus. The value of manure considers only a share of the applied P as N is applied in excess of crop needs

Cyprus	Value (Euro)	Allocation ME	Allocation mixed	Allocation economic
Eggs	27.5	63%	0.62	0.86
Poultry meat	2.6	37%	0.36	0.08
Manure	0.7		0.02	0.02

Table A7.4 gives an overview of soft wheat production in EU-28 from the CAPRI database. On the average, the sum of N in crop residues, atmospheric deposition and manure is 65 kg N ha^{-1} yr^{-1} for a crop uptake of 138 kg N ha^{-1} yr^{-1}. Most of the N-input comes from the application of mineral fertilizer. Therefore, manure is not applied in excess of the economic optimum and its value can fully be considered for the allocation of upstream burden to soft wheat production. Application of P is 48 kg P_2O_5 ha^{-1} yr^{-1} with about 80 percent of retention in the crop and the same reasoning applies. No data for N or P from soil stock resources are available.

The situation is different if only looking at the case of Cyprus (see Table A7.5): here, N input from crop residues and atmospheric deposition is already larger than the uptake in crops and thus manure application can be assumed to be completely in excess of crop needs. This is even though crop yield is low and likely below its potential optimum, however, the data suggests that there are other limiting factors than nutrients. For P, P in manure is about 17 percent above crop uptake (assumed physical optimum). Assuming an economic optimum for P fertilizers at 20 kg P_2O_5 ha^{-1} yr^{-1} the share of applied fertilizer equivalent to use is the sum of the application until economic optimum (P1, full fertilizer equivalent value) plus half the fertilizer value applied between economic and physical optimum (P2): P1=20/33=0.61; P2=0.5 · (28-20)/33=0.12. P1+P2=0.73. As a result, only 2 percent of the upstream burden is allocated to manure and thus to soft wheat, while 98 percent of the burden is distributed between eggs and poultry meat (see Table A7.5).

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LIST OF SYMBOLS

C_f	Cost per unit of nutrient (Euro [kg nutrient]$^{-1}$)
C_Y	Variable costs for crop production that is proportional to the yield (e.g. drying) (Euro kg harvest ha^{-1})
F_R	Fertilizer recovery rate (kg nutrient harvested [kg nutrient applied]$^{-1}$)
f_{mx}	Physical optimum fertilizer application rate with which a yield of 95 percent of the maximum yield is achieved (kg nutrients ha^{-1} yr^{-1})
$f_{econopt}$	Economic optimum fertilizer application rate (kg nutrients ha^{-1} yr^{-1})
f_{eq}	Mineral fertilizer equivalent factor (kg nutrient in mineral fertilizer [kg nutrient in manure]$^{-1}$)
I	Revenue from selling the crop (Euro kg harvest^{-1})
M_{full}	Amount of manure nutrient applied with full fertilizer equivalent value
M_{half}	Amount of manure nutrient applied with half fertilizer equivalent value
M_{waste}	Amount of manure considered as waste
N_{input}	Total nutrient input (kg nutrients ha^{-1} yr^{-1})
N_{plant}	Uptake of nutrients into total plant biomass (kg nutrients ha^{-1} yr^{-1})
N_{ssc}	Nutrient soil stock changes (kg nutrients ha^{-1} yr^{-1})
$N_{surplus}$	Nutrient losses to the environment (kg nutrients ha^{-1} yr^{-1})
N_{cres}	Nutrient uptake into crop residues (kg nutrients ha^{-1} yr^{-1})
N_Y	Nutrient content (kg nutrient in harvested product kg harvested product^{-1})
NUE_f	Nutrient use efficiency for mineral fertilizer application (kg nutrient in useful outputs [kg total nutrient input]$^{-1}$)
NUE_m	Nutrient use efficiency for manure application (kg nutrient in useful outputs [kg total nutrient input]$^{-1}$)
P_Y	Price of crop at farm level (Euro kg harvest^{-1})
Q_f	Application of mineral fertilizer (kg nutrients ha^{-1} yr^{-1})
Qm	Application of mineral manure (kg nutrients ha^{-1} yr^{-1})
x	Model parameter determining the curvature of the crop response curve
Y	Crop yield (kg biomass harvested ha^{-1} yr^{-1})
Y_{mx}	Maximum crop yield under no nutrient limitations (kg biomass harvested ha^{-1} yr^{-1})
Y_{mn}	Minimum crop yield without application of nutrient(s) (kg biomass harvested ha^{-1} yr^{-1})Uptake of nutrients of the crops stems for nutrient applications of previous years or from mineralisation soil organic matter or soil bedrock

Appendix 8

Excreta deposition and spatial variability, source/site factors affecting N and P loss, and index methods for estimating nutrient losses

HETEROGENEOUS EXCRETA DISTRIBUTION

Excreta nutrient deposition by grazing animals is primarily a function of nutrient intake in consumed feed, the proportion retained in animal products, where animals spend time, and the density of animals. In grazed systems, the excreta are often heterogeneously distributed across the farm landscape (Gourley *et al.*, 2015; Fu *et al.*, 2010). Collected excreta management was described in the Housed livestock section.

In improved grazing-based operations, farmers manage animals to utilize forage production from pastures, and although they may also purchase additional feed for their livestock, they generally have less control of animal diets, with highly variable feed quality and nutrient content of excreta. For example, Aarons and Gourley (2015) found that dairy cows grazing pastures with markedly different P contents (ranging from 0.15 – 0.50 percent P), had corresponding P concentrations in dung ranging from 0.37 to 1.27 percent.

In extensive systems, the grazing activity and therefore the pattern of excreta depositions, mainly depend on the water resources. This is also influenced by other factors such as land slopes, the heterogeneity of vegetation, and the seasonal variation in the availability and quality of pastures.

The N and P loads from grazing animal dung and urine deposition may be high. For example, the deposition of a single dairy cow urine patch can apply the equivalent of between 500 - 1200 kg N ha^{-1} (Rotz *et al.*, 2005). A summary of research on rates of P deposition in dung patches gave averages of 35 and 280 kg P ha^{-1} equivalent for sheep and cattle respectively (Haynes and Williams 1993).

Within a grazing-based farm, areas which receive animal excreta can be divided into four types: (i) areas where animals are highly managed, such as dairy shed, yards and feed pads (excreta is typically collected from these areas), (ii) areas where animals are forced to be in high densities, such as laneways, feeding areas, and holding areas (most excreta is typically uncollected), (iii) areas where animals choose (or are encouraged) to be in high densities, such as stock camps, shade and wind protection, gateways, watering points, feed and mineral supply (excreta here is typically uncollected), and (iv) areas where animals are generally in low densities such as when grazing (excreta is uncollected) and where nutrient deposition will be spatially and temporally highly variable.

In pig grazing systems the main cause of variation in N and P concentrations in soils is the behaviour of pigs. While pigs deposit urine mainly near their rest areas, the dung deposition is correlated to grazing activities (Blumetto *et al.*, 2012).

The accumulation of excreta nutrients in specific areas within the farm, above agronomic requirements has the potential to disproportionately contribute to nutrient loss.

Accurately determining the amounts and efficiencies of excreta collection and nutrient recycling through excreta on grazing operations is generally estimated based on the relative amount of time animals spend in various farm locations and farmer collection practices. This requires the following information:

i. excreta N and P (g N and P day^{-1}),
ii. where the excreta nutrients were excreted (i.e. barns, barn yards, feed bunks, feed pads, milking parlour, holding paddocks, laneways, and grazed pastures),
iii. the size of each particular area,
iv. the number of animals that were present in each area,
v. the proportion of each day that animals spent in each area,
vi. the proportion of excreta collected from these areas,
vii. how excreta were collected, and
viii. where and how collected excreta was stored.

P AND N LOSS ASSESSMENT

While the resources, time and labour required for directly measuring nutrient losses in field-based studies can be high, the use of mechanistic and empirical models to predict nutrient losses from grazing-based animal production systems are also complex and time consuming to parametrize and validate. Therefore, a widely adopted approach has been to develop indices that assist in predicting the risk of nutrient loss from a field or part of the landscape (Sharpley *et al.*, 2003).

The risk of nutrient loss is the combination of the likelihood and magnitude of loss, as influenced by climatic conditions, landscape features, and land management. Nutrient loss indices are generally based on identifying key sources of nutrients and factors involved in transport and delivery to receiving waters. Where a high likelihood of nutrient transport and delivery coincides with a significant nutrient source, there is an increased risk of nutrient loss (Figure A8.1). Most work developing nutrient loss and environmental risk indices has been concerned with P.

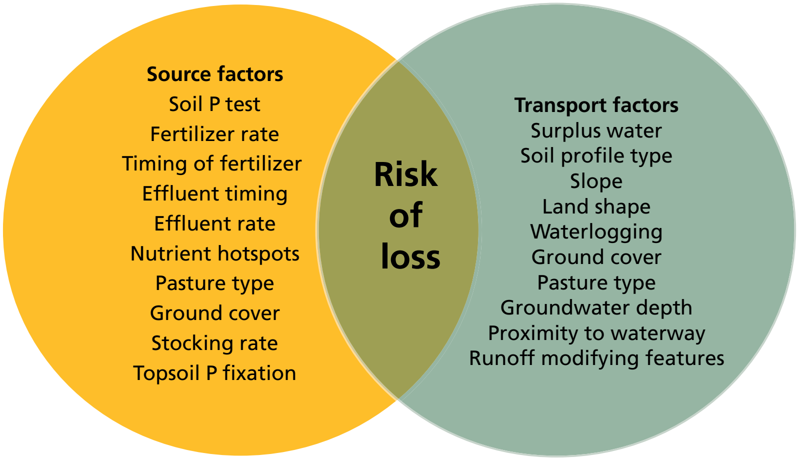
Because the potential for nutrient loss depends on a combination of characteristics specific to each paddock or land management unit, the appropriate management for each paddock can vary. For example, paddocks undergoing similar soil fertility tests but different drainage characteristics, slope, pasture type, or management will have different risks of nutrient loss. Nutrient loss indices can therefore help identify the risks of nutrient loss on different parts of farms, explain why these risks occur, and explore nutrient management options which can minimize nutrient losses.

P INDEX METHODS

In Pennsylvania, the United States of America, P index source indicators used are soil test P, fertilizer application rates and methods, and manure application rates, methods and P source coefficients. The transport indicators used are erosion, surface runoff potential, subsurface drainage, distance to a body of water and evaluation of management practices (Sharpley *et al.*, 2003).

In the United States of America and other countries like Uruguay, erosion is commonly estimated using The Universal Soil Loss Equation (USLE) or the Revised

Figure A8.1
A diagrammatic representation of factors influencing the source, transport and risk of loss of nutrients



Source: Gourley *et al.*, 2007

Universal Soil Loss Equation (RUSLE), which is determined by six factors to predict the long-term average annual soil loss (A). The factors are rainfall erosivity (R), soil erodibility (K), topography (L and S) and pasture/cropping management (C).

Equation A8.1

$$A= R \cdot K \cdot L \cdot S \cdot C \cdot P$$

For Europe, where many countries have developed national erosion mapping systems, it is more appropriate to use the maps instead of USLE/RUSLE equation. Some example sources are ADAS erosion monitoring project and NSRI erosion risk map (Heathwaite *et al.*, 2003).

Potential runoff can be estimated based on the USDA curve number method which is an efficient method for determining the approximate amount of direct runoff (Q) from a rainfall event (P) in a particular area. The equations consist on the following factors: area's hydrologic soil group, land use, treatment and hydrologic condition.

Equation A8.2

$$Q= (P- I_a)^2 / (P - I_a + S)$$

Where; Q is runoff (L)
P is rainfall (L)
S is potential maximum soil moisture retention after runoff begins (L)
I_a is the initial abstraction (L)

Runoff can also be estimated using soil hydrological classifications such as HOST (Heathwaite *et al.*, 2003). Another simulation model of P runoff from crops and pasture is APLE (Vadas, 2017).

Although the P index concept is widely adopted, the development of the index has changed due to local topography, hydrology and management conditions that influence P transport (Sharpley *et al.*, 2003).

P RUNOFF FROM STORED MANURE

Limited literature is available regarding runoff P losses from manure stored outdoors. Methods to estimate N and P losses in overland flow and other forms of runoff from manure stored outdoors could be derived from the regression equations of the Larney *et al.* (2014) study. Mean total N losses generated from straw bedding dairy compost was around 57 mg (m^{-2} of manure surface area) $minute^{-1}$, while the corresponding value for total P was 8.3 mg (m^{-2} of manure surface area) $minute^{-1}$. A tier 2 method to estimate N and P runoff from stored manure based on this is described below:

Equation A8.3

$$F_{nutrient} = E \cdot Area \cdot CF_{windrow} \cdot T_{RunoffRainfall}$$

where E represents the export coefficient for the nutrient of interest (N: 60 mg [m^{-2} of manure surface area] $minute^{-1}$; P: 8 mg [m^{-2} of manure surface area] $minute^{-1}$), and the surface area of the windrows or stockpiles is determined from the storage area (m^2) multiplied by an area conversion factor ($CF_{windrow}$). To provide an estimate of annual nutrient flow ($F_{nutrient}$) the annual duration of runoff generating rainfall is applied ($T_{RunoffRainfall}$; minutes). It is notable that dissolved P forms in runoff represented a large proportion of total P losses (92 to 96 percent).

Relationships are also provided enabling estimation of runoff losses based on manure or compost N or P content. While a strong relationship was not observed for total P losses in runoff versus manure total P, a linear relationship was observed between water soluble manure-P and total dissolved P in run-off:

Equation A8.4

$$Concentration\ in\ runoff = 6.1 + 0.042 \cdot P_{ws}$$

where C_{runoff} is the total dissolved P concentration (mg $litre^{-1}$), and P_{ws} is the water-soluble P (mg kg^{-1}). The research team used 127 mm hr^{-1} simulated 20-minute rainfall events. Incorporation of this result into Equation A8.3 is modified as follows:

Equation A8.5

$$F_{nutrient} = (4.7 + P_{ws} \cdot 0.0044) \cdot E \cdot area \cdot CF_{windrow} \cdot T_{RunoffRainfall}$$

This observation is supported by earlier work which indicated a strong relationship between simulated rainfall extraction of P from manures and composts (Sharpley and Moyer, 2000).

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Appendix 9

Fertilizer production

The use of N and P fertilizers can have a significant effect on total N and P emissions and the related environmental impacts, and therefore primary data on the fertilizer types and rates of application for feeds shall be used. When primary data is unavailable (e.g. for production of brought-in feeds), then the fertilizer type, composition and rate of application should be based on regional or national data for the region/country from which the feed is sourced. Otherwise, generic data could be used (e.g. see Tables 3, 4 and 5 in LEAP Feeds database document).

Fertilizer production consumes approximately 1 – 2 percent of global energy on an annual basis. By far, the main energy requirement is the fuel and feedstock requirements for ammonia manufacturing, which is equal to some 87 percent of the industry's total energy consumption. For economic and environmental reasons, natural gas is the primary hydrocarbon feedstock in ammonia synthesis, from which almost all N fertilizers are derived. Therefore, the production processes that use less natural gas per unit of ammonia output reduce manufacturing costs.

Energy efficiency in the mass production of N-based fertilizer products has been significantly improved since its inception in the early 20th century – and modern fertilizer factories are quickly approaching the theoretical minimum of energy consumption when producing ammonia.

The voluntary International Fertilizer Association (IFA) benchmarking survey (for 2013-2014 data points) included participation from a total of 66 ammonia plants located in 26 countries, representing approximately a quarter of global ammonia production. Information was gathered for the survey on the participating plant's average net energy efficiency during the previous year based on the following calculation:

Net Energy Efficiency = Feed + Fuel + Other Energy / NH_3 production

These calculations include the energy to produce ammonia as well as the Equation 9.1 energy used for operations, such as start-ups, shut-downs and catalyst reductions. Indirect emissions, or "offsite" emissions related to energy imports, were also calculated to more accurately reflect the overall energetic and environmental footprint of the plants' operations.

On an annual basis, ammonia production facilities generally do not operate at their design energy efficiencies, which are based on continuous operation with equipment and the catalysts in good condition. The plants with a good production year can operate at energy efficiencies approaching the design levels. However, plants with frequent outages, inefficient equipment or poor catalyst activity will have a much higher energy usage than their design. This effect along with the inherent differences in plant design energy efficiencies accounts for some of the large variation in energy efficiencies across the survey base.

Due to the variety of manufacturing processes and raw materials, no single process can be identified as the best practice technology to produce ammonia. However, apart from China, using coal for almost all its ammonia production, a vast majority of ammonia produced worldwide uses natural gas as a raw material.

The average net energy efficiency and production summary for the 66 ammonia plants surveyed over the two-year period was 36.0 GJ (gigajoules) mt NH_3^{-1}, ranging from 24.5 to 49.4 GJ mt NH_3^{-1} - with the top quartile performing in the range of 28 to 33 GJ mt NH_3^{-1}. The latter figures are comparable to theoretical design efficiencies and are near the optimum efficiency level of approximately 28 - 29 GJ mt NH_3^{-1} for a new plant.

There has been a 4 percent improvement in net energy efficiency since the 2002-2003 benchmarking exercise. Overall, an ammonia plant built today uses around 30 percent less energy per tonne of ammonia produced than one built 40 years ago. Technical advances have accompanied economic changes and restructuring has rewarded more efficient producers. In markets where energy costs are high, the average energy consumption in Europe and North America has been drastically reduced through the revamping and closing of inefficient plants. Energy costs have also led to new state-of-the-art units being built in regions like North Africa and the Middle East with abundant sources of affordable natural gas.

Moreover, the move towards higher capacity plants has helped implement more efficient technologies. Capacity upgrades offer a cost-effective opportunity to install more efficient technology. Comparisons of current performance against Best Practice Technologies (BPT) indicate that there is still room for improvement. The BPT energy requirement for the top ten percentile natural gas-based ammonia production facilities operating today is 32 GJ per tonne of ammonia in net energy consumption. This suggests that revamping less efficient existing plants would increase energy efficiency (and decrease CO_2 emissions) by an additional 10 percent. However, the cost would be significant for certain facilities, in some cases exceeding USD 20 million per site.

Finally, the energy requirement for coal-based plants is significantly higher per tonne of ammonia than for natural gas-fired facilities, and a coal-based unit produces roughly 2.4 times more CO_2 per tonne of ammonia than a natural gas-based unit. In view of the availability and the relative costs of energy sources in different regions as well as the policy imperative in China to achieve food security through ensuring domestic fertilizer supply, coal-based ammonia synthesis is expected to increase in coming years. Moreover, Carbon Capture and Storage (CCS) could be an important means to minimizing CO_2 emissions related to coal-based and non-coal-based production in the mid- to long- term.

The following Table A9.1 presents the results of a survey by Fertilizers Europe in 2014.

Table A9.1: Emissions of N_2O and carbon dioxide from fertilizers for European mineral fertilizer production and use in 2011

			GHG emissions (GWP 1000 years, IPCC, 2007)								Energy consumption
			Fertilizer production	Fertilizer use					Fertilizer production + Use		Fertilizer production
Fertilizer product	Symbol	Nutrient content	At plant gate	CO_2 from urea hydrolysis	Direct N_2O from use	Indirect N_2O from use	Indirect N_2O via NO_3^-	CO_2 from liming and CAN	Total	Total	On-site
					kgCO_2e kg product^{-1}				kgCO_2e kg product^{-1}	kgCO_2e kg nutrient^{-1}	MJ kg product^{-1}
Ammonium nitrate	AN	33.5% N	1.18	0.00	1.26	0.01	0.35	0.27	3.06	9.14	14.02
Calcium Ammonium nitrate	CAN	27% N	1.00	0.00	0.89	0.01	0.28	0.20	2.40	8.88	11.78
Ammonium nitrosulphate	ANS	26% N 14% S	0.83	0.00	1.10	0.01	0.27	0.40	2.62	10.09	10.61
Calcium nitrate	CN	15.5% N	0.68	0.00	0.65	0.00	0.16	0.00	1.50	9.67	7.23
Ammonium sulphate	AS	21% N 24% S	0.58	0.00	0.98	0.02	0.22	0.50	2.30	10.95	8.07
Ammonium phosphates	DAP	18% N 46% P_2O_5	0.73	0.00	0.76	0.01	0.19	0.34	2.03	11.27	6.76
Urea	Urea	46% N	0.91	0.73	2.37	0.28	0.48	0.36	5.15	11.19	23.45
Urea ammonium nitrate	UAN	30% N	0.82	0.25	1.40	0.10	0.32	0.24	3.13	10.43	13.84
NPK 15-15-15	NPK	15% N 15% P_2O_5 15% K_2O	0.76	0.00	0.56	0.01	0.16	0.12	1.61	10.71	7.59
Triple superphosphate	TSP	48% P_2O_5	0.26	0.00	0.00	0.00	0.00	0.01	0.27	0.56	0.18
Muriate of potash	MOP	60% K_2O	0.25	0.00	0.00	0.00	0.00	0.00	0.25	0.43	3.00

Source: Fertilizers Europe, 2014, Energy efficiency and greenhouse gas emissions in European N fertilizer production and use

Appendix 10

Example data for upstream processes for fertilizer manufacturing emissions and for energy use and emissions for animal product processing and electricity

FERTILIZER MANUFACTURING EMISSIONS

Limited data on fertilizer manufacturing emissions are available. An average for N_2O emissions from the nitric acid production from Kool *et al.* (2002) is 7 kg N_2O tonne nitric acid^{-1}, with a range of 5-9. These values coincide with the IPCC (2006) default values. For urea production in Europe, EFMA (2000) gave values for emissions from urea production of 0.9-4.1 kg NH_3 (to air) t urea^{-1} (average c. 1.8), 0.5-2.2 kg urea (to air) t urea^{-1} and 0.01-0.61 kg NH_3 (to water) t urea^{-1}.

Data on P emissions during fertilizer manufacturing are difficult to obtain. In an early paper, Silva and Kulay (2003) gave an estimate of P emissions in the effluent to water from superphosphate production in Brazil of 0.65 kg P tonne superphosphate^{-1}. Table A10.1 gives a summary of some N and P emissions from the manufacturing of some common European fertilizers, obtained using ecoinvent version 3.2.

Table A10.1: Example values for N and P losses from manufacturing of European fertilizers

Fertilizer type	N or P form lost	Location of loss	N or P form lost per kg fertilizer (g kg^{-1})
Ammonium Nitrate	Ammonium, ion	water/river	0.74
	Ammonia	air/high population density	0.57
Calcium ammonium nitrate	Ammonium, ion	water/river	0.96
	Ammonia	air/high population density	3.2
Urea	Ammonium, ion	water/river	0.36
	Ammonia	air/high population density	3.5
Single superphosphate	Phosphate	water/river	1.9
Triple superphosphate	Phosphate	water/river	1.9
Monoammonium phosphate	Phosphate	water/river	0.01
	Ammonia	air/high population density	0.13
Diammonium phosphate	Phosphate	water/river	0.01
	Ammonia	air/high population density	0.22

Source: Ecoinvent version 3.2

Table A10.2: Total energy use (electricity and fuels) in Danish and Norwegian abattoirs (from best available technologies in the abattoirs and animal by-products industries 2005)

	Cattle	Sheep	Pig	Poultry
kWh tonnes of carcass^{-1}	90-1094	922 - 1839	110-760	152 - 860

Table A10.3: Typical energy (electricity and fuels) use range during processing of drinking milk

	Separation/ Standardisation	Homogenization	Pasteurisation	Sterilisation	Cooling	Filling/ Packing
MJ kg milk^{-1}	0.004-0.040	0.023-0.031	0.050 -0.210	0.08-0.4	0.019-0.190	0-035-0.036

Table A10.4: Electricity-related NO_x emissions per energy source

Energy source	kgNO_x MWh electricity produced^{-1}
Hard coal	0.3 - 3.9
Lignite	0.2 - 1.7
Natural gas	0.2 - 3.8
Oil	0.5 - 1.5
Nuclear power	0.01 - 0.04
Biomass	0.08 - 1.7
Hydropower	0.004 - 0.06
Solar energy	0.15 - 0.40
Wind	0.02 - 0.11

Source: Turconi *et al.*, 2013

ENERGY USE DURING ANIMAL PRODUCT PROCESSING

An indication of energy use in abattoirs is given in Table A10.2.

Table A10.3 lists the energy use for some of the unit operations involved in the milk industry. Data are based on Brush *et al.* (2011), De Jong (2013), Xu *et al.* (2012) and the International Dairy Federation (2005).

ELECTRICITY NO_X EMISSIONS

Table A10.4 lists the range of NO_x emissions per MWh electricity generation, distinguished by energy source (Turconi *et al.*, 2013). This study showed that fuel quality, plant energy efficiency, plant age and the technology used strongly affect the amount of NO_x emitted into the atmosphere.

Appendix 11

Background principles for eutrophication and acidification

EUTROPHICATION: ENVIRONMENTAL CAUSE-EFFECT CHAIN

Nutrients used to produce feed crops may leach or be carried by runoff into surface water after field application. This process can provide limiting nutrients (e.g. N and P) to algae and aquatic vegetation in excess of natural rates, which may drive a cascade of ecosystem changes, including alterations in aquatic species composition, biomass, or productivity (Henderson, 2015). While many countries have regulations aimed to contain (e.g. catchment basins) or limiting (e.g. field buffer zones) the flow of nutrients (e.g. EU nitrates directive or water framework directive) into surface or groundwater, such approaches are not always effective, and some countries lack such regulations.

Quantifying eutrophication directly from livestock or crop production systems, with access to streams or near streams or water bodies, is difficult given the multitude of factors that may influence the environmental fate of the emitted compounds, the response of the receiving ecosystems, and the effects on the exposed species that compose an ecological community.

LANDSCAPE ATTENUATION OF REACTIVE N AND P

Emissions of reactive N compounds into the atmosphere can result in the deposition of the same compounds in terrestrial and aquatic ecosystems. Once deposited from the air, the reactive-N compounds can be regarded as emissions to terrestrial or aquatic systems and be modelled as waterborne forms. LCIA methods should account for this deposition, allowing the practitioner to determine impacts, e.g. from freshwater due to airborne emissions.

Sources of waterborne N-inputs (mainly dissolved inorganic N, DIN) are typically classified as point or non-point sources, mainly for management purposes, depending on the nature of the emission if it occurs at specific locations (e.g. sewage water discharges or direct emissions to rivers or to marine coastal waters) or diffused in the landscape (e.g. surface runoff and leaching from either natural or agricultural soils) respectively.

N and/or P can potentially contribute to the impacts of aquatic eutrophication. As noted in sections 5.3 and 5.4, there are site-specific differences in the extent of limitation of N and P to ecosystem impacts, with P more commonly being limiting in freshwater bodies and N in marine ecosystems. N emissions to water can be attenuated by denitrification in groundwater systems (Mayorga *et al.,* 2010; Van Drecht *et al.,* 2003), sedimentation, abstraction (consumption) and denitrification in surface freshwater systems (Seitzinger *et al.,* 2006), and further denitrification and advection in coastal marine waters. This attenuation reduces the N substrate and will therefore mitigate the eutrophication potential (Nixon, 1996; Cosme *et al.,* 2017).

P is the most common limiting plant nutrient in freshwater systems and its emission to the system can cause freshwater eutrophication (Correll, 1998; Smith *et al.*, 2006). P emissions, either to soil or to aquatic systems, undergo a series of abiotic and biotic processes that may slow transport, or possibly sequester P in sediments or in mineral forms with reduced bioavailability. In both terrestrial and aquatic systems, most P is sorbed to particulates, rather than existing as dissolved orthophosphate (PO_4^{3-}). Thus, sorption controls soil solution and aquatic concentrations of inorganic P (Froelich 1988; Sharpley 2006).

In rivers and lakes, P may cycle through dissolved, sorbed, and inorganic or organic forms, as a result of abiotic and biotic processes (Haggard and Sharpley 2006). P may be retained in streambeds, especially during low and base flow conditions. However, episodic storm events may re-suspend particulate P (House *et al.,* 1995). Thus, sorption processes influence aquatic transport, precipitation and dissolution, microbial and algal uptake, and floodplain/wetland retention (Haggard and Sharpley 2006). The joint action of these abiotic and biotic processes attenuates the original P-emissions and contributes to the mitigation of their (freshwater) eutrophication potential.

EUTROPHICATION PATHWAYS

Terrestrial eutrophication

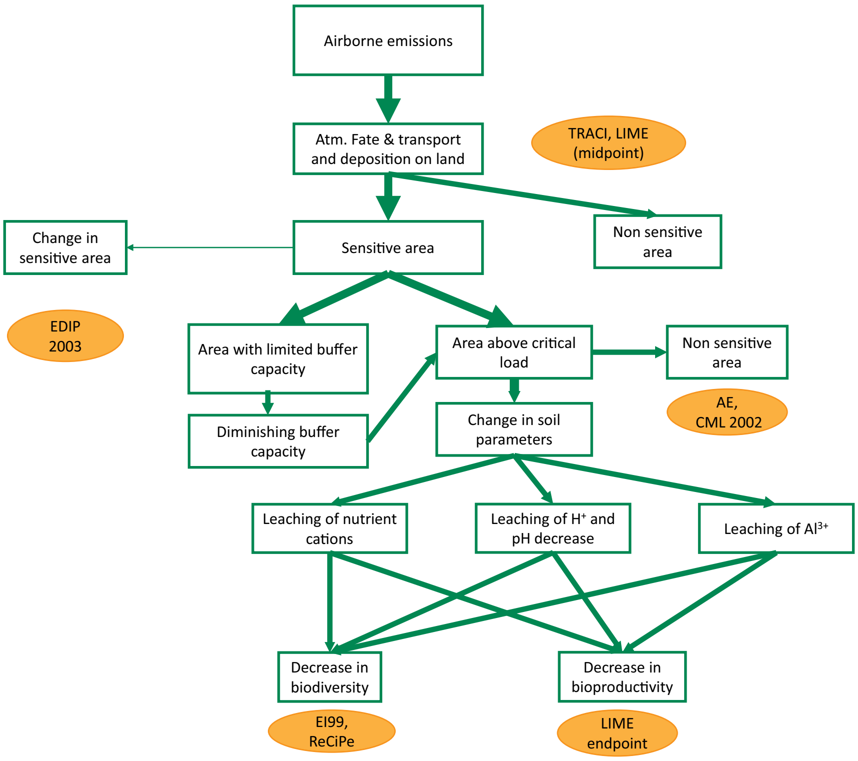
Terrestrial eutrophication originates from the deposition of airborne-N compounds (N oxides, NO_x, from combustion processes, and ammonia, NH_3 volatilized from agricultural activities). Airborne emissions of P-forms are not prevalent; hence terrestrial eutrophication is associated with N-compounds.

Terrestrial plants are usually N limited (Crouzet *et al.*, 2000; Hornung *et al.*, 1994). Excessive supply of N may change the structure and function of terrestrial ecosystems by favouring a (typically) limited number of N-adapted species (Henderson, 2015). This may in turn change the plant community from nutrient-poor (e.g. heath lands, dunes and raised bogs) to nutrient rich plant communities, altering ecosystem structure. Secondarily, it may also change the tolerance of populations to disease or other stressors (e.g. drought, frost), as well as impacts on other species in the terrestrial ecosystem, and contribute to an overall loss of species richness, systems productivity and functioning (EC-JRC, 2010). The primary impact on the plant community leads to secondary impacts on other species in the terrestrial ecosystem (Figure A11.1).

Aquatic eutrophication

Increased input of growth-limiting plant nutrients to well-lit layers of rivers, lakes and coastal waters promotes benthic and planktonic growth of autotrophs (periphyton and phytoplankton, respectively). The cascading cause-effect chain of excessive loading of either P or N into both freshwater and marine systems, may cause changes in the structure and function of ecological communities. The accumulation of planktonic biomass leads to turbidity of the water column and shading of bottom substrates, or to the change of species composition in the community and to the appearance of toxic or harmful algal blooms (HAB); and in both cases leading to the loss of habitat for fish and other plant species - see more on impacts on biodiversity in LEAP principles for the assessment of livestock impacts on biodiversity (FAO, 2016). The eventual sink and decay of this organic matter may lead to excessive consumption of dissolved oxygen in bottom layers; in this case leading to potential onset of hypoxia or anoxia conditions that lead to death or disappearance of animal species. The most sensitive and least mobile are affected first; physiological and behavioural responses may buffer the impact on species but as oxygen depletion intensifies, death or escape follows (Breitburg, 1992; Diaz and Rosenberg, 1995; Gamperl

Figure A11.1
Cause effect chain for eutrophication with reference to the indicators available in various impact assessment methods



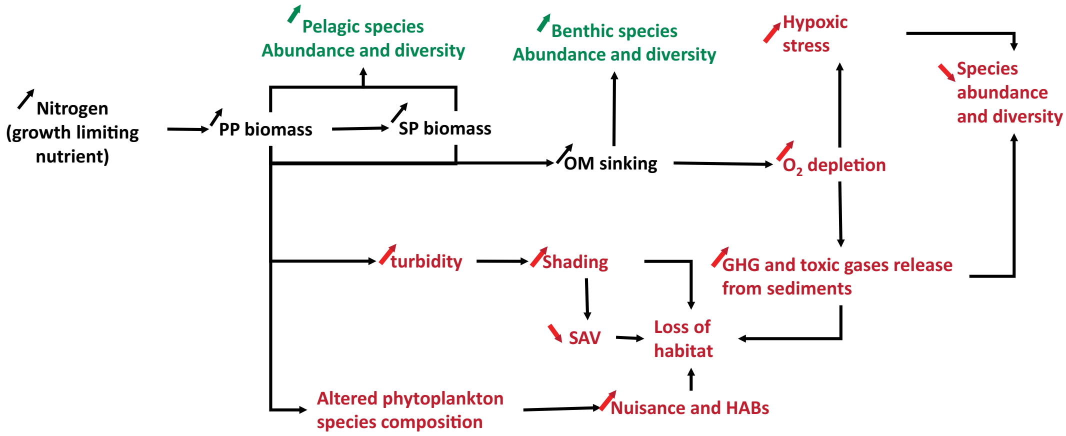
Source: EC-JRC, 2011

and Driedzic, 2009; Perry *et al.*, 2009). At anoxic levels, anaerobic bacteria change their terminal electron acceptor to nitrate, sulphate and carbon dioxide which leads to the release of e.g. hydrogen sulphide and methane from the sediments (Middelburg and Levin, 2009; Reed *et al.*, 2011; Steckbauer *et al.*, 2011). Eutrophication is one of the most severe and widespread causes of disturbance to aquatic ecosystems (Diaz and Rosenberg, 2008; Dodds *et al.*, 2009; GESAMP, 2001). Positive impacts (albeit short-term) may also be found with increasing abundance and diversity of either pelagic or demersal animal species as a result of increased food availability; not limited to planktivorous but also including predator species.

The environmental impact pathways described above are the basis for the aquatic eutrophication characterization factors, although at different levels of completeness and relevance (see Figure A11.1). Figure A11.2 shows the cause-effect chain for marine eutrophication triggered by N-loadings to surface coastal waters (Cosme, 2016).

Although the various impacts mentioned may occur, either on terrestrial or aquatic environmental compartments, in the LCA context, the endpoint eutrophication impacts indicator quantifies the potential loss of species as a proxy for the dimension of biodiversity loss. The same applies to other endpoint or damage indicators that contribute to the ecosystems, like acidification.

Figure A11.2
Schematic representation of the causality chain of cascading effects of N enrichment of coastal waters. Green text corresponds to positive effects and red text to harmful effects to the marine ecosystem



Legend:
primary producers (PP), secondary producers (SP), organic matter (OM), oxygen (O2), submerged aquatic vegetation (SAV), greenhouse gases (GHG), harmful algal blooms (HABs).

Source: adapted from Cosme, 2016.

ACIDIFICATION

Many livestock production system processes can result in emissions of NO_x, NH_3 and SO_x leading to a release of hydrogen ions (H^+) when the gases are mineralized. Acidification is frequently characterized in terms of sulphur dioxide equivalents. The potential terrestrial acidification impacts of beef cattle production systems in the United States were estimated to be 328 g SO_2e per kg carcass weight (Lupo *et al.*, 2013). The main contributors to this impact were manure emissions and handling (286 g SO_2e), followed by minor contributions from feed production (23.2 g SO_2e) and mineral and supplement production (11.5 g SO_2e).

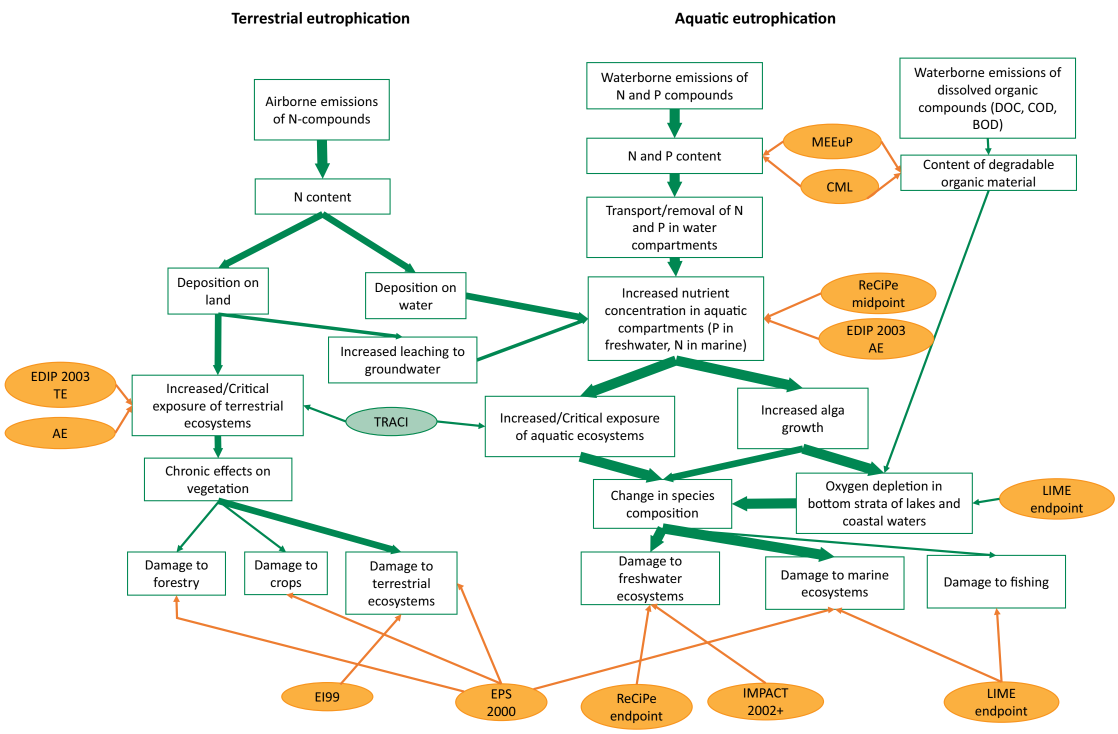
ATMOSPHERIC FATE AND ATTENUATION OF N COMPOUNDS

In the atmosphere, N compounds are transported via advection and, to a lesser degree, dispersion and diffusion. Ammonia and oxides of N may react with other substances. Oxides of N may react with hydroxide to form nitric acid and may react with light and volatile organic compounds to form ozone. Ammonia may form fine particles through reactions with sulfuric and nitric acids. Finally, compounds may be returned to terrestrial or aquatic systems via dry and wet deposition. These reactions and transport mechanisms depend on local atmospheric conditions, such as temperature, atmospheric stability and precipitation.

During their time of transport and transformation in the atmosphere, substances may be transported hundreds of kilometres, although deposition is largest nearest the source of emission (Potting *et al.* 1998; Roy *et al.*, 2012b). In a global model, approximately half the mass of ammonia emissions was predicted to be deposited within a 2° x 2.5° region containing the source of emissions, and 70-80 percent on the same continent; whereas approximately a quarter of N

Figure A11.3

Cause-effect chain for acidification with reference to the different indicators available



Source: from EC-JRC, 2011

oxides are predicted to be deposited in the same region and 50-70 percent on the same continent (Roy *et al.*, 2012b).

Acidification pathway

The deposition of acidifying substances (described above) in terrestrial and aquatic systems, can lead to the release of H^+ that may result in reduced pH, decreased alkalinity, and other biogeochemical reactions (van Zelm *et al.*, 2015). Ammonia can be oxidized through bacterial action to nitric acid, and thus also contribute to acidification. These reactions may have implications for several ecosystem parameters, such as base saturation, the ratio between base cations and aluminium, the ratio of aluminium to calcium, soil solution pH, dissolved Al concentration (Posch *et al.*, 2001). pH changes may lead to mobilization of aluminium and subsequent toxicity, while plants may lose the ability to regulate P or magnesium, may have reduced biomass productivity, may have trouble flowering and reproducing, and acid tolerant plants may begin to outcompete other species (Falkengren-Grerup 1986, Zvereva *et al.*, 2008, Roem and Berendse 2000). The impact pathway for terrestrial acidification is shown in Figure A11.3.

Different terrestrial and aquatic ecosystems react differently to the introduction of acidifying substances, largely driven by the buffer capacity of the system, which is strongly influenced by the underlying geology of the area. Systems rich in carbonate-bearing minerals, such as limestone, tend to have higher buffer capacity than areas with a less reactive substrate, such as granite or soils with very few base cations (van Zelm *et al.,* 2015). The time scale in which a terrestrial system begins to experience acidification depends on biogeochemical processes in the resilience of plants and other soil components to perturbation (van Zelm *et al.*, 2007).

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Appendix 12

Use of Biosolids as fertilizer in agriculture

Increasing global population to more than 9 billion by 2050 (Godfray *et al.*, 2010) results in many challenges in tackling food security. One of the challenges is soil degradation referring to processes like soil erosion (by water and wind), compaction, loss of organic matter, loss of soil biodiversity, contamination, acidification and salinization (European Commission 2006). Whilst the challenges need tackling, the increasing population also offers opportunities that need to be harnessed appropriately. Increasing population will need adequate sanitation facilities which can be more developed centralised facilities (i.e. wastewater treatment plants) producing sludge which is then treated to form biosolids) or less developed isolated facilities (urine diverting toilets, pit latrines) in developing countries that requires further treatment to produce composted material. Biosolids derived from either of these sanitation facilities can offer opportunities to be used as fertilizers in agriculture as reported by Deeks *et al.* (2013) and Pawlett *et al.* (2015) in a developed country context and by Diaz-Aguado *et al.* (2017) in a developing country context.

Biosolids, having undergone rigorous treatment to kill pathogens, can be used as a fertilizer rich in nutrients such and N and P making it suitable for application to agricultural land. Due to the stable nature of the organic compounds, the potentially mineralizable N in biosolids seldom surpasses ~ 12 percent of the total N content and the mineralization occurs within the first 2 weeks after application to soil (Cordovil *et al.*, 2006). However, the sources of biosolids also have a strong influence on the organic N which is potentially mineralizable. Laboratory tests have shown variations from 4 percent to more than 60 percent of the N mineralized after application to soils. Smaller values are associated with aerobically digested materials and those stabilized by composting. Smith *et al.* (1998) categorized biosolids into four different categories: those with high potential to accumulate nitrate and therefore with high leaching risk, those with a low to intermediate potential, those that immobilized N in the soil before releasing nitrate and those where the organic N was resistant to breakdown. Nitrification from organic amendments including biosolids, is a function of thermal time (with a base value of 0 °C) and pH, with faster nitrification occurring at soil pH near neutrality.

Biosolids can be a potential source of P for crops in agriculture and can potentially be a renewable source of fertilizer. Biosolids have been turned into fertilizers by combining it with urea and potash as an N and K source respectively to formulate organo-mineral fertilizers. Deeks *et al.* (2013) have shown that over a period of three years when organo-mineral fertilizers were applied to combinable crop in field scale trials, no significant difference in yield was observed when compared to conventional fertilizers. Pawlett *et al.* (2015) also found similar response when organo-mineral fertilizers were applied to grassland. Whilst this is encouraging and shows that biosolids can be used as a renewable source of P fertilizer, one of the challenges that has not been addressed is the energy cost for drying the organo-mineral fertilizers which were pelletised and dried up to 90 percent dry matter. Energy cost of drying biosolids is a challenge that has not been fully resolved yet.

Charlton *et al.* (2016a, b) carried out a meta-analysis on soils that have had biosolids applied over many years from Long-term Experimental sites in the United Kingdom with a specific focus on the effect of Cd, Zn and Cu on soil microbial biomass and N_2 fixing rhizobia. The results showed that Cd did not have detrimental effects on these biota, whilst Zn and Cu had some ill-effects depending on the treatments but showed signs of recovery.

Life Cycle Assessment (LCA) was carried out on the use of organo-mineral fertilizers in agriculture with the functional unit of sewage sludge produced per head of population. Life Cycle Impact Assessment covers the environmental impacts or burdens of the flows of matter and energy that are of direct concern to the world we live in. There are five important ones that relate to biosolids and the handing of energy, organic carbon, nutrients and combustion.

An LCA was carried out as part of a large EU Framework 7 project, known as End-o-Sludg, aiming to use several wastewater treatment technologies to reduce the generation of sludge. However, when sludge is produced it is generally blended with N and K sources, dried and pelletised to produce organo-mineral fertilizers which can be used as a renewable P fertilizer.

The technologies to reduce sludge production reduce all burdens with an exception of acidification on the largest plants but is very sensitive to any saving in energy usage over the previous systems and the need to maintain or improve phosphate removal from the effluent. There is technical speculation that it may remove so much carbon from the effluent that the activated sludge process changes and may require additional carbon. The activated sludge process is important for denitrification and some N_2O loss.

The technologies to process sludge to produce fertilizers are very sensitive to the extent that they can discontinue the use of heavy fuel oil to run a thermal dewatering unit. It is worth noting that the baseline would also be improved with the application of waste heat recovery technology, but for both systems waste heat is less available in Northern Europe and Scandinavia where winters are deeper and longer, and district heating systems are more common than in the United Kingdom. Generally, the ability of farming to utilise additional nutrients without loss to the environment comes into question as does the use of urea to improve the fertility and agronomic attractiveness of the sludge pellets (Organo-Mineral Fertilizer or OMF) resulting in upward pressure on acidification and global warming.

In the case where both End-O-Sludg Systems are used the effects are largely additive and complimentary resulting in all burdens being reduced for any level of parameter sensitivity. The one exception is the efficacy of phosphate removal from the effluent. Often the sensitivities show that if key processes on the plant are managed well then it more than compensates for the implications of N losses at farm level.

Transport is never really sensitive in the models despite concerns about the fossil energy that is required for bulk haulage of sludge. Greater use of transport can be made if it helps find better uses for sewage sludge, such as ground better able to receive it.

Overall, the systems model-based approach to the LCA of the End-O-Sludg technologies has stimulated systems thinking and systemic insights during the iterative data-results cycle with the project. The work shows that to reduce environmental burdens systemic interventions are required (Sandars and Williams, 2013).

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Appendix 13

Construction of the matrices for the calculation of the life cycle nutrient use efficiency

The supply and use of framework to account for nutrient flows is presented in Table A13.1 based on Uwizeye et al. (2016). The table enables cross-checking mass balances for both product and process at each stage of the supply chain (Suh and Yee, 2011). A mass balance is applied to the product in a way that the sum of the products delivered by the system (A, B and C) equals the sum of intermediate, recycled, final consumption and export of the product delivered by the system. For example, the sum of the products of cropping (e.g. grain and straw harvested and crop residues) equals the sum of crop products (recycled crop residues in the field, feed intake by animals, and exported food crop for human consumption). Based on Table A13.1, the matrix INP refers to the intermediate products used by each process. The matrix PROD refers to total products produced at each stage. The matrix RES refers to nutrient extracted from nature or mobilized from other sources. The matrix SC defines the change in stock and NNB to the nutrient losses at each stage. Furthermore, the final consumption refers to nutrient in end-products delivered to consumers as well as export and indicates nutrient exported to other production systems (e.g. manure applied to legumes and vegetables). Based on these matrices, the life cycle nutrient use efficiency can be calculated. The equations are given in Section 7.4. It is important to note that the mass balance shall be achieved at each stage to avoid mistakes.

Table A13.1: Construction of the matrices for the calculation of the life cycle nutrient use efficiency at chain level

		Product			Process					
		Crop/ Pasture	Animal production	End-products*	Cropping	Breeding	Processing	Final consumption	Export	Total
product	Crop/pasture				Crop residues	Feed intake	0	0	Food crop	A
	Animal co-products				Manure recycled	0	Live animals and raw products	0	Exported animal or manure	B
	End-products				0	0	0	Animal end-product	0	C
					INP[1]					
process	Crop production	Crop and pasture harvested, crop residues	0	0						
	Animal production	0	Manure recycled, live animals and products	0						
	Processing	0	0	Processed animal products						
		PROD[2]								
	Resources mobilisation				BNF, synthetic fertilizer, atmospheric deposition, Manure from other species	Swill, protein-rich supplement, mineral	0			
					RES[4]					
	Change in stock				Stock Change	Stock Change	Stock Change			
					-SC[5]					
	Waste generation				Nutrient Losses	Nutrient Losses	Nutrient Losses			
					NNB[6]					
	Total	**A**	**B**	**C**	**A**	**B**	**C**			

[1] INP: Matrix of aggregated inputs to each stage

[2] PROD: Matrix of products of each stage

[3] IMP: Matrix of imported products, applied as inputs to stage

[4] RES: Matrix of resources mobilised from the nature or other agricultural activities

[5] SC: Matrix of stock change at stage

[6] NNB: Matrix of nutrient losses at each stage

* end-product: edible and non-edible products

** By-products from food or by-fuel industries

Source: Uwizeye *at al.*, 2016

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CASE STUDIES TO ILLUSTRATE INVENTORY DATA AND RESULTS FROM A RANGE OF LIVESTOCK SYSTEMS

Case study 1

Lamb production in New Zealand to consumption in the United Kingdom

The following case study was based on an average New Zealand (NZ) sheep and beef farm in North Island hill country. Average farm survey data from 163 farms collected by Beef+LambNZ (2015) has been used. The production of lamb, processing it in an average abattoir (based on survey data from a range of NZ abattoirs), shipping it refrigerated to the United Kingdom (United Kingdom), a retail stage, home consumption after cooking by roasting, and including the final waste (sewage) stage was observed. All intermediate transport steps were accounted for. Thus, it was a cradle-to-grave study (Ledgard *et al.*, 2011).

The functional unit was 1 kg sheep meat purchased in the United Kingdom.

Relevant farm data is:

1. **Area:** The total utilized farm area (excluding areas in indigenous woody vegetation) was 411 ha. This was based on 408 ha of permanent perennial grass/clover pasture and 3 ha of a brassica forage crop.
2. **Animals:** Sheep and beef cattle were grazed together. Sheep were 1578 ewes (65 kg live-weight, LW), 28 breeding rams, a replacement rate of 27 percent and a lambing percentage of 125. Cattle were 120 breeding cows (500 kg LW), 3 breeding bulls and 239 growing heifers and steers (including purchased cattle). Calving percentage was 87.
3. **Animal production:** Net sheep sales were 59.9 t lamb LW and 21.0 t other sheep LW. Sheep wool sales were 12.4 t greasy wool.
4. **Farm system:** Animals were grazed together outdoors all year round (i.e. no housing or manure management system) with excreta returned directly to soil.

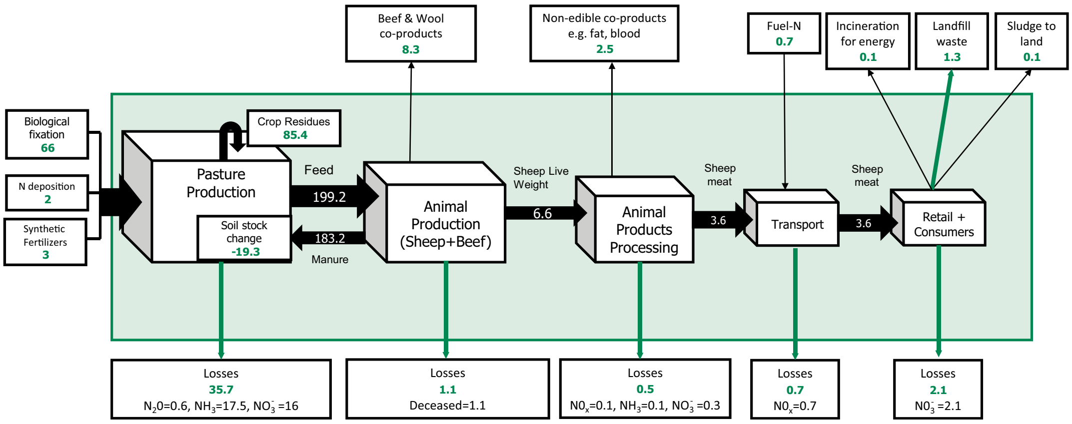
Allocation between sheep and cattle was based on the relative feed intake by each animal type (i.e. biophysical allocation of 56 percent to sheep). Similarly, a biophysical allocation between sheep LW sold for meat and wool of 65 percent: 35 percent was based on the protein requirements for LW and wool production (Wiedemann *et al.* 2015).

Table CS1.1 gives a summary of farm inputs, outputs, animal feed intake and emission of N and P.

Other relevant post-farm inventory data were:

5. **Abattoir:** The percentage of carcass weight relative to live-weight was 50 percent. Primary data for a sheep-only abattoir (average of 11 plants) was used. Specific fuel (coal, gas and LPG) and electricity use were 2.0 and 2.1 MJ tonne lamb processed^{-1}. Waste water was collected and processed via a multiple pond and wetland system, before discharge into waterways (0.9 kg N t LW processed^{-1}).
6. **Shipping:** A refrigerated ship travelled 18,390 km with an estimated fuel use of 0.116 L kg meat^{-1} (based on a range of published values).
7. **Retail:** It was assumed that the frozen sheep meat spent 5 days in a retail cabinet (Carlsson-Kanyama and Faist 2000).

Figure CS1.1
N flows in an NZ hill country sheep and beef farm system (on a per hectare basis) through to consumption of sheep meat in a United Kingdom household



8. **Household:** Sheep meat was assumed to be roasted (using standard recommendations) using 9 MJ kg^{-1} (Foster *et al.* 2006).
9. **Wastewater (sewage):** The model of Munoz *et al.* (2008) modified for meat was used to estimate wastewater processing and emissions from the United Kingdom sewage treatment systems.

Allocation between meat and non-edible co-products (88 percent to meat) was based on economic allocation from a five-year average of prices. LCA involved use of Simapro version 8.3 and ecoinvent version 3.3.

A summary of all N flows is given in Figure CS1.1.

SUMMARY OF RESULTS AND RELEVANT LEARNINGS:

Cradle-to-farm-gate:

Almost all farm N emissions were from animal excreta deposited on pasture (particularly urine-N at 65 percent of all excreted N) and were dominated by ammonia and leached N (Table CS1.1). Estimates of these were based on use of well-validated country-specific tier-2 models (Wheeler *et al.*, 2003; MfE 2016). NO_x emissions from direct fuel use were small and total background emissions from all N forms were negligible, adding 1 percent to the direct emissions (mainly as NO_x from fertilizer production).

Farm P emissions were dominated by soil-P runoff/erosion and fertilizer-P runoff. These represent potential losses, as calculated by a country-specific tier-2 model.

Farm N surplus was largely determined by legume N_2 fixation inputs (66 kg ha^{-1} yr^{-1}), while the relatively low farm P surplus was mainly determined by fertilizer-P inputs (Table CS1.2). Generic research indicates that this hill country is accumulating carbon and N but there are no reliable methods to calculate it and so it has not been accounted for. The farm N footprint of total reactive N losses was mainly determined by ammonia and leached N from animal excreta, while the P footprint was driven by P runoff/erosion from soil and fertilizer (Tables CS1.1 and CS1.2).

Table CS1.1: Summary of inventory for the average NZ North Island hill country sheep and beef farm

		Amount	%N, %P	Data quality (Primary or Secondary)	How calculated (if relevant)	Data type & source	Reference
Inputs (kg ha^{-1} yr^{-1}):	Fertilizer-N (urea)	3	46	1°	NZ av	Farm survey, Industry	Beef+LambNZ 2015
	Fertilizer-P (superphosphate)	7	9.1	1°	NZ av	Farm survey, Industry	
	Legume N fixation	66		1°,2°		f. yield, %legume, %N, root-N	Ledgard *et al.*, 2001
	Atm. N deposition	2		2°	NZ av		
	Electricity (kWh)	5603		2°	Farm survey		
	Fuel use (L)	5720		1°	Fuel expenditure	Farm survey	Beef+LambNZ 2015
Animal Intake:	Pasture (DM)	6615		1°	Energy req model	NZ GHG Inventory	MfE 2016
	Pasture %N, %P		3.0, 0.3	2°,2°		NZ feed database	
	Forage crop (DM)	25		2°	NZ av yield	NZ feed database	
	Forage crop %N, %P		2.7, 0.26	2°,2°		NZ feed database	
Outputs (kg ha^{-1} yr^{-1}):	Net sheep LW sold	196		1°		Farm survey	Beef+LambNZ 2015
	Sheep sold (N, P)	6.6, 1.4	3.4, 0.7	2°,2°	NZ av	NZ/Int. publ.	
	Wool sold	30		1°		Farm survey	Beef+LambNZ 2015
	Wool (N, P)	3.3, 0.003	11, 0.01	2°,2°	NZ av	NZ/Int. publ.	
	Net cattle LW sold	147		1°		Farm survey	Beef+LambNZ 2015
	Net cattle sold (N, P)	5.0, 1.0	3.4, 0.7	2°,2°	NZ av		

		Amount		Method tier no.	How calculated (if relevant)	Data type & source	Reference
Farm emissions (kg ha^{-1} yr^{-1}):	Leached-N	16		Tier 2	OVERSEER model	f. Site factors, Excreta-N, Fert-N	Wheeler *et al.*, 2003
	N_2O	1.6		Tier 2	IPCC (2007)	Country-spec. EF	MfE 2016
	NH_3	17.5		Tier 2	IPCC (2007)	Country-spec. EF	MfE 2016
	NO_x (direct)	0.6		Tier 1		f. Fuel use	Ecoinvent
	Reactive N (indirect)	0.4		Tier 1	Simapro	f. Inputs, e.g. fert., electricity	Ecoinvent
	Runoff-P	0.7		Tier 2	OVERSEER model	f. Site factors, Fert-P	Wheeler *et al.* 2003

Circularity of N and P on farm was high due to recycling via animal excreta, which was nearly four-fold higher than the sum of the new external N and P inputs. Partial life cycle (cradle-to-farm gate) N and P use efficiency were 61 and 87 percent, respectively (see section 6.1). This was associated with high recycling via excreta, but the output in animal products was low relative to the amount of N and P in feed consumed and in external N and P inputs (Table CS1.1).

Table CS1.2: Summary of cradle-to-farm-gate (unless noted otherwise) results for nutrient indicators and impact categories

	Supply chain (kg ha^{-1} yr^{-1}; incl. cattle)	Sheep (kg ha^{-1} yr^{-1})	Sheep (g kg LW sold for meat^{-1})	Sheep (g kg wool^{-1})
Use indicators:				
Gross N surplus	19	15		
Gross P surplus	3.7	2.1		
N footprint			59	209
P footprint			1.3	4.5
N circularity – Inputs (*Icirc*); Outputs (*Ocirc*)	73%; 95%			
P circularity – Inputs (*Icirc*); Outputs (*Ocirc*)	72%; 92%			
N use efficiency (%)				
plant;	89%			
animal;	99%			
processing;	84%			
cradle-to-processor-gate (life-cycle-NUE)	42%			
Impact Category indicators:				
Eutrophication (CML; aquatic+terrestrial) g PO_4e			27	93
Eutrophication (freshwater) g PO_4e			4.0	14
Eutrophication (marine; ReCiPe 2008) g Ne			14	49
Acidification (CML) g SO_2e			117	409

Table CS1.3: Summary of cradle-to-grave results for nutrient indicators and impact categories for sheep meat produced in New Zealand hill country, processed in New Zealand, shipped to the United Kingdom and consumed in the United Kingdom after cooking by roasting. The functional unit (FU) was 1 kg sheep meat purchased in the United Kingdom

	To farm gate	Processing	Trans-port	Retail & consumer	Waste (sewage)	Total
Resource use indicators:						
N footprint (g N kg FU^{-1})	104	2.0	4.8	1.3	23.3	135
P footprint (g P kg FU^{-1})	2.3	0.36	0.002	0.44	1.9	5.0
Impact Category indicators:						
Eutrophication (CML; aquatic+terrestrial) g PO_4e kg FU^{-1}	47	1.9	2.2	1.9	19.5	72
Eutrophication (freshwater) g PO_4e kg FU^{-1}	7.0	1.1	0.07	1.36	5.88	15
Eutrophication (marine; ReCiPe 2008) g Ne kg FU^{-1}	24.6	2.0	0.62	0.25	23.5	51
Acidification (CML) g SO_2e kg FU^{-1}	205	0.23	13.5	9.0	2.4	230

Sheep consumed 56 percent of all animal feed intake (44 percent by cattle) and this was used to allocate emissions between sheep and cattle. However, calculated emissions also recognised the relatively lower N leaching from sheep excreta than from cattle excreta (Hoogendoorn *et al.*, 2011) and that sheep produce co-products of LW sold for meat and wool.

All life cycle stages and Impact Assessment

The N and P footprints were dominated by the farm and sewage stages of the life cycle (Table CS1.3).

Impact Category indicator calculations used methods as described in section 5.4 (not to be added together). For Eutrophication Potential (CML, 2003; using CML-IA baseline v3.04), the farm and sewage stages were dominant contributors, with both N and P sources being important. The sewage stage included an 18 percent contribution from COD.

For freshwater eutrophication potential, the CML method was used for the NZ stages (farm and processing) because NZ surface waters are a mix of being N and/or P limited (McDowell & Larned, 2010). However, for the other post-processing stages the ReCiPe (2008) method (based on P only for Europe) was used because the meat was sold and consumed in the United Kingdom. For freshwater and marine eutrophication indicators, the farm leached-N value was adjusted for 50 percent attenuation (between bottom of root-zone and surface waterways) based on published NZ research. For marine eutrophication potential (ReCiPe 2008), the sewage and farm stages had a similar relative contribution, driven mainly from N emissions to water.

Acidification Potential was dominated by the farm stage, with the next main contributors being the transport and retail+consumer stages. The later stage was dominated by SO_2 from coal burning for United Kingdom electricity, whereas the main contributor for other life-cycle stages was gaseous N emissions.

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Case study 2

Beef and sheep extensive grazing system in Uruguay

This case study was based on one representative extensive Uruguayan beef and sheep farm on North of the country. It used real farm data from one year (2014-2015). The scope of analysis was from cradle to farm gate. The functional unit was 1 kg of equivalent meat[12] produced at farm.

Relevant farm data is:

1. **Area:** The total effective grazing area utilized by the farm was 1399 ha. This was based on 100 percent of natural pasture with a dry matter production of 4500 kg DM ha^{-1}.
2. **Animals:** Beef and sheep cattle were grazed together. Cattle were 323 breeding cows (375 kg LW), 9 breeding bulls, 98 mature 3-year old steers, 123 Rising 2-year old steers, 108 Rising- 1-year old steers, 106 Rising-1-year old heifers, and 228 calves. The pregnancy percentage was 83. Sheep were 1029 ewes (40 kg live-weight, LW), 40 breeding rams, 384 hoggets, 926 lambs (less than 1 year old), 776 lambs (1-2 years old), pregnancy was 89 percent.
3. **Animal production:** Net cattle sales were 101.6 t LW and purchase were 2.8 t LW, while net sheep sales were 18.6 t sheep LW and purchase 0.2 t sheep LW. Sheep wool sales were 8.5 t greasy wool.
4. **Farm system:** Animals were grazed together outdoors all year round (i.e. no housing or manure management system) with excreta returned directly to soil.

A summary of all N flows is given in Figure CS2.1.

SUMMARY OF RESULTS AND RELEVANT LEARNINGS:

Almost all farm N emissions were from animal excreta deposited on pasture (urine-N represents 48 percent of all excreted N) and were dominated by ammonia and leached N (Table CS2.1). Estimates of these were based on IPCC equations and default emission factors. A summary of N flows is given in Figure CS2.1.

Farm P emissions were dominated by runoff of soil-P, as calculated by a country-specific tier-2 model (Perdomo *et al.,* 2015). This was based on 0.47 kg P ha^{-1} of particulate–P from erosion (1 tonnes ha^{-1} yr^{-1}) using a country-specific erosion model (Garcia Prechac *et al.,* 2004) and 0.36 kg P ha^{-1} of dissolved-P, where 0.06 were losses from the soil (3 ppm P Bray I) and 0.3 kg P ha^{-1} were from the dung (using equation in Appendix A8).

Farm N surplus was determined mainly by legume N_2 fixation and atmospheric deposition inputs (2.4 and 5 kg N ha^{-1} yr^{-1}), with brought-in feed equivalent to only 0.58 kg N $ha^{-1}yr^{-1}$. There is high uncertainty (>±100 percent) around these first numbers, with N_2 fixation based on an assumption of 1 percent legume in the pastures and the deposition was a general number from low input areas. The farm P surplus was negative mainly determined by low inputs of P in purchased concentrates and purchased

[12] Equivalent meat_ represents the addition of kilograms of beef and sheep plus kilograms of wool. Kg Equivalent meat= kg beef + kg sheep + (kg wool · 2.48)

Table CS2.1: Summary of farm inputs, outputs, animal feed intake and emission of N and P

		Amount	%N, %P	Data quality (Primary or Secondary)	How calculated (if relevant)	Data type & source	Reference
Inputs (kg ha^{-1}):	Fertilizer-N	0		1°		Farmer	
	Fertilizer-P	0		1°		Farmer	
Brought-in feeds:	Supplement 1 (DM)	16.08	2.24%N 0.3%P	1°,2°		Farmer, Mieres *et al.* (2004)	
	Supplement 2 (DM)	12.38	1.76%N 0.3%P	1°,2°		Farmer, Mieres et al (2004)	
	Legume N fixation (N)	2.4		1°,2°		f. yield, %legume, %N, root-N	Ledgard *et al.*, 2001
	Atm. N deposition (N)	5		2°		Published data	
	Electricity (L fuel)	1000		1°		Farmer	
	Fuel (L)	1000		1°		Farmer	
	Net Beef LW bought	2.0		1°		Farmer	
	Net sheep LW bought	0.13		1°		Farmer	
	Net Livestock LW bought (N, P)	0.06 (N) 0.02 (P)		1°		Farmer	
Animal Intake:	Pasture (t DM ha^{-1})	2.53			Energy req model	NRC	Becoña *et al.* 2014
	Pasture %N, %P		1.28 %N, 0.18%P	2°,2°			Mieres, 2004
Outputs (kg ha^{-1})	Net beef LW sold	70.7		1°		Farmer	
	Net sheep LW sold	13.2		1°		Farmer	
	Beef LW sold	2.3 0.57	3.2%N, 0.8 %P	2°			
	Sheep LW sold	0.43, 0.11	3.2%N, 0.8%P	2°			
	Wool sold	6.06		1°		Farmer	
	Wool	0.68, 0	11.2 %N, 0.01%P	2°,2°			
		Amount		**Method tier no.**	**How calculated (if relevant)**	**Data type & source**	**Reference**
Emissions (kg ha^{-1}):	Leached-N	2.1		Tier 2	Modified IPCC (2007)	f. Excreta-N, Fert-N	MfE 2016
	N_2O	0.9		Tier 2	IPCC (2007)	IPCC	IPCC
	NH_3	5.9		Tier 2	IPCC (2007)	IPCC	IPCC
	Runoff-soluble P	0.06		Tier 2	P index	f. Site factors, Fert-P	Perdomo, et al 2015
	Particulate P runoff	0.47		Tier 2	Erosion 6.1 P index	f. Site factors, Fert-P	Garcia Prechac *et al.*, 2004. Perdomo *et al.*, 2015

Figure CS2.1
N flows in a case-study beef and sheep farm in Uruguay expressed in kg N ha^{-1}

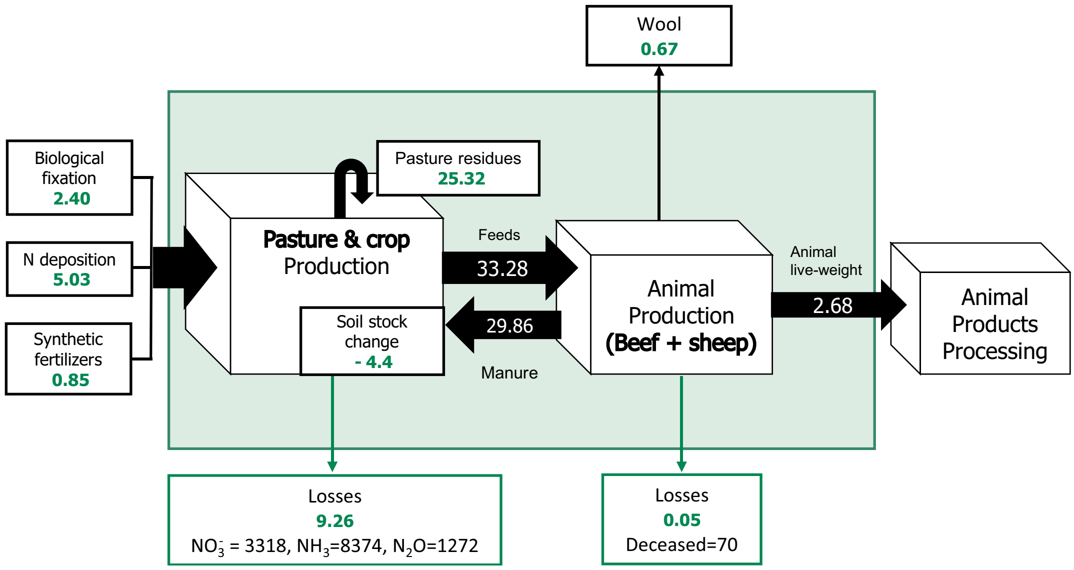

Table CS2.2: Summary of cradle-to-farm-gate results for nutrient indicators and impact categories

	Whole farm (kg ha^{-1} yr^{-1})
Nitrogen use indicators:	
N surplus (excluding soil stock change)	-4.4
P surplus (excluding soil stock change)	-0.4
N circularity input	0.81
N circularity output	0.91
N use efficiency: plant (%)	85%
N use efficiency: animal (%)	99%
Partial life cycle NUE (cradle-to-farm gate) (%)	-10%
Impact Category indicator:	
Eutrophication (aquatic+terrestrial; CML 2002) kg PO_4e	5.2

animal compared to the total P output in products of 0.68 kg P ha^{-1} (Table CS2.2). There is very high uncertainty about whether there is 'natural' release of P from soil minerals in these soils, which have been in native grassland and grazed for over 200 years (Tieri *et al.*, 2014).

There was high NUE at the farm stage associated with low external N inputs and some apparent mining of soil N reserves, although the latter was associated with the high uncertainty in estimates as noted above. This resulted in an apparent partial life cycle NUE of -10 percent. However, it is likely that this system is not mining soil N reserves but that there is some free-living N_2 fixation occurring in these soils (see Appendix 3). A value of 2 kg N ha^{-1} yr^{-1} from free-living N_2 fixation would be sufficient to change the partial life cycle NUE from -10 percent to +10 percent.

This illustrates the significance of small changes in the amount of N flows in the various N pools, which have high uncertainty in their values.

Beef production in Uruguay is mainly on the natural vegetation, "Campo natural", determining an extensive but resilient system tolerant to a large variation in climatic conditions. These are systems with very low amounts of inputs based on a biophysical approach to match animal demand with pasture growth in conditions of high climate variability between years. This aspect determines a constraint of measuring nutrient balance when it only accounts for one year. Natural pasture contains about 400 different native grasses and a low number of legumes (about 1 percent), with a high variability in nutrient content and production, resulting in uncertainty in estimation of the N and P intake by animals.

The lack of national research to estimate N and P losses, determined that IPCC default values were used to estimate gaseous losses (leaching losses were based on NZ grazing factors), and this aspect could have influenced an overestimation in the results.

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Case study 3

Egg (medium size) production, in combination with pigs and cereal production in Sweden

In Sweden there is a free and voluntary advisory programme called "Focus on nutrients" (http://www.greppa.nu/om-greppa/om-projektet/in-english.html). The programme welcomes all farmers with more than 50 ha of arable land or 25 animal units. It started in 2001 and in 2016 about 8500 farmers representing 1 M ha were members. This is about 40 percent of targeted farmers and 52 percent of targeted arable land.

Originally the programme concentrated on nutrients and nutrient losses and all members started with a nutrient balance on the farm. The programme has been extended with a long range of advisory packages including climate impact. The calculations are made in a programme called VERA made by the Swedish Board of Agriculture. The data from the farm survey are primary data. Contents in fodder are primary data from industry. Most other data is secondary. The reference is VERA, Swedish Board of Agriculture with one exception. The values for leaching of N and P is adjusted according to the official environmental monitoring http://www.slu.se/institutioner/mark-miljo/miljoanalys/dv/registersida/

The example below is from a medium sized farm in the middle of Sweden with mainly egg production and cereals.

Relevant farm data is:

1. **Area:** The total utilized farm area (excluding forest) was 85 ha. The crops were barley (78 ha), wheat (3.5 ha) and ley (3.5 ha).
2. **Animals:** The main production was eggs from laying hens (9500 hens). Young hens are bought and kept in production 15 months. To get use of cracked eggs and home-produced barley, 20 pigs per year were raised. The piglets were brought to the farm.
3. **Egg production:** 21 kg eggs/hen and 15 months
4. **Crop production:** 354 000 tonnes of cereals are sold from the farm, some as wheat flour in the farm shop. The production from the ley is sold to a neighbour.
5. **Farm system:** The hens are kept inside all year round. The manure was used on the farm.

Table CS3.1 gives a summary of farm inputs, outputs and calculated emissions of N and P to waterways.

SUMMARY OF RESULTS AND RELEVANT LEARNINGS:

The gross farm N and P balances were small because of the multiple outputs and relatively low nutrient inputs. However, the farmer was concerned about low crop protein content, especially in the wheat, and about low cereal yields. The P content in soil is good and there is no fertilizer (P or N) used, although manure from the poultry is applied to the cereals.

Table CS3.1: Summary of annual inventory and nutrient flows for a mixed 85 ha farm in Sweden

		Amount Kg, l, kWh	Kg N	Kg P	Kg K	Kg CO_2e
Inputs (kg farm^{-1} yr^{-1}):						
Animals	Young hens	9120	246	55	26	12770
	Piglets	500	13	3	1	1600
Brought-in feeds:	Poultry feed	35100	9480	1791	2458	193000
	Legume N fixation		114			
	Atm. N deposition		340			
	Seeds	13500	230	45	58	5400
Bedding	chips	1000	6	1	1	130
Energy	Diesel	7500				24300
	Electricity (water power)	150000				690
Total inputs or GHG emissions			***10429***	***1895***	***2544***	***237890***
Outputs (kg farm^{-1})						
Animals	Hens	12160	328	72	35	
	Eggs	159600	3016	319	255	
	Pig meat	3000	77	16	6	
Crops	Hay, DM	20000	351	60	500	
	Cereals	354000	5805	1203	1522	
Total outputs			***9577***	***1670***	***2318***	
Gross nutrient surplus (kg ha^{-1} yr^{-1})			***+10***	***+2***	***+3***	
Emissions (kg ha^{-1} yr^{-1}):	Leaching, runoff		9	0.3	?	

A recommendation to the farmer was to sell some of the manure and buy mineral N fertilizer to increase the yields and especially the protein content of the wheat used as wheat flour sold in the farm shop.

Case study 4

Fully grazing dairy cattle supply chain in Rwanda

1. OVERVIEW

This case study was based on the grassland-based dairy cattle system, which is found in the Gishwati area, in Western Province of Rwanda. The primary feed resources are mixed pastures composed by 80 percent of Kikuyu grass (*Pennisetum clandestinum*) and 20 percent of white clover (*Trifolium spp.*). The dairy cattle are pure breed or crossbreed between Ankolé and Holstein or Brown Swiss.

The functional unit was 1 kg fat- and protein-corrected milk (FPCM), and the system boundary was from "cradle-to-primary-processing."

Relevant farm data is:

1. **Area:** The total utilized grazing area used was around 11472 ha shared among 1038 smallholder farmers.
2. **Animal production:** 35,710,438 kg FPCM and 458,813.3 kg of meat
3. **Animal Categories:**

Category	Number
Adult female	13427
Adult male	766
Replacement female	3186
Replacement male	467
Young female	4066
Young male	1203
Calves	7878
Adult female sold	1071
Young female sold	0
Young male sold	1038
Calves sold	3424
Deceased Adult female	873
Deceased calves	812
Total	**38211**

4. **Farming system:** Animals were grazed together outdoors all year round (i.e. no housing or manure management system) with excreta returned directly to soil.

Allocation between dairy and beef was based on the biophysical allocation recommended in the LEAP guidelines for environmental assessment of large ruminants supply chains (87 percent:13 percent) (FAO, 2016a).

Table CS4.1: Summary of farm inputs, outputs, animal feed intake and emission of N and P

		Amount	Data quality (Primary or Secondary)	How calculated (if relevant)	Data type & source	Reference
Inputs (kg ha^{-1} yr^{-1}):	Manure N[1]	282		Manure deposited + Manure applied	Field survey	
	Legume N fixation	56			Estimated	
	Atm. N deposition	6.25				Dentener, 2006
	Pasture (kg DM)	14800			Farm survey	
	Biomass/crop residues[2] (kg N)	66.6				
Outputs (kg ha^{-1} yr^{-1})	Total Beef LW sold	114.9			Farm survey	
	Total milk produced (FPCM cow^{-1} yr^{-1})	5156.6			Farm survey	
Other parameters	N content grass	2.72%			Feedipedia	
	Milk Protein content	3.5%				
	Milk Fat content	3.8%				

[1] 50 percent of manure is recycled, another 50 percent is applied as "external manure".

[2] Crop residues include 60 percent of biomass recycled from pasture and 40 percent from external crop residues.

2. CALCULATIONS

Animal stage:

The feed intake was estimated based on metabolizable energy requirement for maintenance, activity, pregnancy, and lactation at 2595 t N (Figure CS4.1). The manure recycled was estimated to be 1616 t N, whereas 617 t N are exported out of the farming systems including 389.8 t N as exported manure to no feed crops and vegetables and 227 t N as animal products (mainly meat and milk).

Pasture/crop stage:

We estimated N input, output, losses and stock change based on Uwizeye *et al.* (2016) and grass utilization at 56 percent based on GLEAM (FAO, 2016b). The biomass recycled was estimated to be 458 t N. We estimated a negative stock change of 1863 t N, meaning that this system depends highly on organic soil N.

Processing stage:

The losses at the processing level were estimated to be 19 t N mainly dominated by organic waste from the abbatoir. Milk loss was not significant.

3. SUMMARY OF RESULTS AND VALUABLE LEARNINGS:

Almost all farm N emissions were from animal excreta deposited on pasture (particularly urine-N at 65 percent of all excreted N) and were dominated by ammonia and leached N (Table CS4.1). We used IPCC guidelines (IPCC, 2006) to estimate different N emission compounds. Table CS4.2 summarizes NUE_N at each production stage and Table CS4.3 summarizes the cradle-to-primary-processing results for nutrient indicators and impact categories.

Figure CS4.1
N flow in grazing dairy systems in Gishwati (Rwanda) in kg N ha^{-1}.
Manure imported refers to the manure applied to the grassland from other farms or deposited from other livestock species

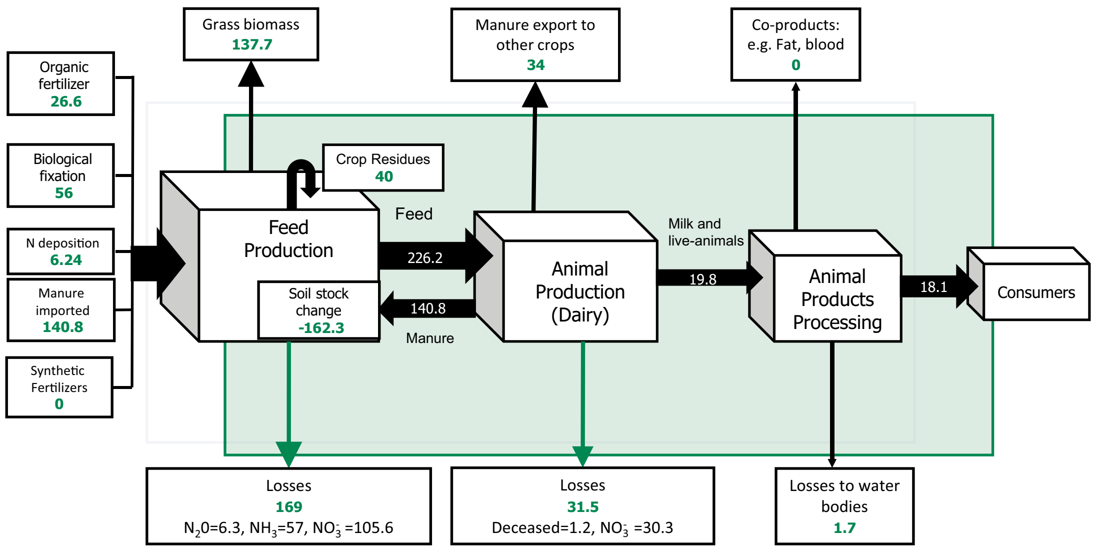

Table CS4.2: Summary of NUE at various stage of the supply chain

NUE_N Pasture production	NUE_N Animal Production	NUE_N Processing (Milk and abattoir)
59%	86%	92%

Table CS4.3: Summary of cradle-to-primary-processing results for nutrient indicators and impact categories

	Entire supply chain
Life-cycle-NUE_N	13%
Per ha	
Life-cycle-NNB_N	197 kg N ha^{-1} y^{-1}
Leached-N (Tier 2)	125 kg N ha^{-1} y^{-1}
N_2O (Tier 2)	7 kg N ha^{-1} y^{-1}
NH_3 (Tier 2)	65 kg N ha^{-1} y^{-1}
Per kg FPCM	
N loss per milk	0.056 kg N $FPCM^{-1}$ y^{-1}
Leached-N (Tier 2)	0.036 kg N $FPCM^{-1}$ y^{-1}
N_2O (Tier 2)	0.002 kg N $FPCM^{-1}$ y^{-1}
NH_3 (Tier 2)	0.018 kg N $FPCM^{-1}$ y^{-1}
Eutrophication kg PO_4e (CML, 2003)	0.016 kg PO_4e
Acidification kg SO_2e (CML, 2003)	0.026 kg SO_2e
N circularity	44%

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