Resource Efficiency in Practice – Closing Mineral Cycles
Final report
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### Disclaimer

(1) The information and views set out in this report are those of the authors and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

(2) The report is based on information collected up to 2015.

### Acknowledgements

This report benefitted from valuable insights provided by numerous experts and practitioners. The authors would like to thank all the external experts that participated to the project. The authors would in particular like to acknowledge the contributions of the regional and national experts (public institutions, NGOs, farmers associations, researchers) from the eight studied regions who offered valuable contributions to the case studies.
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# Glossary

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<td>AAP</td>
<td>Average Annual Population</td>
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<tr>
<td>ABP</td>
<td>Animal By-Product</td>
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<tr>
<td>AD</td>
<td>Anaerobic Digestion</td>
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<td>ADAS</td>
<td>An environmental consultancy in the United Kingdom</td>
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<td>ADEME</td>
<td>Agence de l’environnement et de la maîtrise de l’énergie (French environment and energy management agency)</td>
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<td>AEOS</td>
<td>Agri-Environment Option Scheme</td>
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<tr>
<td>AILE</td>
<td>Association d’Initiatives Locales pour l’Energie et l’Environnement (French Association of Local Initiatives in the field of Energy and Environment)</td>
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<td>ANSES</td>
<td>Agence nationale de sécurité sanitaire de l'alimentation, de l'environnement et du travail (French Agency for Food, Environmental and Occupational Health &amp; Safety)</td>
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<tr>
<td>ASTEE</td>
<td>Association Scientifique et Technique pour l'Eau et l'Environnement (Scientific and Technical Association for Water and Environment)</td>
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<tr>
<td>ATMS</td>
<td>Application Timing Management System</td>
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<tr>
<td>BalticSTERN</td>
<td>Systems Tools and Ecological-economic evaluation – a Research Network</td>
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<tr>
<td>BAT</td>
<td>Best Available Techniques</td>
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<tr>
<td>BETTER</td>
<td>Business Environment and Technology through Training Extension and Research</td>
</tr>
<tr>
<td>BREF</td>
<td>Best Available Techniques (BAT) Reference Document</td>
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<tr>
<td>C</td>
<td>Carbon</td>
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<tr>
<td>C/N</td>
<td>Carbon-to-nitrogen ratio</td>
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<tr>
<td>Ca</td>
<td>Calcium</td>
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<tr>
<td>CaCO₃</td>
<td>Calcium carbonate</td>
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<tr>
<td>CAP</td>
<td>Common Agricultural Policy</td>
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<tr>
<td>CAPRI</td>
<td>Common Agricultural Policy Regionalised Impact model</td>
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<tr>
<td>CDM</td>
<td>Camp Dresser &amp; McKee Ltd</td>
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<tr>
<td>CEVA</td>
<td>Centre d’Etude et de Valorisation des Algues (Center for study and promotion of algae)</td>
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<tr>
<td>CH₄</td>
<td>Methane</td>
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<td>CHS</td>
<td>Confederación Hidrográfica del Segura</td>
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<tr>
<td>CITEPA</td>
<td>Centre Interprofessionnel Technique d'Etudes de la Pollution Atmosphérique (Interprofessional centre for atmospheric pollution studies)</td>
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<tr>
<td>CLM</td>
<td>Dutch independent consultancy active in sustainable food, farming and rural development</td>
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<td>CLRTAP</td>
<td>Convention on Long-range Transboundary Air Pollution</td>
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<tr>
<td>CNC</td>
<td>Comité National de la Conchyliculture (National Committee for Shellfish Farming)</td>
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<td>CO₂</td>
<td>Carbon dioxide</td>
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CORPEN | Comité d'Orientiation pour des Pratiques agricoles respectueuses de l'Environnement (Orientation comity for environment-friendly agricultural practices)
CP | Crude Protein
CRC | Clean up and Restoration Costs
CREM | Centro Regional de Estadística de Murcia (Murcia Regional Statistics Centre)
CSO | Central Statistics Office
CVM | Contingent Valuation Method
CW | Constructed Wetland
DAAS | Danish Agricultural Advisory Service
DAFF | Department of Agriculture, Fisheries & Food
DAFM | Department of Agriculture, Food and the Marine
DAW | Deltaplan Agrarisch Waterbeheer (Deltaplan Agricultural Water Management in Dutch)
DEFRA | Department for Environment, Food and Rural Affairs
DKK | Den Danske Krone (Danish Krone – official currency of Denmark)
DLG | Deutsche Landwirtschafts-Gesellschaft (German agricultural Association)
DM | Dry Matter
DND | Degenerative Neurologic Disease
DRAAF | Direction Régionale de l'Alimentation, de l'Agriculture et de la Forêt (Regional head office for food, agriculture and forests)
DREAL | Direction Régionale de l'Environnement, de l'Aménagement et du Logement (French Regional Directorate for Environmental, Planning and Housing)
DS | Danish Straits
EAFRD | European Agricultural Fund for Rural Development
EC | European Commission
EEA | European Environment Agency
EEC | European Economic Community
EMEP | European Monitoring and Evaluation Programme
EPA | Environmental Protection Agency
EPNB | Expert Panel on Nitrogen Budgets
ERSAF | Ente Regionale per i Servizi all' Agricoltura e alle Foreste
EU | European Union
FADN | Farm Accountancy Data Network
FAO | Food and Agriculture Organisation
FAS | Farm Advisory System
FDSEA | Fédération Départementale des Syndicats d'Exploitants Agricoles (French Departmental Federation of Farmer Unions)
FECOAM | Federación de COoperativas Agrarias de Murcia (Spanish Federation of Agricultural Cooperatives for the Murcia Region)
FSS | Farm Structure Survey
GAEC | Good Agricultural and Environmental Condition
GDP | Gross Domestic Product
<table>
<thead>
<tr>
<th>Acronym</th>
<th>Description</th>
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</thead>
<tbody>
<tr>
<td>GEUS</td>
<td>De Nationale Geologiske Undersøgelser for Danmark og Grønland (Geological Survey of Denmark and Greenland)</td>
</tr>
<tr>
<td>GHG</td>
<td>Greenhouse gas</td>
</tr>
<tr>
<td>GHK</td>
<td>GHK Holdings Limited, a worldwide consultancy service to public and private sector clients</td>
</tr>
<tr>
<td>GIE</td>
<td>Groupement d’Intérêt économique (Economic Interest Grouping)</td>
</tr>
<tr>
<td>GIS</td>
<td>Geographic Information System</td>
</tr>
<tr>
<td>GLAS</td>
<td>Green Low-Carbon Agri-Environment Scheme</td>
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<tr>
<td>GVA</td>
<td>Gross Value Added</td>
</tr>
<tr>
<td>GWP</td>
<td>Global Warming Potential</td>
</tr>
<tr>
<td>H₂O</td>
<td>Water</td>
</tr>
<tr>
<td>H₂PO₄²⁻</td>
<td>Dihydrogen phosphate</td>
</tr>
<tr>
<td>H₂S</td>
<td>Hydrogen sulphide</td>
</tr>
<tr>
<td>HELCOM</td>
<td>Baltic Marine Environment Protection Commission – Helsinki Commission</td>
</tr>
<tr>
<td>HPO₄⁻</td>
<td>Hydrogen phosphate</td>
</tr>
<tr>
<td>IED</td>
<td>Industrial Emissions Directive</td>
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<tr>
<td>IEEA</td>
<td>Eastern river basin</td>
</tr>
<tr>
<td>IEGBNISH</td>
<td>Shannon river basin</td>
</tr>
<tr>
<td>IEESE</td>
<td>South Eastern river basin</td>
</tr>
<tr>
<td>IESW</td>
<td>South Western river basin</td>
</tr>
<tr>
<td>IFA</td>
<td>International Fertiliser Industry association</td>
</tr>
<tr>
<td>IFIP</td>
<td>Institut de la Filière Porcine</td>
</tr>
<tr>
<td>IFREMER</td>
<td>Institut français de recherche pour l’exploitation de la mer (French Research Institute for Exploitation of the Sea)</td>
</tr>
<tr>
<td>IMIDIA</td>
<td>Instituto Murciano de Investigacion y Desarrollo Agrario y Alimentario (Development Institute for Agriculture and Food in Murcia)</td>
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<tr>
<td>INEA</td>
<td>Innovation and Networks Executive Agency</td>
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<tr>
<td>INEMAR</td>
<td>Inventario Emissioni Aria</td>
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<tr>
<td>INRA</td>
<td>Institut National de la Recherche Agronomique (French National Institute for Agronomic Research)</td>
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<tr>
<td>INSEE</td>
<td>Institut National de Statistiques et des Etudes Economiques (National Institute for Statistics and Economic Studies)</td>
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<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
</tr>
<tr>
<td>IRSTEA</td>
<td>Institut national de Recherche en Sciences et Technologies pour l’environnement et l’agriculture (French national Research Intstitute of Science and Technology for Environment and Agriculture)</td>
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<tr>
<td>ISTAT</td>
<td>Instituto nazionale di statistica</td>
</tr>
<tr>
<td>ITAVI</td>
<td>Institut Technique de l'AViculture</td>
</tr>
<tr>
<td>JRC</td>
<td>Joint Research Centre</td>
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<tr>
<td>K</td>
<td>Potassium</td>
</tr>
<tr>
<td>K⁺</td>
<td>Potassium ion</td>
</tr>
<tr>
<td>KIWA</td>
<td>Kiwa is a Dutch independent organization providing highly qualified certification</td>
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<tr>
<td>KT</td>
<td>Kattegat</td>
</tr>
<tr>
<td>Acronym</td>
<td>Full Form (with explanations if necessary)</td>
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<tr>
<td>KZGW</td>
<td>Krajowy Zarzad Gospodarki Wodnej (Polish enterprise: Krakow Water Management Board)</td>
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<tr>
<td>LRTAP</td>
<td>Long-range Transboundary Air Pollution</td>
</tr>
<tr>
<td>LSU</td>
<td>Livestock unit</td>
</tr>
<tr>
<td>LTO</td>
<td>Land- en Tuinbouw Organisatie (Dutch Federation of Agriculture and Horticulture)</td>
</tr>
<tr>
<td>LUCAS</td>
<td>Land Use / Cover Area frame statistical Survey</td>
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<tr>
<td>LULUC</td>
<td>Land Use and Land Use Change</td>
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<tr>
<td>MAGRAMA</td>
<td>Ministerio de AGRicultura, Alimentación y Medio Ambiente (Spanish Ministry of Agriculture, Food and Environment)</td>
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<tr>
<td>MD</td>
<td>Market damages</td>
</tr>
<tr>
<td>MEDDE or MEDDTL</td>
<td>Ministère de l'Ecologie, du Développement durable et de l'Energie of Ministère l'Écologie, du Développement durable, des Transports et du Logement (French Ministry in charge of Environment)</td>
</tr>
<tr>
<td>Mg</td>
<td>Magnesium</td>
</tr>
<tr>
<td>MNP</td>
<td>Milieu-en Natuurplanbureau (Netherlands Environmental Assessment Agency)</td>
</tr>
<tr>
<td>MS</td>
<td>Member State</td>
</tr>
<tr>
<td>N</td>
<td>Nitrogen</td>
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<tr>
<td>N/P</td>
<td>Nitrogen-to-phosphorus ratio</td>
</tr>
<tr>
<td>N₂</td>
<td>Dinitrogen</td>
</tr>
<tr>
<td>N₂O</td>
<td>Nitrous oxide</td>
</tr>
<tr>
<td>NAP</td>
<td>Nitrates Action Plan</td>
</tr>
<tr>
<td>NDN</td>
<td>Nitrification-DeNitrification</td>
</tr>
<tr>
<td>NEC</td>
<td>National Emissions Ceilings</td>
</tr>
<tr>
<td>NEEDS</td>
<td>New Energy Externalities Developments for Sustainability</td>
</tr>
<tr>
<td>NH₃-N</td>
<td>Ammoniacal nitrogen</td>
</tr>
<tr>
<td>NH₃</td>
<td>Ammonia</td>
</tr>
<tr>
<td>NH₄⁺</td>
<td>Ammonium</td>
</tr>
<tr>
<td>NLWKN</td>
<td>Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz</td>
</tr>
<tr>
<td>NMD</td>
<td>Non Market Damages</td>
</tr>
<tr>
<td>NO</td>
<td>Nitric oxide</td>
</tr>
<tr>
<td>NO₂⁻</td>
<td>Nitrite</td>
</tr>
<tr>
<td>NO₃⁻</td>
<td>Nitrates</td>
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<tr>
<td>NOC</td>
<td>N-nitroso compounds</td>
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<td>NOₓ</td>
<td>Nitrogen oxide</td>
</tr>
<tr>
<td>NUTS</td>
<td>Nomenclature of Territorial Units for Statistics</td>
</tr>
<tr>
<td>NVZ</td>
<td>Nitrate Vulnerable Zones</td>
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<tr>
<td>O</td>
<td>Oxygen</td>
</tr>
<tr>
<td>O₂</td>
<td>Dioxygen</td>
</tr>
<tr>
<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
</tr>
<tr>
<td>OM</td>
<td>Organic Matter</td>
</tr>
<tr>
<td>ONEMA</td>
<td>Office National de l’Eau et des Milieux Aquatiques (French national agency for water and aquatic environments)</td>
</tr>
<tr>
<td>Abbreviation</td>
<td>Description</td>
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<tr>
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<tr>
<td>P</td>
<td>Phosphorus</td>
</tr>
<tr>
<td>P₂O₅</td>
<td>Phosphorus pentoxide</td>
</tr>
<tr>
<td>PAC</td>
<td>Policy Actions Costs</td>
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<td>PACA</td>
<td>Provence Alpes Côte d’Azur (French region)</td>
</tr>
<tr>
<td>PBL</td>
<td>a Dutch leading manufacturer of lawnmower blades and agricultural cutting parts</td>
</tr>
<tr>
<td>PDO</td>
<td>Protected Designation of Origins</td>
</tr>
<tr>
<td>PGI</td>
<td>Protected Geographical Indication</td>
</tr>
<tr>
<td>PM</td>
<td>Particulate Matter</td>
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<tr>
<td>PO₄³⁻</td>
<td>Phosphate ion</td>
</tr>
<tr>
<td>PSIU</td>
<td>Piano Straordinario di Interventi Urgenti (Emergency action Plan)</td>
</tr>
<tr>
<td>PTUA</td>
<td>Programma di Tutela e Uso delle Acque (Programme for the Protection and Use of Water)</td>
</tr>
<tr>
<td>PURPLE</td>
<td>Peri-Urban Regions PLatform Europe</td>
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<tr>
<td>PUVD</td>
<td>Passive Use value damages</td>
</tr>
<tr>
<td>PWC</td>
<td>Price Waterhouse Coopers</td>
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<tr>
<td>RBD</td>
<td>River Basin District</td>
</tr>
<tr>
<td>RBMPs</td>
<td>River Basin Management Plans</td>
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<td>RDP</td>
<td>Rural Development Programmes</td>
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<tr>
<td>REA</td>
<td>Renewable Energy Act</td>
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<tr>
<td>REFIT</td>
<td>Renewable Energy Feed In Tariff</td>
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<tr>
<td>REPS</td>
<td>Rural Environment Protection Scheme</td>
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<tr>
<td>RSBP</td>
<td>Royal Society for the Protection of Birds</td>
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<tr>
<td>RWMA</td>
<td>Regional Water Management Authority</td>
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<tr>
<td>S</td>
<td>Sulphur</td>
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<tr>
<td>SAPARD</td>
<td>Special Accession Programme for Agriculture and Rural Development</td>
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<tr>
<td>SATAGE</td>
<td>Service d’Assistance Technique à la Gestion des Epandages (Technical Assistance Service Management Spraying)</td>
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<tr>
<td>SDAGE</td>
<td>Schémas Directeurs d’Aménagement et de Gestion des Eaux (Scheme for planning and managing water at the basin level)</td>
</tr>
<tr>
<td>SEAI</td>
<td>Sustainable Energy Authority of Irland</td>
</tr>
<tr>
<td>SIAM</td>
<td>Sistema de Informacion Agraria de Murcia (Agricultural Information Systems of Murcia)</td>
</tr>
<tr>
<td>SO₂</td>
<td>Sulphur dioxide</td>
</tr>
<tr>
<td>SOC</td>
<td>Soil Organic carbon</td>
</tr>
<tr>
<td>SOM</td>
<td>Soil Organic Matter</td>
</tr>
<tr>
<td>SWR</td>
<td>Soil Water Retention</td>
</tr>
<tr>
<td>TCSL</td>
<td>Techniques Culturales Sans Labour (Tillage Conservation techniques)</td>
</tr>
<tr>
<td>TSG</td>
<td>Traditional Speciality Guaranteed</td>
</tr>
<tr>
<td>UAA</td>
<td>Utilised Agricultural Area</td>
</tr>
<tr>
<td>UNECE</td>
<td>United Nations Economic Commission for Europe</td>
</tr>
<tr>
<td>UNFCCC</td>
<td>United Nations Framework Convention on Climate Change</td>
</tr>
<tr>
<td>US</td>
<td>United States</td>
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<tr>
<td>Abbreviation</td>
<td>Definition</td>
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<tr>
<td>US EPA</td>
<td>United States Environmental Protection Agency</td>
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<tr>
<td>UVD</td>
<td>Use Value Damages</td>
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<tr>
<td>W</td>
<td>Watt</td>
</tr>
<tr>
<td>We</td>
<td>Watt electronic</td>
</tr>
<tr>
<td>WFD</td>
<td>Water Framework Directive (EU)</td>
</tr>
<tr>
<td>WTP</td>
<td>Willingness To Pay</td>
</tr>
<tr>
<td>WWF</td>
<td>World Wildlife Fund</td>
</tr>
<tr>
<td>WWTP</td>
<td>Waste Water Treatment Plant</td>
</tr>
<tr>
<td>ZAR</td>
<td>Zone d’Accès Restreint (Reinforced Action Zones)</td>
</tr>
<tr>
<td>ZES</td>
<td>Zone en Excédent Structurel (Structural Excess Zones)</td>
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</tbody>
</table>
1. Introduction

1.1 Context

Nutrients such as nitrogen (N), phosphorus (P) and potassium (K) are essential elements for living organisms, including plants, animals and bacteria. They are used as fertilisers in agriculture to guarantee high yields and quality products. However, the increasing demand in food production, but also in feed, fibre and fuel, has resulted in an increasing use of N, P and K. In addition, a number of inefficiencies are observed in the mineral cycles. Inefficient use may occur through over-fertilisation, when the amount of fertiliser applied – and the level of N, P and K already present in the soil – exceeds crops’ nutritional needs.

A persistent surplus of N and P in the soil leads to environmental impacts on water (nitrate pollution, eutrophication, acidification, etc.), air (air quality and acid rain), climate (GHG emissions contributing to global warming), soil (acidification and accumulation of heavy metals) and biodiversity (loss of species). Furthermore, nutrient surpluses also have adverse consequences on socio-economic aspects: human health issues, additional costs for drinking water treatment, etc. All impacts depend on what practices are implemented and on local conditions in terms of climate and soil but also on the eventual magnitude of nutrient overload in environmental compartments.

Focusing on resource use is particularly important when a specific resource is or is going to be scarce, and its further depletion could impede human well-being and economic growth. Focusing on impacts is relevant when the use of a resource poses immediate threats to human and ecosystem health. Moreover, the inter-connection of the N, P and K cycles requires that the balance of nutrients be considered globally, and in relation to the carbon cycle. Thus, increased resource efficiency can be achieved by (Tan & al., 2013):

- Using fewer resources to fulfil the same needs;
- Increasing the (socio-economic) value and benefits from the use of (the same amount of) resources;
- Reducing the environmental impacts and damage associated with the use of resources.

Current EU policies aim at improving one or several aspects of nutrient efficiency: Nitrates Directive, Water Framework Directive and other “daughter directives” relating to water, National Emissions Ceilings (NEC) Directive, Industrial Emissions Directive (IED), Common Agricultural Policy (CAP), Renewable Energy Directive, etc. Furthermore, some possible solutions to improve nutrient efficiency are already in place in the EU. They include processing techniques, nutrient recycling (e.g. through recycling of manure and crop residues) and fine-tuning the fertilisation of cropland and grassland (including adequate determination of crop needs, the use of new fertilisation technologies and improved water and soil management). Nonetheless, the real challenge of closing mineral cycles is to transfer these policies into concrete measures that can be implemented by farmers at the regional level, and in saturated areas in particular.
1.2 Objectives

The scope of the project is the EU-28, with a strong focus on a selection of nutrient saturated EU regions. The project covers all three primary macronutrients: nitrogen (N), phosphorus (P) and potassium (K).

The project has two main objectives:

The first objective is to identify the most promising measures at the regional and farm levels, in particular in saturated areas, to improve the use of nutrients and to reduce their negative impacts. These include increasing resource efficiency at the farm level, using processing techniques and transferring manure from saturated areas to non-saturated areas.

Furthermore, possible measures will be considered as good practices for a given region when considered relevant and feasible in its specific context. Operational characteristics of the solutions should be tailored to match the requirements of specific farming systems and provide the most cost-effective solutions for N, P and K while also addressing the pressing environmental issues in a given area.

Specific actions related to this first objective are the following:

- Identify the effects of agriculture on the nutrient cycles as well as the subsequent impacts of nutrient excess on the various environmental compartments, on climate, and on living organisms;
- Identify saturated areas by analysing nutrient budgets for N and P at the EU level. This will allow targeting relevant regions for subsequent activities of the project, in particular dissemination of good practices;
- Identify nutrient-scarce areas;
- Describe major impacts related to the farming systems in each of the targeted regions;
- Describe, through case studies in targeted regions, the causal links between environmental phenomena resulting from high nutrient surpluses and economic sectors that are affected, with a view on qualifying, and quantifying when possible the cost of impacts resulting from an inefficient use of N, P and K;
- Identify the potential solutions to reduce the impacts of agricultural activities conducted in saturated areas on air, water, soil, climate, biodiversity and human;
- Identify the good practices that could be implemented at farm-level considering local conditions such as climate, soil, existing legal requirements, etc.;

The second objective is to communicate the information gathered to stakeholders, especially farmers, farmers’ associations, and regional decision-makers in an educational way to empower them to take action at their level. Four regional conferences as well as a final conference in Brussels\(^1\) took place during the course of the project and yielded useful feedbacks from targeted end-users on the content of this report and the developed dissemination materials (i.e. leaflets prepared for several EU regions).

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\(^1\) 28 October 2014, Portlaoise, Ireland – http://mineral-cycles.eu/ireland;
2. Effects of agriculture on nutrient cycles

This chapter gives an overview of the main effects that agriculture can have on nutrient cycles. First, in section 2.1, it describes how nutrient cycles operate by detailing possible transformations and transfers of nutrients in the different environmental compartments and how natural phenomena can be affected by human activities. Second, in section 2.2, it details the possible causal links between agricultural practices and surpluses of nutrients. Third, in section 2.3, it presents the potential environmental issues posed by nutrient surpluses.
2.1 Transformation and transfer of nutrients in the different environmental compartments (soil, water, air, biota)

2.1.1 Nutrient cycles in natural ecosystems

*Nitrogen*

Nitrogen is present in the air, soil and water. Nitrogen compounds in nature are divided into two groups: nonreactive and reactive. Nonreactive N is gaseous nitrogen (N\textsubscript{2}). Reactive nitrogen occurs as nitrogen oxide (NO\textsubscript{x}), nitric acid (HNO\textsubscript{3}), ammonia (NH\textsubscript{3}), nitrates (NO\textsubscript{3}\textsuperscript{-}), nitrite (NO\textsubscript{2}\textsuperscript{-}), ammonium (NH\textsubscript{4}\textsuperscript{+}), and organic nitrogen compounds (e.g. urea or proteins). They are generally scarce in the natural environment. N\textsubscript{2} and reactive N include all biologically, photochemically, and radiatively active N compounds in Earth’s atmosphere and biosphere (Galloway, et al., 2003). The amount of reactive nitrogen is artificially increased, however, by converting inert nitrogen gas into mineral fertilisers and by fuel combustion (EEA, 2012a).

Due to the many different forms in which nitrogen occurs in the environment and the complexity of the nitrogen cycle, the environmental impacts of nitrogen are very diverse and can affect various ecosystems. There are several processes that occur in the air, the soil and the water that comprise the natural nitrogen cycle (see Figure 1).
Nitrogen is one of several plant macronutrients. It is necessary to build amino acids, which are the base for proteins from which multitudes of plant components are built (e.g., structural components such as cell walls, functional components such as chlorophyll or enzymes, etc.). An excess of nitrogen can be as harmful to the plant as a shortage of the nutrient (see section 2.2.2.1). Plants take up nitrogen in form of ammonium and nitrate, mainly from the soil via the plants’ roots. These nitrogen compounds in soils have three natural sources: nitrogen fixation by bacteria, decomposition of organic matter, and atmospheric deposition.

**Nitrogen fixation by bacteria**: Specific bacteria can fix the elementary, nonreactive nitrogen from the atmosphere. Some of these bacteria live in symbiosis with plants such as leguminous species. Others occur freely available in the soil.

** Decomposition of organic matter** is the source of the biggest share of plant available nitrogen. Organic matter such as dead plants, animals or microorganisms, as well as faeces, contains organically bound nitrogen that is not available for plant uptake. Bacteria and fungi facilitate the decomposition from organically bound nitrogen to ammonium, which is available to plants.

**Atmospheric deposition**: Lightning causes nitrogen molecules to be broken, which then combine with oxygen in the air forming NO\(_x\). These dissolve in rain, converting into nitrates that are transported to the soil and are part of the natural N deposition. Atmospheric deposition is the main source of reactive nitrogen for natural land and forests (Leip, et al., 2011).

Three forms of nitrogen are present in the soil: ammonium, nitrate and nitrite. They are in constant transformation as they are part of metabolic processes of soil bacteria. These bacteria transform a large share of ammonium to nitrate (**nitrification**). The reaction happens quickly after ammonium becomes available so that most of the ammonium is transformed to nitrate before plants take it up through their roots. Other types of bacteria transform nitrate to elemental nitrogen gas (**denitrification**). During these transformation processes, nitrite (NO\(_2^-\)) is formed, which is toxic to cells but quickly transformed to nitric oxide (NO) and nitrous oxide (N\(_2\)O) and, in an additional step, to elemental nitrogen (N\(_2\)). This reaction occurs in microsites of anaerobic conditions contained in aerobic soils (Leip, et al., 2010). These microsites are typically formed after the incorporation of carbon-rich materials into the soil. The decomposition of these materials at high soil moisture levels reduces the levels of O\(_2\) and increases the microbial demand for NO\(_3^-\), which in turn leads to the production of N\(_2\)O.

In degassing, nitrogen oxides (NO\(_x\)) will interact with atmospheric compounds. After degassing from soils, most NH\(_3\) is transferred back to the terrestrial ecosystem shortly after being emitted and deposited onto land and soils, increasing atmospheric deposition of nitrogen (Galloway, et al., 2003). The deposition also occurs on water. In water, N is present in the same dissolved forms as in soils (ammonium, nitrate and nitrite). Nitrate occurs mostly in water, as it is highly soluble and mobile in water. An important part of N in surface waters is in the form of organic N (as dissolved organic N and particulate organic N).\(^2\) Apart from atmospheric deposition, nitrogen compounds in water stem from leaching and run-off processes from soils. As well as in soils, denitrification and nitrification occur in water. These processes are a natural sink for reactive nitrogen forms in water.

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\(^2\) This part is dominant in semi-natural catchments throughout Europe and remains a significant component of the total N load even in nitrate-enriched rivers (Durand, et al., 2011)
Box 1 – Focus on run-off and leaching (common to all nutrients)

**Run-off** occurs at the soil surface during heavy rainfall or other water erosion processes. Along with run-off water, excess nutrients can be transferred from agricultural fields to surface water bodies, either dissolved in water or associated with soil particles and organic matter. Run-off intensity varies with topography (slope), climate (frequency and intensity of rainfalls), land use (grass, crops, bare soil) and soil structure and texture (sand, clay, silt), which determines surface permeability and the capacity of soils to retain water. In general, run-off especially occurs on low-permeable and waterlogged soils (Groenendijk & Kroas, 1999; Groenendijk, et al., 2005; Schroder, et al., 2010) and more intense run-off and nutrient transfer is expected on steeper slopes (Zingg, 1940). In addition, nutrient transfer via overland flow is more important during winter due to high water discharges. Furthermore, risks of run-off, and thus transfers of nutrients, are higher in heavy soils (i.e. with high clay content) impeding the infiltration of water relative to light soils (i.e. with high sand contents) which present high infiltration capacities (Groenendijk & Kroas, 1999; Groenendijk, et al., 2005; Schroder, et al., 2010).

**Leaching** consists in the infiltration and drainage of water in the soil, carrying away excess nutrients contained in the soil out of reach of roots. Dissolved in groundwater, nutrients can reach ever deeper soil layers, down to the aquifer. The behaviour of nutrients and their transfer through the soil to groundwater is determined essentially by soil texture, organic matter content, moisture content, temperature, and pH. For instance, nutrient transfers to groundwater are expected to be greater in sandy soils, with low retention capacity, than in clay or silty soils. Another example is that nutrients can easily be leached from highly organic soils (e.g. peat) because organic matter improves soil structure, facilitating water flows through soil and thus nutrient transfers to aquifers (Bomans, et al., 2005).

Some nutrients are preferably more subject to either one or another of the above two mechanisms. This is the case on the one hand for nitrogen, which is highly soluble and mobile in water and thus mostly transferred from topsoil to groundwater via leaching, and on the other hand for phosphorus which is mainly retained by solid matter and primarily transferred to surface water through run-off of water carrying soil particles (Schroder, et al., 2010; INRA, 2013a).

Run-off transfers nutrients to water bodies faster than leaching. Because of the time lag between the application of nutrients to the soil and their leaching into groundwater bodies, increases in pollution of groundwater can still be observed despite the actions taken to prevent or reduce nitrate leaching (Deneufbourg, et al., 2010).

When groundwater is connected to surface water, a third mechanism allows nutrient transfer, namely the flow from groundwater to surface water. This specifically happens when the level of groundwater tables rises up to topsoil water after intense rainfalls and when groundwater tables finally show on the surface either at the bottom of a watershed filled by a river or at the emergence point of a non-permeable soil layer on which the table flows (Groenendijk, et al., 2005; INRA, 2013a).

The time for nutrients to transfer to water bodies depends notably on the watershed structure and the location where the mechanism takes place in the watershed (i.e. at the top, the middle, or the bottom of the watershed) (Molénat, et al., 2009; INRA, 2013a).
Phosphorus

Phosphorus is a major component of the genetic material and cell membranes of living organisms. In plants, phosphorus is also essential for photosynthesis, which is a process that converts solar energy into chemical energy, and for the production of seeds and fruit (European Commission, 2013a; Cordell, 2010). Animal organisms, including humans, get phosphorus from the plants and animals they consume, while plants uptake phosphorus mostly from the soil solution.

In soil, phosphorus occurs in (Ramade, 1999) (see Figure 2):

- Solid form as a mineral, linked to soil particles such as clay, iron and aluminium minerals or calcium carbonates or as organic matter;
- Ionic form in soil solution as orthophosphates ($\text{PO}_4^{3-}$, $\text{H}_2\text{PO}_4^{2-}$, $\text{HPO}_4^{-}$)

In soil, phosphorus mostly exists in forms bound to soil particles and incorporated in organic matter (50 – 90 %) rather than in forms dissolved in the groundwater (10 – 50 %) (Molénat, et al., 2009). The majority of phosphorus forms available in soil cannot be directly taken up by plants. Plants only take up

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3 The speed of run-off ranges from 1 meter per second to 1 meter per minute whereas the infiltration rate throughout the soil can reach approximately 1 meter per year (Molénat, et al., 2009; INRA, 2013a).
Orthophosphates, mostly $\text{H}_2\text{PO}_4^-$ and to a lesser extent $\text{HPO}_4^{2-}$ (Frossard, et al., 2004).

Orthophosphate is naturally present in the soil solution due to the release of phosphorus when phosphate minerals dissolve. Depending on soil pH, phosphate minerals occur as apatite in case of basic pH or as iron and aluminium phosphates under acidic conditions (University of Hawai - College of Tropical Agriculture and Human Resources, 2014). The dissolution of minerals to plant-accessible orthophosphates is highest in conditions of pH between 6 and 7 in the soil (Frossard, et al., 2004).

Orthophosphates can also be released by desorption. Desorption describes the release of orthophosphates from soil particles such as metal oxides and hydroxides or clay minerals (Frossard, et al., 2004). Whether the orthophosphate is taken up or released by soil particles depends on orthophosphate concentration as well as on soil and climate conditions (mineral type, texture, soil organic matter (SOM) content, pH, temperature and water saturation) (University of Hawai - College of Tropical Agriculture and Human Resources, 2014).

Orthophosphate can also be formed during decomposition processes by microorganisms. Organic matter such as animal excreta or dead organisms builds another source of phosphorus in the soil. The share of organic phosphorus varies typically between 30 and 65 % of total phosphorus in soil, but sometimes up to 90 % (Harrison, 1987; Schulte & Kelling, 1996; Turner, 2008; Johnston & Steen, 2000). Bacteria, mycorrhizal fungus or plant roots split the phosphate bonds with the help of enzymes. As a result, orthophosphate is released (Tadano, et al., 1993; Dighton, 1983). The effectiveness of the reaction is dependent on the physicochemical properties of the soil such as SOM content, pH, temperature, moisture, and oxygen concentration (Davet, 1996; Cornell University - College of Agriculture and Life Sciences, 2005).

As phosphates are mainly bound to solid matter, they are primarily transferred to surface water through run-off water carrying soil particles (Schroder, et al., 2010; INRA, 2013a). Leaching of phosphorus may also take place, especially if the components that are capable of bonding with phosphate are unavailable or if the concentration of phosphorus in the soil is too high and thus the capacities of soil particles for bounding phosphate are exhausted. In the water compartment, phosphorus is in the mineral form of $\text{H}_2\text{PO}_4^-$ and $\text{HPO}_4^{2-}$ ions for pH values between 2 and 12 although the maximum solubility occurs close to pH 6.5 (FAO, 1987). A sink for phosphorus is the adsorption to sediments and the long-term development to apatite (Lucas, 1997).

Phosphorus is also lost through wind and water erosion of the soil. Wind and water transport the particles to which phosphate is bound to surface waters.

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4 Their concentration in the soil solution is about 0.2 mg/L (Schvartz, et al., 2005), ranging from 0.001 mg/L to 1 mg/L according the soil type (Brady & Weil, 2002).
Potassium plays a key role in absorption of nutrients by plant and maintenance of cell turgor (rigidity) through its transfer function and role in the regulation of water in plants. It also participates in protein and sugar synthesis by activating production enzymes (Johnston, 2003).

In soils, potassium occurs in ionic form ($K^+$) in the soil solution or in solid form (K) in soil rocks and minerals, adsorbed in clay particles and as organic matter (see Figure 3). Potassium ions are directly available in soil solutions but only represent 0.1 to 0.2 % of the potassium in soils (Krizic, n.d.). As plants are only capable of taking up potassium in ionic form, most potassium cannot be assimilated by plants.

The main source of directly available potassium for plants is the decomposition of organic matter derived from plants and animals.

**Dissolution of soil rocks and minerals**: a large majority of soil potassium (90 to 98 %) is found in soil rocks and minerals. As part of feldspars and mica, it is integrated into the crystalline structure of these minerals. In acidic environments, the minerals dissolve and liberate potassium ions into the soil solution (Savaget & Bevan, 1988; Brantley & Stillings, 1994; Basak & Biswas, 2008). This reaction is very slow (International Potash Institute, 2012) and depends on the physico-chemical characteristics of the surrounding materials. In soils with considerable amounts of mica, freeze and thaw cycles (physical soil weathering) also contribute to the release of potassium (Laboski, 2011).

**Desorption of clay minerals**: 1 to 10 % of the soil potassium is adsorbed in these interlayers of clay minerals. This fraction is called the non-exchangeable fraction of potassium in the soil. It becomes available when the soil minerals are decomposed through weathering. Other factors can affect
exchangeable potassium. Wetting and drying macerates and shrinks the clay structure. These processes increase or decrease the amount of exchangeable potassium in soils.

2.1.2 Main drivers having an effect on nutrient cycles

2.1.2.1 Environmental conditions

Soil conditions

Nutrient transformation and transfer within the terrestrial ecosystem is heavily influenced by soil type and the presence of negatively charged soil particles such as clay minerals. The type of rock from which the soil formed, the soil texture, soil structure and soil organic matter (SOM) content highly affect water and nutrient retention capacity in soil (Lockwood, et al., 2014, ongoing):

Soil type. which derives from parent bedrock material, influences the natural content of nutrients in the soil. Apatite, a phosphorus-containing mineral, is of volcanic origin. It is present in the soil and is dissolved eventually into phosphate and other products. With aging of the soil, the available phosphorus content thus increases. After it has become available through the dissolution of the parent material, phosphorus often forms sparingly soluble salts with calcium, magnesium, and iron.

Soils rich in organic matter, e.g. chernozems, occur in Europe as a result of the ice age. Organic matter is a nitrogen source in soils. Also, drained bogs consisting mainly of organic matter are very fertile but the enhanced mineralisation processes that occur in these soils are a major source of greenhouse gases.

Soil texture represents the physical texture of soil related to the relative proportion of three texture classes: clay (<0.02 mm), silt (<0.075 mm) and sand (<2mm). The higher the amount of clay is, the higher the capacity to store cationic (positively charged) nutrients. Soil texture has an influence on the risk of leaching. Indeed, clay soils have a greater capacity to retain water, and hence store nutrients, than sandy soils. Due to the cation exchange between clay minerals and soil solution, the nutrients are then available to plants. Soils rich in clay shrink in case of drought. They build macropores in which fertilisers can reach deeper soil layers directly, out of the reach of plants. In general, soils with high clay content are more prone to run-off, resulting in a loss of nutrients. Soil texture also influences potential nutrient emissions to air. Indeed, anaerobic conditions are easily created in clay soil and may results in air emissions (Rochette, 2011).

As regards nitrogen compounds, only ammonium can be stored in clay minerals. Nitrate is negatively charged and hydrophilic and therefore rather present in the soil solution. With water percolating through the soil, nitrate is leached from the soil. All forms of nitrogen can be washed out also with the run-off water that transports soil particles and soil solution at the same time.

Soil structure is the arrangement of solid parts of soil and pore space located between them (Marshall, et al., 1979). Soil structure highly influences the risk of leaching and run-off. Indeed, soil structure drives the soil water retention (SWR) capacity. Thus, soil with low SWR capacity may be more quickly saturated in water than a soil with a high SWR capacity, inducing leaching and/or run-off and pollution by nutrients.

Soil structure is influenced by soil texture and SOM content. Moreover, surface structure mostly influences the risk of run-off. The formation of a waterproof, superficial soil layer resulting from rain action (known as a “slaking crust”) increases the risk of run-off. It also creates anaerobic conditions that may lead to nitrogen emissions.

Compaction from heavy machinery, livestock grazing, or poor maintenance of the soil structure affects the porosity and thereby the water infiltration capacity, which leads to increased denitrification and less mineralisation of organic nitrogen (Ruser, et al., 1998; Velthof, et al., 2011; Duiker, 2004). Nitrate losses
from leaching will decrease while organic nitrogen in soil organic matter will increase. Since compaction reduces root growth, it can also reduce nutrient uptake by plants (Duiker, 2004). In addition, poor water infiltration can cause more surface run-off into water bodies.

**Soil organic matter (SOM)** is the organic fraction of the soil and is composed of several fractions with different levels of decomposition. SOM includes elements such as carbon, hydrogen, nitrogen and oxygen that are components of organic compounds. Soil organic carbon (SOC) is the main component of SOM by weight. It refers to the amount of carbon stored in soils and is expressed in g C/kg soil. Soil organic matter is a key indicator of soil quality and productivity, and is affected by nutrients introduced through the application of manure and chemical fertilisers.

![Soil organic matter (SOM) diagram](image)

Source: (DEFRA, 2007)

**Figure 4 – Synchronicity between the disappearance of soil N content, N supply and crop uptake**

Organic matter also has the potential to bind to certain types of nutrients. The cation exchange capacity (CEC) of organic matter, i.e. its capacity to retain cationic nutrients, is 4 to 50 times higher than clay (Cornell University - College of Agriculture and Life Sciences, 2007). In particular, this means that, through mineralisation, organic matter is a high source of nitrogen available for plants that is retained in soil and consequently not lost as ammonia. The SOM content also influences the potential nutrient losses: if the nutrient content of soil is not taken into account by the farmer when he calculates the dose of fertiliser he needs to apply, the risk of over-fertilisation is high, potentially leading to nutrient losses.

In Europe, 45 % of soils are thought to have a low or very low organic content matter (0-2 % organic carbon). These soils are located primarily in southern Europe but also in areas of France, the UK and Germany.

Land degradation encompasses many different issues that affect soil characteristics. It also includes **soil erosion**. **Soil erosion** consists in the removal of soil particles containing nutrients by water, wind or agricultural practices such as tillage (Eckelmann, et al., 2006; Velthof, et al., 2011). It is a natural process, occurring over geological time (Joint Research Centre, 2012a) and modulated by geographical, soil and climatic conditions. Human activities, and agriculture in particular, can accelerate erosion up to a level where it becomes irreversible when the erosion rate is higher than the rate of soil formation (1 t/ha/yr within a time span of 50-100 years (Kirkby, et al., 2004)). Soil losses affect nutrient cycles since it decreases nutrient content in soil and its nutrient (and water) retention capacity, potentially reducing assimilation of nutrients by plants. Landslides involve the mass movement of soil from a sloping location.

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5 Soil formation can vary from 0.3 to 1.4 t/ha/yr (Verheijen F., Jones, Rickson, & Smith, 2008). Hence, tolerable rates of erosion can be variable following rates of soil formation under different geoclimatic conditions.
due to physical causes, such as excessive rainfall, which can cause large amounts of soil containing N, P, and K to run-off into water bodies in addition to decreasing the availability of fertile soil in that location (Velthof, et al., 2011). In particular, erosion is a significant source of P loss. While P losses by drainage can make up between 12-60 % or more of phosphorus transport, erosion can make up 40-90 % (Bergstrom, et al., 2007). Current estimates of P losses by erosion amount to 10 kg P/ha in arable land (Smit, et al., 2009). The chronic release of phosphorus from sediments causes persistent eutrophication issues, possibly during several years (Carpenter, 2005).

**Water availability**

Water availability in soil is fundamental for nutrient cycles and plant growth since nutrients are available to plants only in a dissolved form in the soil solution. For instance, plants can only directly use N in a solution ($\text{NH}_4^+$ and $\text{NO}_3^-$) to meet their N requirements. Similarly, plant roots absorb P mainly as orthophosphate ($\text{H}_2\text{PO}_4^-$ or $\text{HPO}_4^{2-}$) from the soil solution (BIO Intelligence Service, 2013). This means that plants cannot correctly assimilate the nutrients under water stress. Thus, the nutrients that are not assimilated by crops stay in the soil and may potentially be leached if the surplus is too high. Therefore, closing nutrient cycles requires addressing different challenges and proposing different solutions for regions with and without water issues (EEA, 2007a).

Water availability depends on:

- Water resources available, related to precipitation (and hence climate), water resources from surface water and groundwater, and water abstraction;
- Soil water retention (SWR) capacity that is linked to SOM and soil structure, in particular bulk density.

In 2007, approximately 17 % of the EU territory was experiencing water scarcity issues and the phenomenon has continued to worsen since then (European Commission, 2012a). Water stress and scarcity$^6$ are generally more pronounced during summer months in southern Europe. Water scarcity is expected to affect half of EU river basins in 2030 (European Commission, 2012b) (EEA, 2005a).

Soil water content is one of the factors that determine nutrient concentrations in the soil solution and their availability for plants (Janssen, et al., 1990). Soil water retention (SWR) capacity is a key parameter of soil water content since it ensures the capture, storage and release of water in soil. Soil texture, soil structure and soil organic matter play direct and preponderant roles in the mechanisms of SWR as they influence the overall porosity, pore-size-distribution and specific surface area. Soil moisture is also a determining factor of the quantity of emissions produced, as it influences the supply of oxygen and the availability of $\text{NH}_4^+$ and $\text{NO}_2$ to nitrifying bacteria, and $\text{NO}_3^-$ and $\text{NO}_2$ to denitrifying bacteria. Moreover, high soil moisture content may dilute fertilisers while low moisture may decrease the infiltration of fertilisers into the soil.

**Climate conditions**

Two main climatic parameters affect nutrient cycling: temperature and precipitation.

**Temperature** has an influence on seed germination and photosynthesis, which in turn have an effect on crop growth. Ultimately, temperature influences nutrient uptake. The regulation by temperature of

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$^6$ A water body is considered to be under limited availability, or “stress”, when the abstraction of freshwater represents 20 % of the long-term average freshwater resources. Severe scarcity occurs where this percentage exceeds 40 %.
evapotranspiration processes that release soil water into the atmosphere also affects soil water availability for plants.

In addition, temperature also affects the organic matter content of soil. In particular, the degradation of organic matter is higher in Mediterranean climates and leads to air emissions (Turbé, et al., 2010). Hence, agricultural land requires more nutrient inputs than in other regions.

**Precipitation** provides water to plants and recharges groundwater. It can affect fate of nutrients in several manners, according to the quantity of rainfall, the duration and the frequency. The effect of precipitation also depends on the type of soil, the landforms and the land cover since water uptake varies according to the type of crops and the related evapotranspiration. Long and intense precipitation events may saturate soils with water and thus induce leaching. In the winter months, precipitation is generally higher. The risk of losing the applied fertilisers through run-off and leaching processes is therefore higher, which is among the reasons for the requirement of the Nitrates Directive to define periods when the land application of certain types of fertiliser is prohibited (Cherrier, et al., 2014). If soils become waterlogged because of too much rain and a low crop uptake, denitrification can occur, leading to N\(_2\)O emission and N loss from the agricultural nutrient cycle. Combined with frost, the soil water in the soil pores freezes. Therefore, some pores are clogged, whereas bigger pores stay open. The water, and with it the nutrients, will drain through macropores to the groundwater. Heavy rainfall may also cause erosion that may result in a significant loss of phosphorus.

### 2.1.2.2 Anthropic activities affecting nutrient cycle

Nutrient cycles are affected and even disturbed to some extent by human activities. It concerns industry, agriculture, as well as households. For instance, forms of phosphorus from laundry detergents and fertilisers both contribute to phosphorus overload in water. While natural nutrient cycles are fairly well balanced, the application of organic and mineral fertilisers to the soil to improve its fertility can lead to an excess of nitrogen, phosphorus and potassium (see section 2.1.2.1).

**Nitrogen**

Agricultural and industrial processes have caused significant and unprecedented changes to the global nitrogen cycle and introduced excessive amounts of reactive nitrogen into environmental systems (EEA, 2012a). With the invention in the beginning of the 20\(^{th}\) century of the Haber-Bosch process to fix nitrogen for mineral fertilisers at an industrial scale, the input of reactive nitrogen in the nutrient cycle has increased. The agricultural systems produce more feed, fodder and fibre, but also more livestock waste and nutrients remain in the environment.

Excessive inputs of organic and chemical nitrogen into the soil can affect the rates of the N transformation processes, including nitrification, denitrification, mineralisation, immobilisation, ammonia emission, and it threatens the storage, filtering, and buffering functions of the soil. Generally, higher inputs of nitrogen compounds to agricultural soils results in greater degassing. Nitrogen oxides (NO\(_x\)), degassing from the application of mineral fertilisers will interact with atmospheric compounds. Nitrogen inputs also affect soil organic matter levels, and low SOM can cause more water and air losses of

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\(^7\) See the paragraph on soil conditions above

\(^8\) The Haber-Bosch process synthesises ammonia by reacting hydrogen with atmospheric-N\(_2\). The Haber-Bosch process has changed the way nitrogen fertilisers are produced and used and has increased the availability and use of fertilisers. It is an important part of the "Green Revolution" of the 20th century.
nitrogen from the nutrient cycle (Velthof, et al., 2011). When there are high levels of nitrates in the soil, it is likely that they will be leached due to their solubility.

**Phosphorus**

Organic and mineral phosphorus inputs influence the phosphorus cycle. Sustainable management of the phosphorus cycle is required to address potential phosphorus excess in soil, but also to prevent future potential phosphorus depletion and the security of phosphorus supply (Cordell, 2010; European Commission, 2013c). Indeed, unlike nitrogen, phosphorus is mined and cannot be synthesized. Phosphorus fertilisers are mainly made from apatite that is dissolved with an acid to make phosphoric acid ($\text{H}_3\text{PO}_4$). In solution, phosphoric acid can dissociate and form orthophosphate ions (acid-base reaction) that are directly available for plants in the soil. In this way, the fertiliser is directly available for plants in the soil.

A phosphorus peak is estimated to be reached in the second half of the 21st century, with an average estimate of 20709 (European Commission, 2013a). Once the peak has been passed, the production volume of phosphorus may decrease10 (Cordell, et al., 2013).

Since about 90% of mined phosphorus is used to produce food and animal feed, the agriculture sector is very dependent on mined phosphorus. In addition, phosphate rock reserves are not equally geographically distributed since three quarters of the reserves are located in Morocco and Western Sahara. Consequently, the EU is almost totally dependent on imported phosphorus (European Commission, 2013a).

**Potassium**

Organic and mineral fertilisers are an important source of potassium in agricultural soils. Mineral potassium fertilisers (such as potassium chloride or sulphate) are obtained from potassium precipitates from marine sources, by mining or by evaporation of brines (International Plant Nutrition Institute, 2010).

Excessive potassium inputs may increase the quantity of potassium that is not fixed on soil particles or precipitated into minerals and thus possibly leached11. Excessive concentration of ions in soil solution may induce aggregation of water contained in soil pores and lead to water run-off or leaching downward to the groundwater. Wind and water erosion are also important causes of potassium losses from the soil. Indeed, the clay particles to which potassium generally binds, are very fine particles (less than 2µm) and easily erodible. Water erosion transports soil particles but can also dissolve potassium towards waterways. Hence, the small quantities of potassium released to soils, in addition to high loss risks by leaching especially in sandy soils, and erosion may expose crops to potassium deficiency (Askegaard, et al., 2003).

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9 The study estimates a probable peak between 2051 and 2092. A previous study from the same author estimates the peak between 2030 and 2040 (European Commission, 2013a).

10 A "resource peak" describes the point in time when the production volume of a resource reaches a peak and then declines, constrained by the energy requirements and economics of extracting lower quality and less accessible reserves.

11 Annual leaching loss of potassium from the soils in a humid region under agricultural production (receiving only moderate rate of K fertiliser) is usually about 25 to 50 kg K/ha (Krzic, n.d.).
2.1.3 Interaction between cycles

The N, P and K cycles presented above are one component of a complex system of interactions between various biogeochemical cycles (C, H₂O, S, O, Ca, etc.). It is essential to keep in mind that each of these cycles should be understood in relation to the others as they are all closely inter-connected. The interactions occur between nutrients but also at different scales (cell, organism, land, etc.) and in different compartments (air, water, and land).

**Carbon and nitrogen cycles** interfere mainly in the soil compartment. The fluctuation in C/N ratio over time, in response to shifting substrate quality and the presence of microorganisms, influences reactions related to carbon and nitrogen and thus the availability of both nutrients. N fixation is enhanced by high efficiencies and low microbial C/N ratios, whereas the reverse conditions favour N mineralisation.

Regarding **phosphorus and nitrogen interactions**, though some occur in the soil compartment, the N/P ratio impacts the aquatic environment to a greater extent. In the soil, biologically active phosphorus controls the nitrogen cycle. Microbial growth processes are the main points at which N cycling is adjusted to the P supply in soil-plant-atmosphere systems. In aquatic systems, the N/P ratio impacts the development of primary producers such as phytoplankton. At the scale of plants, nitrogen inputs influence phosphorus availability to plant and therefore improve plant growth (Peyraud, et al., 2012) (IPNI, 1999). Yield is also higher when nitrogen, phosphorus and potassium are used together instead of individually (IPNI, 1999). For instance, if potassium inputs are insufficient, nitrogen is not well assimilated by plants and the yield can be limited by up to half its potential (Johnston, 2003).

**Nitrogen cycles also interact with sulphur cycles.** The S and N cycles are intertwined due to influences of both elements on the soil environment and the tendency of each element to become directly involved with the other in organisms or in enzymes.

As nutrient cycles are in constant interaction, a change in one cycle may have major impacts on all cycles at different scales (cell, plant, land) and in different environments.

2.2 Effects of agriculture on nutrient transformations and transfers

Agricultural practices can have an impact on the fate of nutrients across the different compartments. In addition to environmental conditions, ultimate effects of nutrients on the environment and human health depend on a combination of practices related to the farming production system, the choice of agricultural practices and the intensity of production.

According to the CAPRI farm typology, the farming production systems can be based either on:

- crop production, relying on specialised or mixed crops including arable and forage crops (cereals, oilseed, and protein crops) and permanent crops (fruit, citrus, olives, and vines); or
- breeding production, relying on specialised or mixed livestock including cattle-dairying, cattle-rearing and fattening, sheep, goats, pigs and poultry; or
- a mix of crops and livestock.

These different farming production systems require that choices be made regarding numerous practices needed to produce livestock – such as the feeding practices and manure storage as detailed in section 2.2.1 – and/or to produce crops – such as fertilisation, soil and water management as detailed in section 2.2.2.

The combination of different practices chosen determines the intensity of the production system, from intensive to extensive. While intensive farming systems (known as conventional) are largely based on use of plant protection products and fertilisers aiming at maximising farming production yields, extensive
farming systems target optimised yields requiring moderate application of plant protection products and take into account critical parameters like natural fertility and texture of the soil, climate and water availability as well as plant and animal nutrient requirements. Some farming systems (commonly called integrated or sustainable) adjust their activity between intensive and extensive practices whereas others (known as organic) totally ban the use of chemical products.

The following sections describe the notable impacts of individual and combined agricultural practices on nutrient transformations and transfers.

2.2.1 Role of livestock production

Livestock rearing is a significant source of emissions to air, water and soil. These arise primarily from the production and deposition of manure (emissions from manure to air comprise NH₃, N₂O and Particulate Matter (PM), nutrients loss to water but also from enteric fermentation). Emissions from livestock rearing are determined by variables including type of livestock raised, type of livestock housing used and livestock feeding regimens.

Livestock can be reared indoors or outdoors, depending on the type of livestock and the breeding system. For example, cattle (beef and dairy) can be raised indoors, outdoors, or more commonly spend time both indoors and outdoors. Poultry (both meat broilers and layers) can be raised either in enriched cages or in non-cage systems, and can be raised exclusively indoors or raised in housing with access to outdoor areas (free-range poultry). Pigs can be raised either entirely indoors, or in indoor housing with outdoor access.

2.2.1.1 Type of livestock breeding

Animal type

In terms of overall number of livestock, poultry is the most abundant type of livestock in the EU (over 1 billion head, 2010), followed by pigs (over 153 million heads, 2010), and cattle (approx. 112 million head, 2010) (Eurostat data). Figure 5 shows the relative importance of different types of livestock across the EU. It shows that France, Spain and the UK are the Member States with the largest number of livestock, followed by Poland, Italy and Germany.
Changes in the total number of livestock between 2007 and 2010 show a slight decrease in the number of dairy cows (5%) and a larger decrease (13%) in the number of pigs. The number of cattle has remained relatively stable, as has the number of laying hens. However, these figures are for total livestock numbers at the EU level, and do not reflect the variation in livestock density across different EU Member States, and across different regions. In Austria, Croatia, Denmark, Finland, Italy, Latvia, Luxembourg, the Netherlands, Portugal and Spain, the total LSU\textsuperscript{12} of the holdings with livestock has increased between 2007 and 2010 (Eurostat, 2013a).

The livestock density is also an important element to take into account with regards to the management of nutrients. For example, the Figure 6 shows total livestock units per hectare at the NUTS 2 level across EU Member States (in LSU/ha UAA, 2010). It highlights certain NUTS 2 regions in Spain, the Netherlands, Denmark and France as having the highest livestock densities in the EU (all above 4 LSU/ha UAA).

\textsuperscript{12} Livestock unit (LSU) is a reference unit that facilitates the aggregation of livestock from various species and age, based on the feed requirement of each animal type. The reference unit (= 1 LSU) is defined by Eurostat as the grazing equivalent of one adult dairy cow producing 3 000 kg of milk annually, without additional concentrated foodstuffs. Further information available from:
Nutrients in livestock rearing

Manure produced by livestock can differ in composition and, in particular, the level of dry matter content. Whilst there is no precise distinction between the types of manure, the composition and dry matter content vary according to the type of livestock, feed, housing system and bedding material used.
The main types of manure are as follows:

- Slurry is produced in livestock rearing systems using concrete or slats instead of straw bedding. Slurry consists of excreta produced by livestock mixed with rainwater and wash water and in some cases, waste bedding and feed;
- Liquid manures are sometimes distinguished from slurry and designate the liquid fraction of manure obtained after using separation techniques for manure treatment;
- Solid manure comprises material from covered straw yards, excreta from livestock or solids from mechanical slurry separators; and
- Litter based farmyard manures contain the material (e.g. straw, wood shavings) that have been used as bedding for animals that has absorbed the faeces and urine.

Different livestock contribute to the overall manure production in different ways, for example, most of the manure from pig rearing is slurry whereas the majority of manure from poultry is deep litter.

There is no official reporting of the amount of manure produced across the EU, however a recent report estimated that the entire manure production in the EU nears 1.4 billion tonnes (Agro Business Park, 2011). The detail of the manure per livestock and type of manure for the EU is presented in Table 1.

<table>
<thead>
<tr>
<th>Manure Type</th>
<th>Cattle</th>
<th>Poultry</th>
<th>Pigs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Solid</td>
<td>27</td>
<td>8</td>
<td>8</td>
</tr>
<tr>
<td>Liquid</td>
<td>5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Slurry</td>
<td>41</td>
<td>3</td>
<td>84</td>
</tr>
<tr>
<td>Deep litter</td>
<td>28</td>
<td>97</td>
<td>3</td>
</tr>
</tbody>
</table>

Source: (Agro Business Park, 2011)

Figure 7 details the amount of livestock manure by type across the EU-27.
Retention and excretion of nutrients varies widely between different animal types, and are significant factors in determining the overall level of nutrient efficiency in livestock production. Efficiency can be defined as the proportion of N contained in manure that can be recovered by crops over more than one season. Characteristics of the manure, such as the amount of nitrogen (i.e. ammonium-N, nitrate-N and uric acid-N) potentially available for uptake by the crop, vary according to the type of manure. For example, slurries and poultry manures contain around 35-70 % of total N, whereas farmyard manures (which includes the material (e.g. straw, wood shavings) that have been used as bedding for animals and have absorbed the faeces and urine) contain only 10-25 % of total N (AEA, 2010a). It was estimated that only 52 % of the N excreted by livestock is recycled as plant nutrient, meaning that 48 % of it is not recovered by plants (AEA, 2010a). The variation of manure N efficiency per livestock according to an EU-wide review conducted in 2010 is presented in Annex 1.

The amount of available phosphorus contained in manure varies according to the type of livestock (for further information see section 2.2.1.2). For example, a recent UK guidance booklet for farmers indicated that short-term availability of phosphorus contained in the manure is lower than with inorganic fertilisers. Whilst lower, it nevertheless represents 60 % of the phosphorus available in cattle, pig and poultry farmyard manure and 50 % of the phosphorus available in slurry (ADAS Gleadthorpe Research Centre, 2001a). Longer-term the effectiveness reaches 100 % for all livestock manures13 (ADAS Gleadthorpe Research Centre, 2001a). In the Netherlands, an indicative value for the working coefficient14 of P for any type of animal manure is 1.0 with repeated annual applications. For single applications of farmyard manure, the P working coefficient can be assumed to be 0.5. (FAO, 1996)

Table 2 presents a summary of some of the data available for excretion rates of nitrogen and phosphorus for different livestock. The table highlights nitrogen variations within a single livestock category (note differences in units for certain sources). Information on typical content of nutrients N, P and K in different type of manure is presented in Annex 2.

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13 The concept of effectiveness refers to the share of nutrient contained in the manure available for uptake by the crops (ADAS Gleadthorpe Research Centre, 2001a).

14 The working coefficient represents how much nutrient from a mineral fertiliser is used to produce the same yield as an application of animal manure which contains 100 kg of the nutrient
### Table 2 – Mean N and P Excretion factors for different livestock

<table>
<thead>
<tr>
<th>Species</th>
<th>Mean N Excretion (kg N per animal per year) in draft BREF on Intensive Livestock 2013</th>
<th>N excretion (kg N AAP-1 a-115) in EMEP Guidebook, 4B</th>
<th>Mean P Excretion (kg P per animal per year) in draft BREF on Intensive Livestock 2013</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fattening pigs</td>
<td>11.79 (DE, annual excretion: kg/place) 10.8-12.96 (NL, annual excretion: kg/place) 15.4 (IT, annual excretion: kg/place)</td>
<td>12.1</td>
<td>0.73</td>
</tr>
<tr>
<td>Sows</td>
<td>26.2 – 29.5 (NL case study) depending on nitrogen content in feed and number of weaned piglets; 20.4 (biphasic feeding) – 24.6 (standard feeding) in FR example</td>
<td>34.5</td>
<td>4.04</td>
</tr>
<tr>
<td>Dairy Cows</td>
<td>N/A</td>
<td>105</td>
<td>N/A</td>
</tr>
<tr>
<td>Non-dairy cows</td>
<td>N/A</td>
<td>41</td>
<td>N/A</td>
</tr>
<tr>
<td>Broilers</td>
<td>0.24 (IR, kg/bird place/yr.) 0.36 (IT, kg/bird place/yr.)</td>
<td>0.36</td>
<td>0.09 (IR, kg/bird place/yr.)</td>
</tr>
<tr>
<td>Laying Hens</td>
<td>0.56 (IR, kg/bird place/yr.) 0.66 (IT, kg/bird place/yr.)</td>
<td>0.77</td>
<td>0.12 (IR, kg/bird place/yr.)</td>
</tr>
<tr>
<td>Ducks</td>
<td>N/A</td>
<td>1.26</td>
<td>N/A</td>
</tr>
<tr>
<td>Turkeys</td>
<td>1 (IR kg/bird place/yr.)</td>
<td>1.64</td>
<td>0.4 (IR kg/bird place/yr.)</td>
</tr>
</tbody>
</table>

Source: draft BREF on Intensive Livestock refers to annual excretion in kg/place for finishing pigs (Joint Research Centre, 2013a).

**Breeding system**

Livestock production generally consists of transforming feed into meat, milk or eggs. The breeding systems and animal housing systems chosen have a significant impact on nutrient efficiency and emissions from livestock production. As outlined previously, animals can be kept indoors, outdoors, or spend a proportion of their time indoors/outdoors. Livestock housing systems vary according to the livestock, e.g. for poultry rearing cage and non-cage systems are available. Other variables include the systems for removing and storing (internally) the produced manure (the latter is addressed further in section 2.2.1.3 below).

The type of housing used has an impact on the amount of emissions produced. Non-cage systems may lead to higher emissions of dust due to the presence of litter material and the increased animal activity. However, this impact can be mitigated by the frequent removal of manure from the housing (Joint Research Centre, 2013a). The amount and type of manure produced during livestock production is also influenced by the breeding system used for livestock. This includes:

- **The housing type used for livestock.** For example, intensive livestock rearing systems using concrete or slats produce a liquid manure (slurry), whereas livestock raised in covered straw yards produce litter-based farmyard manures and solid manures (slurry separators can be used to separate slurry into a very liquid fraction and a solid fraction);

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15 AAP represents the number of animals of a particular category that are present, on average, within the year. If the number of animals present on a particular day does not change over the year, a census of the animals present on a particular day will give the AAP. However, if the number of animals present varies over the year, e.g. because of seasonal production cycles, it may be more accurate to base the AAP on a census of the number of animal places.
- **The length of time spent by livestock indoors versus outdoors.** Keeping animals indoors (e.g. poultry, pig, cattle in some systems) is linked to higher loss of ammonia, due to the higher temperatures which increase the loss of nutrients (Kaveolis, 2006). However, keeping animals housed enables the control of the feed intake and the collection of excrement and manure and a more efficient application of this to crops. Weekly removal from the manure belts and transfer to covered storages have been found to reduce nutrient losses and ammonia emissions by 50 % compared with bi-weekly removal (Oenema, et al., 2012). Livestock that is reared mostly outdoors (e.g. sheep, cattle in some systems), tend to leave their excrements in areas where the animals concentrate, leading to very low nutrient utilisation and crop uptake (EEA, 2000). This also results in less manure produced indoors that can possibly be used to fertilise crops. However, well managed grassland, e.g. through rotational grazing, allow recycling of the excreted nutrients by fertilising grass while avoiding local excess of nutrients that may lead to nutrient losses (Cellier & Peyraud, 2012);

- **The addition of bedding materials (litter) or water to manure;**

- **The number and type of animals present.** Greater concentrations of animals in smaller spaces, e.g. as in some poultry systems, and some cattle rearing systems, leads to higher temperatures in the fouling areas, which are linked to greater relative loss of ammonia (Kaveolis, 2006); and

- **The livestock feeding regime.** Changes in animal feeding have a significant impact on ammonia emissions from all stages of the animal manure management chain, and can decrease N\textsubscript{2}O emissions and odour (see section 2.2.1.2).

- **The intensity of livestock raising practices also influences the nutrient content of manure.** For example for dairy cows, as the yield of milk production increases, the quantity of manure excreted and the nutrients contained in the manure increase. For instance, in Scotland, guidance issued by the authorities to farmers estimates that dairy cows with up to 6 000 L of annual milk yield produce weekly 0.29 m\textsuperscript{3} of manure while dairy cows with over 9 000 L annual milk yield produce weekly 0.45 m\textsuperscript{3} of manure (The Scottish Government, 2014). However, the quantity of nitrogen excreted per litre of milk is lower than for low producing cows (FAO, 2006; Cellier & Peyraud, 2012; Weiss, et al., 2007).

<table>
<thead>
<tr>
<th>Milk yields in dairy cows</th>
<th>Volume of manure excreted per week (m\textsuperscript{3})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dairy cow over 9 000 litres milk yield</td>
<td>0.45</td>
</tr>
<tr>
<td>Dairy cow, 6 000 to 9 000 litre milk yield</td>
<td>0.37</td>
</tr>
<tr>
<td>Dairy cow, up to 6 000 litre milk yield</td>
<td>0.29</td>
</tr>
</tbody>
</table>

Source: (The Scottish Government, 2014)

### 2.2.1.2 Feeding practices

Livestock retain between 15 to 45 % of the nitrogen intake and 20 to 40 % of phosphorus intake (Dourmad, et al., 2009) (Ferket, et al., 2002). Nutrients that are not assimilated by animals are excreted through manure and urine. The more nutrients contained in manure and urine, the more it may lead to increasing nitrogen emissions during storage in the form of ammonia and nitrous oxide, and, if it comes in contact with soil, to increasing nutrient surpluses and the associated risk of transfer into other compartments (Westendorf & Williams, 2014).

Thus, adjusting feeding according to the livestock’s growth stage and physiological conditions influences nutrients excretion and thus its potential impacts (FAO, 2013; Dourmad, et al., 2009). Several feeding
practices that improve the absorption of nutrients have been identified (Cellier & Peyraud, 2012; Dourmad, et al., 2009).

Inadequate adjustment of the quantity of feed provided may cause the over-nutrition or undernutrition of animals. While the effect of undernutrition on nutrient excretion remains controversial, excess nutrients introduced through over-nutrition contribute to higher nutrient content in manure: first, feed consumed but exceeding animal needs is not assimilated and results in a higher nutrient excretion rate; then part of the feed may not be consumed by the livestock and may end up in litter (Ferket, et al., 2002) (Ghebremichael, et al., 2007).

Diet composition, in particular protein content, also highly influences livestock nutrient excretions. High protein intake increases ammonia and phosphate concentration in faeces (Iyathurai, 2007; Cole, et al., 2005; FAO, 2006). Manure from cattle fed with high protein fodder degasses a higher amount of ammonia (Cherrier, et al., 2014). For instance, for pigs and broilers, every percentage unit increase in dietary crude protein (CP) results in an increase in ammonia concentration and emissions by about 10 % (Relandeau & Le Bellego, 2004; Relandeau, et al., 2000). High protein intake also increases drinking water requirements of livestock for growth and lactation and hence, the volume of slurry and the nitrogen excretions (Ward & McKague, 2007; Rasby & Walz, 2011).

Moreover, the digestibility of protein, i.e. the ratio of proteins degradable by the livestock’s digestive system, influences the share of nutrients assimilated by livestock and thus the share of nutrients excreted (Ferket, et al., 2002). Thus, increasing the ratio of degradable proteins in cattle feed can reduce nitrogen excretion by 15 to 30 %. For instance, diets of dairy cows that are only based on maize silage result in reduction of nitrogen excretion by 25 % compared to diets based on both maize silage and grazing (FAO, 2013). For pig and poultry, increasing the ratio of digestible proteins in feed can reduce phosphorus excretion by 25 to 35 % (FAO, 2006).

To a lesser extent, sugar content also affects nutrient excretion. First experimental findings suggested that high-sugar forage grasses reduce nitrogen content in urine and manure. However, further research is required to support this hypothesis (FAO, 2013).

Lastly, another way of influencing protein digestibility consists in optimising feed structure, in particular feed size, through feed manufacturing technology. For instance, fine grinding and pelleting feed increase protein digestibility (Ferket, et al., 2002).

In a more indirect way, practices related to the production of the animal feed crops should also be mentioned. For instance, in maize and soy production for the production of high protein fodder N2O is emitted due to heavily compacted soils (anoxic conditions) and a high input of fertilisers. As nutrients for feed are mainly imported16, the usage of high protein fodder adds to the imbalance of nutrients in certain locations and thus to the disruption of local nutrient cycles.

Additives can be used to increase nutrient assimilation either directly by degrading feed component or indirectly by modulating microbial flora responsible for nutrient absorption (Ferket, et al., 2002). Phosphorus for instance is mostly present in feed as phytate which is mostly indigestible and thus excreted. Addition of enzymes capable of degrading phytate can allow phosphorus to be better assimilated and thus less excreted by livestock. For pigs, an addition of phytate may reduce phosphorus excretion by 40 to 50 % (Dourmad, et al., 2009; FAO, 2006). Furthermore, some "essential" amino acids

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16 Imported amounts for EU: 9 Mio MT maize, 12.3 Mio MT (soy oilseed), 20.6 Mio MT (soy bean meal). Produced amounts for EU: 64.9 Mio MT maize, 1 Mio MT (soy oilseed), 9.8 Mio MT (soy bean meal) (United States Department of Agriculture 2013, retrieved from Index Mundi 2014)
(constitutive elements of proteins), which can be limiting factors for livestock growth, maintenance and productivity, may lack due to low protein digestibility. Addition of amino acids to animal diets can simultaneously offset such low amino acid availability and prevent excretion of non-degradable protein and thus nitrogen excretion (since proteins contain nitrogen). Other additives commonly used to enhance nutrient digestibility are tannins. However, their expected effects are controversial, ranging from beneficial to noxious on animals’ health (FAO, 2013; Makkar, 2003; Min, et al., 2003).

**Multiphase feeding.** which consists in distributing the required amount of food to animals several times per day instead of feeding them once or twice, also decreases nitrogen excretion (Bourdon, et al., 1995). Experiments carried out on dairy cows evidenced that feeding them twice a day instead of once a day resulted in a higher food intake and thus a lower un consumed amount of food potentially ending up in litter, while significantly increasing milk yield (Sova, et al., 2013). For growing pigs, nitrogen excretion can be reduced by 5 to 8 % by increasing the number of feed phases from two to four, while both nitrogen and phosphorus excretions of grown-up pigs can be reduced by 13 % by increasing the number of feed phase from one to two (Ferket, et al., 2002). Combining multiphase feeding to protein content adjusted to livestock needs leads to an even higher decrease. For instance for pigs, the combination of multiphase feeding and balanced amino acid content may reduce nitrogen emission from 30 % to 50 % compared to a unique phase feeding without protein content adjustment (Dourmad, et al., 2009). In addition, implementing phase feeding decreases costs as a result of the need for less feed to be provided.

In addition, animal genetic improvement can result in a better feed conversion ratio which improves nutrient digestibility and limits excretion. For instance, high producing cows that produce an additional 1 000 L of milk per lactation period (typically one year) excrete 5 % less nitrogen per litre of milk. However, in absolute amounts, they produce more manure and excrete more nutrients per head (FAO, 2006; Cellier & Peyraud, 2012; Weiss, et al., 2007).

Although adjusting feeding practices to livestock needs may allow minimising nutrient excretion, it is difficult to estimate accurately these nutritional requirements since they are moving targets depending on many factors including genetic characteristics of animals (Ferket, et al., 2002)(Valde, et al., 2007). As mentioned above, some of feeding practices are suspected of causing side effects on animal’s growth, production and health (Dourmad, et al., 2009). For instance, the mastitis infection rate of dairy cows might be associated with the silage ratio in the feeding ration due to a possible impact on immune function (Valde, et al., 2007). Lastly, optimising feeding practices might imply extra costs. For instance, care must be taken to avoid that the purchase of additives instead of using feed produced on-site outweighs the benefits brought by the implementation of the measures (FAO, 2006) (Dourmad, et al., 2009).

### 2.2.1.3 Storage of manure: coverage, storage facilities

Under the Nitrates Directive, manure in the Nitrate Vulnerable Zones can only be spread during specific periods (e.g. when the crop grows and requires nutrients) and circumstances may prevent the application of manure (e.g. water logged or frozen soils) (see section 2.2.2.1). Outside the Nitrate Vulnerable Zones, farmers may implement these requirements on a voluntary basis. Storage facilities are built in order to store the manure until the handling is possible and/or allowed. Regulations and guidance are available in most Member States on the design and the building material for the storage facility. In addition, the Nitrates Directive specifies elements related to the storage of manure. In particular, Action Programmes developed for and mandatory in the Nitrate Vulnerable Zones must include rules relating to the capacity of storage vessels for livestock manure. This capacity must exceed that required for storage throughout the longest period during which land application in the NVZs is
prohibited\textsuperscript{17}. In relation to manure storage, these cover the capacity and construction of storage vessels for livestock manure, including measures to prevent water pollution by run-off and seepage into the groundwater and surface water of liquids containing livestock manure and effluents from stored plant materials such as silage (Annex II, Nitrates Directive). Finally, the best available technique reference document (BREF) on Intensive Rearing of Pigs and Poultry adopted under the Industrial Emissions Directive also provides guidance and best practices in relation to the storage of manure, however it is important to note that the BREF is limited to pig and poultry rearing above a certain threshold.

The storage of manure has an impact on the nutrients content and their efficiency. It is important to use appropriate storage techniques to ensure that nutrients are not lost during the storage period and that the manure retains as much of its nutrients as possible which will then, in turn, be available for uptake by crops when spread onto the soil.

### Storage material

The floor of a storage tank for solid manure is usually made of concrete, with or without side walls. The walls prevent rainwater and slurry from leaking. The tanks are commonly made of concrete or steel panels above or below ground. For slurry and liquid manures, storage vessels of different materials (which should be leak proof) are used.

The temporary storage in field of solid manure is allowed in most Member States. Nonetheless, specific prevention measures should be required in particular regarding the storage duration, the distance from water bodies and urban areas, and the presence of other requirements reducing the risk of nutrient losses. For example, some Member States require that stacks are located in different places each year, to avoid leaching or, alternatively, impermeable floors, covers or drainage systems must be used. The distances from watercourses differ in Member States and are set taking into account climatic and geological conditions aiming to ensure adequate protection of water bodies from agricultural run-off as well as to protect drinking water abstraction sources. Some Member States require that the heaps are covered to avoid run-off in case of rainwater penetrating the heap, evaporation of ammonia and odours.

As a general rule, shorter storage times result in the nitrogen content of the manure decreasing less. Furthermore, longer storage times means that organic nitrogen is transformed into ammoniacal N which can be more easily used by the plant but also lead to more ammonia emissions to air. This is due to the natural microbiological activity conducted during the storage phase, which degrades organic matter, mineralises nutrients and increases the concentration of soluble components in the solid portion of the manure (Agro Business Park, 2011).

### Storage capacity

It is important that there is enough storage capacity to ensure that all the manure produced on the farm is stored appropriately until it can be further utilised (e.g. spread, processed) as fertiliser at the correct times, thereby improving nutrient efficiency (as application at incorrect times leads to increased nutrient leaching). In application of the Nitrates Directive, Member States must set the mandatory rules regarding

\textsuperscript{17}Except where it can be demonstrated to the competent authority that any quantity of manure in excess of the actual storage capacity will be disposed of in a manner which will not cause harm to the environment (Annex III, Nitrates Directive).
the required storage capacity (sufficient for storing manure for more than the longest period when land application is prohibited) in the designated NVZs. Furthermore, the Codes of Good Agricultural Practice developed by Member States (mandatory in the NVZs and voluntary in the rest of the territory) should include provisions regarding the periods when the land application of fertilisers is inappropriate and the storage capacity. As part of ensuring compliance with the Nitrates Directive, Member States are required to ensure at least a minimum level of manure storage capacity. Some Member States set a single storage period for all types of manure while other Member States specify different storage periods depending on the type of manure and other elements (Cherrier, et al., 2014).

**Manure coverage**

The *covering of the stored manure* can reduce the availability of oxygen, and the resulting ammonia, N\(_2\)O, and methane emissions. If the manure is left uncovered, the air exchange and the elevated temperature due to the aerobic decomposition lead to losses of ammoniacal nitrogen (referred to as total ammonia nitrogen, or TAN) from solid manure which is a further source of ammonia emissions (Joint Research Centre, 2013a). In addition, covering manure reduces the risk of nitrate leaching and run-off due to precipitation, e.g. for field heaps and solid manure stored on platforms. In the case of heaps, covering materials can be peat, sawdust, wood chips or plastic. Dry peat and sawdust absorb the rainwater, however straw prevents the natural crust from forming on the surface of the manure which in turn can limit nitrogen volatilisation (Joint Research Centre, 2013a). Considering that the extraction of peat is a source of high CO\(_2\) emissions and a non-renewable resource, as such it is not the most appropriate cover to use (Joint Research Centre, 2013a).

It is noteworthy however, that the application of simple roofs to litter heaps may paradoxically lead to an increase of ammonia losses compared to conventional open air storage by 45-60 % (Joint Research Centre, 2013a). This is explained by the fact that the manure surface remains porous, allowing ammonia to diffuse out of the whole heap, whereas conventional heaps exposed to rainfall create a natural crust that is a barrier to ammonia loss. Data has been collected in the UK on the evolution of the P content of manure associated with different types of storage. The total P concentration and loss in leachate from uncovered swine farmyard manure and poultry manure were found to increase slightly during the storage period (from 100 mg/L after 6 months storage to 200 mg/L after 12 months storage for farmyard manures (ADAS Gleadthorpe Research Centre, 2011). It also found that the concentrations and losses of phosphorus decrease over time from manure stored as a covered heap. However, it concludes that there were no strong or consistent patterns in leakage of P from stored manure, although it was noted that covering poultry manure heaps led to reductions of P losses around 8-fold (ADAS Gleadthorpe Research Centre, 2011). The same outcomes were recorded for losses of K. The study concluded that there were no clear patterns in losses of K during the storage of pig and poultry manure. It noted that K losses from uncovered pig manures were substantially more important than those from the uncovered poultry manure heaps (from 13-37 % to 1-3 %). Furthermore, covering poultry manure heaps led to a reduction in K losses around 9 fold (ADAS Gleadthorpe Research Centre, 2011).

In the case of open slurry stores, such as tanks or lagoons, installation of a rigid, plastic or floating flexible cover can reduce ammonia emissions by 80 %, 60 % and 40 % respectively. Covering slurry stores would also reduce CO\(_2\), methane and odour emissions. In an open slurry store, ammonia will volatilise from the surface while being replenished from the lower levels. Emitted ammonia will be removed by the natural air movement and replaced with a lower ammonia concentration. Therefore, installation of a cover over an open slurry store would lead to a higher ammonia concentration in the enclosed space. This higher concentration will reduce further NH\(_3\) emissions from the slurry, so the overall emission rate will decline. However, ammonia emissions, NO\(_x\) (plus ammonium and nitrite) leaching losses and direct and indirect N\(_2\)O emissions following manure spreading might be increased due to the greater readily available N (NH\(_4\)) in slurry (ADAS, 2011).
Control of manure pH and temperature

The pH of the solid manure is important as a higher pH can lead to increased losses of nutrients, especially nitrogen. Indeed, higher pH increases loss of nitrogen through volatilisation into ammonia. The temperature of the slurry tends to match the ambient temperature, with a higher temperature resulting in increased emissions. As a result emptying the slurry store in spring, so that the minimum is stored during the summer, can be a way to avoid emissions. Nonetheless, for manure, inefficient aeration can induce anaerobic fermentation which results in methane emissions and nitrification/denitrification and thus contributing to climate change and odour.

The temperature affects this microbiological activity and accelerates the degradation of the organic matter. This is particularly relevant in cases where the manure is stored in pits below animal housing where the temperature is close to the temperature in the animal houses. Ammonia emissions are increased by passive composting, which is facilitated by access to oxygen. As a result, the smaller the ratio between the surface and the volume of the storage is, the lower the ammonia and odour emissions are (Joint Research Centre, 2013a).

2.2.2 Role of crop production (crops and grass-crops)

2.2.2.1 Fertiliser management

Nutrient requirements of plants

Nutrient requirements of plants are the reason for fertilisation in general. They vary in accordance with the specific cultivated crop, as well as other conditions (soil, climate, etc.). The nutrient requirements vary with the development stage of the crop, as the same quantity of nutrients is not required at the beginning of the growth or at the ripening stage for example. Generally, plant development occurs in stages. Firstly, the seed develops into a plant. The growth of stem and leaf requires different nutrients. After the plant is fully developed, the plant needs nutrients to grow the fruit. This is also dependent on the plant. Most cereal crops reallocate nutrients from leaves to seeds during grain filling and thus do not need additional nutrients. Different crops also have different nutrient needs. For example legumes generally do not need any addition of nitrogen, though all other crops do. To minimise the nutrient loss that would occur when fertiliser is applied at once, split application\(^\text{18}\) is a common farming practice. Manure is often used for the first application. As manure needs to be incorporated and to mineralise before it can be absorbed, it is not possible for all crops to use the manure immediately. Therefore, mineral fertilisers are usually used for later applications.

Plant growth depends on temperature. In periods of high temperature, the decomposition of the soil organic matter (SOM) is accelerated, which increase the release of nutrients available for plant growth. Minimum temperatures for plant growth and development vary between species. For wheat, the literature reports different thresholds for plant development, 5°C being the highest (Porter & Gawith, 1999). In periods of low temperatures, the fertilisers cannot be taken up by plants. That said, a definition

\(^{18}\) Split application is the process of matching nitrogen supply for a pre-established target yield and a given level of soil moisture, and then supplying the remaining nitrogen as moisture conditions improve. Splitting the dose of N fertiliser between more than one occasion during spring and summer can benefit the environment and farm finances. It also reduces the amount of fertiliser, as well as the risk of N leaching due to heavy rains right after spring N fertilisation. If other growth conditions are optimal, split N application can result in yield of better quality and higher quantity. (http://www.balticdeal.eu/measure/split-application-of-nitrogen-n/)
is vague as different plants have different minimum temperatures for growth. The Nitrates Directive requires the establishment of rules on the periods during which the application fertilisers is prohibited\textsuperscript{19}.

Matching the uptake capacity of the crop with the amount of fertilisers applied is important to avoid nitrate (NO\textsubscript{3}\textsuperscript{−}) and phosphorus (PO\textsubscript{4}\textsuperscript{3−}) that are not assimilated by plants leaching to water and resulting potentially in emissions of ammonia (NH\textsubscript{3}) and nitrous oxide (N\textsubscript{2}O) to air. The consideration of nutrient crop needs becomes even more important when manure is used as fertiliser. Indeed, prompt incorporation is required in some Member States to reduce ammonia emissions, and to increase the quantity of nutrients available in soils. Excessive nutrient input in the soil may result in a surplus of nutrients of one type (e.g. N or P). As plant uptake mechanisms are not specific to the type of nutrient, a surplus of a single nutrient might cause inhibited growth and malfunction of plant organs, similar to plant symptoms resulting from a lack of nutrients.

**Type of nutrients as provided by fertilisers**

Fertilisers can be either organic (e.g. manure, stemming from livestock production)\textsuperscript{20} or inorganic (i.e. mineral). Both organic and inorganic fertilisers can provide a range of nutrients to aid plant growth (nitrogen, phosphorus and potassium) in different ratios.

**Nitrogen** is provided as nitrate (NO\textsubscript{3}) and ammonium (NH\textsubscript{4}+), and is contained in both organic and inorganic fertilisers. Another form of nitrogen is urea (CO(NH\textsubscript{2})\textsubscript{2}) that both types of fertilisers contain. Using the enzyme urease, bacteria transform urea to ammonium in the soil. Organic fertilisers provide nitrogen foremost in the form of organically bound nitrogen that has to be mineralised by bacteria and fungi before plants can use it. The provision of both nitrate and ammonium at the same time keeps the pH level in the plant and the soil stable. The uptake of ammonium requires a high turnover of carbohydrates in the plant to detox the ammonia which is harmful to plant metabolism. Plant nutrition high in ammonia results in plants with high water content that are prone to pests (Schulze, et al., 2002). The provision of both nutrients allows a balanced nutrient and carbon budget for the plant. However, due to the chemical properties of nitrate and the ease with which it dissolves in water, nitrates are therefore easily leached into the groundwater. On the other hand, ammonium can be stored and retained in the soil by binding it with clay minerals or organic matter. To limit a loss of nitrogen to the environment, the Nitrates Directive, in Annex III, rules on the limitation of land application of fertilisers, based on the principle of balanced fertilisation and sets the maximum amount of 170 kg N/ha/year from livestock manure, within the Nitrates Vulnerable Zones (NVZ).

**Phosphorus** is contained in inorganic fertilisers as phosphate (PO\textsubscript{4}\textsuperscript{3−}). It is also contained in organic fertilisers such as animal manure and slurry and, to a lesser extent, compost and sludge (Bomans, et al., 2005)\textsuperscript{21}. In the soil, the pH determines the occurrence of phosphate available to plants, as mineral phosphate is a difficult soluble salt whose solubility equilibrium changes with the amount of protons in solutions (this is the chemical definition of pH). Phosphate is retained in the soil by binding with clay

\textsuperscript{19} The period during which application is prohibited are specified by Member States and depend on factors such as fertiliser type, crop type and climatic conditions.

\textsuperscript{20} There are other sources of organic fertilisers such as treated sewage sludge, composts and industrial wastes from the food processing industry, however this study will not focus on these types of organic fertilisers.

\textsuperscript{21} This report highlighted that the contribution of non-manure organic fertiliser to the input of phosphorus to agricultural land remained low and limited to local use due to high transportation and application costs.
minerals or organic matter. The loss of phosphate in the agricultural mineral cycle mainly occurs through soil erosion and run-off rather than leaching.

Some Member States provide requirements regarding the amount of organic phosphorus fertiliser that can be applied. The amount of recommended phosphorus fertiliser greatly depends on the particular soil’s P-fixing properties because a portion of the applied fertiliser may become inaccessible to plants due to quick immobilisation of the phosphate anions in the soil through precipitation and absorption reactions (Smit, et al., 2009). Phosphorus fertilisers applied to the soil in quantities that exceed the uptake capacity of plants can result in P pools in the soil and may result in environmental impacts. Soils with a low P-fertility are soils that have naturally few soil components that are a source of phosphate. Such soils need much more fertilisation than plants would take up. On the contrary, soils with a high P-fertility naturally provide phosphate to plants, thus the P-fertilisation rate should be lower. In the past the thought prevailed that phosphorus was irreversibly fixed in the soil. Therefore, high application levels were common practice to increase the available amount and the potential for uptake by crops in order to meet the aims of high-yield industrial crop production. Long term experiments have shown that the amount of phosphorus applied in excess to plant needs is in fact reversible and may become available for plant uptake in the future (Smit, et al., 2009). Considering these findings, the P-fertility of a soil should be reviewed periodically to determine whether the available P in the soil could be taken up by the crop or whether additional phosphorus is needed to avoid yield depressions.

Inorganic fertilisers and organic fertilisers are sources of potassium. Inorganic fertilisers contain the nutrient in the form of potassium sulphate and potassium chloride. Potassium, once dissolved and present as a cation, is absorbed into clay minerals.

In inorganic fertilisers, the three nutrients N, P and K are often provided in combinations of varying ratios, depending on the compounds that they feature. For example, fertilisers containing a combination of nutrients are diammonium phosphate, ammonium nitrate, ammonium sulphate, urea, and potash nitrate (Shakhashiri, Vitosh 1996). Similarly, in organic fertilisers such as animal manure, the ratio of N, P and K depends on the livestock types, the type of manure and other factors affecting the nutrient intake of the animals (e.g. feed). The application of different types of manure requires different fertilisation management strategies to avoid leaching. For example, applying slurry leads to a high risk of nutrient leaching due to the high concentration of nutrients and low C/N ratio in slurry; whereas application of solid manure may require the addition of an inorganic fertiliser to complete the nutrient requirements (solid manure has a more balanced C/N ratio but have insufficient lower concentration of nutrients) (see section 2.2.1).

The soil and climate conditions must be taken into account when deciding on the type of nitrogen and potassium fertilisation. For example, soils with a high content of sand, and thus higher porosity, tend to have a high leaching rate.

Application methods for fertilisers

For spreading of solid fertilisers, manure and mineral fertilisers, several techniques are available (Joint Research Centre, 2013a). The type of the device influences the performance in terms of accuracy and even spread distribution. The application of liquid fertiliser, mainly slurry in Europe, is done by sprinkling, spraying, or injecting the liquid in the soil. The application of solid manure and slurry can involve incorporation. Injection and incorporation limit the risks of run-off for all nutrients. By reducing the contact area of the fertiliser with the ambient air, it also reduces volatilisation. However, it may increase the risk of leaching in case the fertiliser is injected below the root depth.
**Inorganic fertilisers**

For the application of mineral fertilisers, high precision is required as they contain highly concentrated nutrients. The precision of fertiliser application is dependent on a constant driving speed of the tractor and a constant rotation frequency of the device (Walig, 2010). Wet conditions also affect fertiliser application since solid fertiliser salts in which the nutrients occur dissolve when they come into contact with water.

**Organic fertilisers**

The selection of the application technique depends on the type of fertilisers as well as the type of soil, the slope and the use of the land on which it will be spread (i.e. arable land or grassland). For example, the deep injection of slurry is applicable to arable land, however, it requires a high draught force to inject the fertiliser, which may reduce the herbage yield and damage the crops (Huijsmans, 2003; Joint Research Centre, 2013). The performance of the equipment used to spread the fertiliser can be affected by slopes (Joint Research Centre, 2013a). For example, deep injection with closed slots cannot be done on land with significant slopes as the technique requires heavy equipment that cannot operate in these conditions (Joint Research Centre, 2013).

The application of solid organic fertilisers (e.g. solid manure, litter based farmyard manure) can be done by using one of the three main spreaders described below:

- **Rotaspreader**: a side discharge spreader fitted with a cylindrical body and a shaft. The shaft is fitted with flails running along the centre of the cylinder; as the rotor spins, the flails throw the solid fertiliser to the side of the vehicle. This technique does not allow an accurate discharge rate of the fertiliser or a high level of precision of the application.

- **Rear discharge spreader**: a trailer body fitted with a moving floor (or similar mechanism) that discharges the solid fertiliser to the rear of the spreader. This technique provides a high accuracy of the discharge rate and a high level of precision of the application.

- **Dual-purpose spreader**: a side discharge spreader (rotaspreader) characterised by its capacity to discharge both slurry and solid manure. Similar to the rotaspreader, a dual-purpose spreader does not allow an accurate discharge rate of the fertiliser or a high level of precision of the application.

The spreading of solid organic fertilisers can be supplemented by incorporation which reduces nutrient losses by burying the fertiliser under the soil (Joint Research Centre, 2013a). The incorporation can be done with a plough or other equipment such as rotor harrows or cultivators. The choice of the equipment depends on the soil type and conditions, for example the plough is mainly applicable for solid manures incorporated into arable soils. Ploughs are more efficient in incorporating the fertilisers but slower than

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22 The precision of fertiliser application refers to the accuracy of the targeted and the achieved amount of fertiliser applied on a certain area. The targeted amount of fertiliser is calculated according the crop needs and considering the other possible nutrients inputs such as atmospheric nitrogen deposition or the nutrient residues in soil.

23 Picture: (ADAS, Institute of Grassland and Environmental Research and Silsoe Research Institute, 2001)

24 Picture: (ADAS, Institute of Grassland and Environmental Research and Silsoe Research Institute, 2001)
rotor harrows. As a result the fertiliser remains uncovered on the soil for a longer time, which leads to increased nitrogen volatilisation. Whilst incorporation during or soon after spreading is considered a good practice, surface spreading without incorporation or with late incorporation is often performed.

For the application of liquid organic fertilisers (e.g. slurry), it is necessary to distinguish the transport systems from the distribution systems. There are three main types of slurry transport systems (ADAS Gleadthorpe Research Centre, 2001b):

- **Vacuum tanker**: an air pump is used to suck the slurry into the tanker and to force it out. It can be used for slurry with up to 12 % dry matter content. The air pump does not allow for a high precision in the application rate.

- **Pumped tanker**: a slurry pump is used to get the slurry into and out of the tanker. The pump is either centrifugal or positive displacement. The latter allows for more accuracy in the application rate; however it requires that the slurry is separated or chopped beforehand. This type of slurry transport system provides a better spreading precision than vacuum tankers, but the pump requires more maintenance, especially the positive displacement one.

- **Umbilical hose**: the slurry is fed from the slurry store by a drag hose to the distribution system that is fitted to a transport vehicle (e.g. tractor). The same pumps as for the pumped tanker are used, raising the same limitations. In addition, the hose can be damaged when dragged on abrasive or flinty ground which limits the application of this technique.

These transport systems can all be equipped with one of the four main slurry distribution systems described below (Joint Research Centre, 2013a).

- **Broadcast spreader** consists of a discharge nozzle and a splash-plate applicator. The slurry is spread over the whole soil surface (“broadcast”) using an inclined splash plates to increase the sideways spread. Broadcast spreading is versatile and can be used on most soils; however, it does not spread in a uniform way. Due to the extensive contact between the slurry and air, the loss of nitrogen incurred by this technique is significant.

- **Band spreader** (‘trailing hose’) slurry is discharged by band spreaders located at or just above the ground level in strips through a series of hanging or trailing pipes. The band spreader is fed with slurry from a single pipe, and relies on the pressure at each of the hose outlets to provide an even distribution. The width is typically 12 – 28 m with about 30 – 50 cm between bands. The technique is applicable to grass and arable land; it can cause damages to crops as the hose drags on the ground. This technique results in nitrogen losses through volatilisation of ammonia due to contact between the slurry and the air.  

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25 Picture: (ADAS, Institute of Grassland and Environmental Research and Silsoe Research Institute, 2001)
• **Trailing shoes** is similar to band spreader with a 'shoe' device added to each hose allowing the slurry to be deposited under the crop and onto the soil. The technique is particularly applicable to grassland. The slurry is placed in narrow bands on the soil surface at 16 – 35 cm spacing. The slurry bands should be covered by the grass canopy so the grass height should be a minimum of 8 cm. The systems are available in a range of widths up to 6 – 16 metres. Due to the fertiliser being applied directly onto the soil the losses of nitrogen through volatilisation are reduced.

• **Injection** consists in inserting the slurry under the soil surface; either in open or closed slot.
  
  - For open slot injection, the injection is either shallow (up to 5 cm deep) or deep (over 15 cm deep). This technique is not applicable on rocky or stony soil, nor on shallow or compacted soils where it is impossible to achieve uniform penetration of the knives or disc coulters to the required working depth. This technique is mainly for use on grassland.

  The spacing between the slots is typically 20 – 30 cm, with a typical working width of 6 m that can reach 9 – 12 m. The application rate must be adjusted so that excessive amounts of slurry do not spill out of the open slots onto the soil surface.

  - The closed slot technique can also be shallow (3 – 5 cm depth) or deep (up to 20 cm). The main difference with the open slot technique is that the slurry is fully covered after injection by closing the slots. Shallow closed-slot injection is more efficient than open-slot for decreasing nitrogen loss through ammonia emissions but it is less widely applicable because of the potential mechanical damage. Deep injectors usually comprise a series of tines that can be fitted with lateral wings, to aid lateral dispersion of slurry in the soil. This means that relatively high application rates can be achieved. Tine spacing is typically 25 – 50 cm, with a working width of 2 – 12 m. Although the abatement of emissions of ammonia to air is high, the technique is restricted to arable land because mechanical damage could decrease yields on grassland. Other limitations include soil depth and the clay and stone content, the slope, and a high draught force requiring a large tractor. In specific circumstances there is a greater risk of nitrogen losses from deep injection as nitrous oxide and nitrates.

Where injection techniques are not possible or unavailable, incorporation techniques (see above) may be used for slurries.


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26 Picture: (ADAS, Institute of Grassland and Environmental Research and Silsoe Research Institute, 2001)

27 Picture: G. Provolo
- **Irrigation** is relevant for the application of liquid organic fertiliser to the soil. It consists of diluting the slurry (less than 3% of dry matter content) to then spread it through water irrigation systems. This technique is called “fertigation”.

The irrigator allows for some accuracy in the application and the slurry mixed to irrigation waters can be applied to grassland or growing crops on arable land. It is worth noting that the duration of the spreading operation is longer for irrigation than other techniques. Also, the high volumes of diluted slurry applied may exceed the infiltration capacity of the soil. The length of the spreading time and the natural limitation due to the capacity of absorption of the soil may lead to higher losses of nitrogen and potassium in the period immediately after spreading.

Other techniques include:

- **Variable rate spreading** has been developed as part of the ‘precision farming’ movement. It involves defining the nutrients needs of specific areas by conducting a survey and a soil analysis map. The data collected are used by the spreading device which adapts the amount spread in accordance with the needs identified. This technique is relatively recent and there is no clear agreement on its benefits.

- Finally, the slurry can be **acidified** before spreading in order to reduce nutrient losses. The acidification of slurry is commonly practiced in a few countries (e.g. Denmark, the Netherlands). By adding acid (usually sulphuric acid), the pH of the slurry is lowered (to around 5.5), and the ammonia volatilisation is limited. The slurry is either acidified in the tank immediately before spreading, or it is continuously acidified in a system directly mounted on the slurry spreader.

The main difference between the fertiliser’s application systems is the placement of the fertiliser once released from the tanker (Figure 8). The techniques delivering the fertiliser the closest to the ground and with the least contact with air result in the lowest losses of nitrogen to volatilisation. However if too much fertiliser and nutrients are applied to the crops, even applications that are close to the ground or incorporated can nevertheless lead to nutrient leaching.
It is important to note that whilst the application techniques are relevant for nitrogen and potassium, they are not as significant in relation to the efficiency of the phosphorus contained in the organic fertiliser (Bomans, et al., 2005).

**Application conditions**

In order to avoid nutrient loss, the quantity of fertiliser applied to the crop must match its requirements. It is also important that the fertiliser is applied evenly and accurately as an uneven application can lead to uneven growth of crops and reduced yields (Fertiliser association, 2012). If too many nutrients are applied to the crops, the excess will not be absorbed and is susceptible to leaching and run-off, potentially causing impacts on water (see section 2.3.3).

When applying fertilisers the following elements have to be taken into account:

- **Slopes**: the application of fertilisers on slopes can increase the potential for run-off and subsequent loss of nutrients. On sloping land it is recommended that fertiliser is applied from top to bottom rather than across slopes in order to limit run-off (Fertiliser association, 2012). Member States have adopted application recommendations or requirements regarding applying fertilisers on areas with slopes. These are mostly general requirements that the risk of run-off is taken into account when applying fertiliser. However some Member States have adopted more detailed requirements such as the prohibiting the application of fertilisers to slopes above specific steepness thresholds.

- **Weather**: application of fertilisers in wet weather conditions is likely to lead to run-off as the rain can wash away nutrients before they reach the crops. Similarly, the application of fertiliser to water-saturated, flooded, frozen or snow-covered ground is inappropriate (Council Directive 91/676/EEC). Furthermore, mineral fertilisers are costly, so an application when there is a risk that the fertiliser would not be used by the crops constitutes an economic loss.

- **Soil**: application on a slippery soil can result in an uneven deposition of the fertiliser. In addition, application on slippery surfaces may pose some safety issues (e.g. the vehicle could slide). Soft soils are more vulnerable to compaction, so on these soils it is recommended to avoid using heavy equipment so as to limit compaction (Joint Research Centre, 2013a). Furthermore, the nutrients already present in the soil need to be taken into account when calculating the nutrient requirements.
• **Buffer zones**: these zones are defined in order to protect surface waters, groundwater sources or nature protected areas that could be affected by run-off from the application of fertilisers. In some Member States, rules are in place to set minimum distances to water bodies.

• **Calibration and maintenance of the equipment**: To ensure an even application, spreaders and pieces of equipment used to apply fertiliser need to be calibrated, regularly maintained and adapted to the fertiliser that is being applied (e.g. fertiliser density). The coefficient of variation (CV) is a measure of the uniformity of the fertiliser spread. A high number reflects a poor uniformity. For granular fertiliser, a CV of less than 15% is considered to be acceptable, however, for slurry or manure less than 25% is acceptable (ADAS Gleadthorpe Research Centre, 2001b). The speed of the spreader must be constant to ensure an even spread. Furthermore it is important that the track of the spreading vehicle does not create a path for run-off to berms or ditches;

• **Timing**: the fertiliser needs to be applied at a time where the nutrients can be taken up by the crops (e.g. in period of crop growth). A high nitrate level in the soil induces germination (Schulze, et al., 2002). Therefore, an early application of fertilisers can lead to quicker germination of crops. However, application should take place under suitable climatic conditions, avoiding for instance periods of high precipitation, which would cause leaching of nutrients. In this context, Member States have established closed periods during which fertilisers cannot be applied. Application of mineral fertilisers in a time outside the vegetation period is not beneficial to crop production and can therefore lead to economic losses, as well as a risk of environmental pollution.

• **Incorporation**: In order to facilitate the uptake of nutrients, many Member States have defined incorporation times for organic fertilisers. For solid manure, a 24 hour incorporation time seems to be typical among Member States, while for slurry it ranges from immediate up to 12 hours. It should be noted that prompt incorporation of manure may result in a potential surplus of nitrates in the soil and increase the risk of leaching unless the nutrient balance has been taken into account.

• **The Nitrates Directive** includes a number of requirements relating to the timing, conditions and quantity of fertilisers which can be applied in Nitrate Vulnerable Zones.

### 2.2.2.2 Soil management

Soil management practices affect soil structure and soil quality in different ways. Soil quality, in turn plays an important role in nutrient cycling because of its influence on nutrient and water retention, and the transformation and transfer of nutrients in soils. Soil management affects the water pathways through and over the soils, including the amount and flow of surface run-off during precipitation.

The key soil management practices affecting nutrient flows include types of tillage, the extent and timing of soil coverage, the types of crop rotation, inclusion of legumes and N-fixing crops, residue management and perennial/permanent crops. The applicability and effect of these measures with regards to nutrient excess depends on the farming system, climate, and soil types.

#### Soil tillage

Tillage involves ploughing the soil before, during, or after planting at varying depths. By disturbing the soil structure, tillage modifies aeration and soil moisture. These in turn affect plant roots and soil organisms and thus influence N turnover in the soil. Increased aeration from tillage also accelerates decomposition of organisms and crop residues in the soil (Baldock & Skjemstad, 2000; Paustian, et al., 2000; Six, et al., 2000; Flynn, et al., 2007). Mineralised N is released through this decomposition, so if N-containing substrates are also applied, surplus amounts of N in the soil can occur if the N available is higher than the rate of plant uptake (Chatskikh & Olesen, 2007). Excess mineralised N can lead to nutrient surplus and cause environmental impacts to soil, water, and air.
Tillage intensification (the frequency and depth of tillage) in arable systems tends to decrease soil organic matter and is related to deterioration in soil structure (e.g. loss of aggregate stability, increased crust formation, increased soil compaction, also due to the use of heavy machinery), which increases run-off and soil erosion, and slows down water infiltration and retention (Gobin, et al., 2011). For example, slaking crust can result from tillage breaking soil particles into smaller aggregates and extreme rainfall or water immersion forming a waterproof, superficial soil layer (Herrick, et al., 2001). This creates anaerobic conditions in the soil that negatively impact aerobic organisms through lack of oxygen and increases the risk of leaching due to reduced plant uptake of nutrients and increased denitrification leading to nitrous oxide gaseous losses (Herrick, et al., 2001; Inglett, et al., 2005). On the other hand, soil tillage affects ammonia volatilisation since both infiltration rate and contact area affect volatilisation. Hence, volatilisation is higher when fertilisers are applied to a compacted soil rather than a cultivated soil. Consequently, soil tillage may improve infiltration and reduce ammonia volatilisation (Hutchings, et al., 2009).

Conventional tillage increases the amount of N$_2$O as well as CO$_2$ released into the atmosphere as compared to reduced tillage and zero tillage (Flynn, et al., 2007; Chatskikh & Olesen, 2007). The increase in CO$_2$ emissions is likely linked to increased turnover rates resulting from tillage, and the N$_2$O emissions are “probably mostly related to changes in soil aeration” (Chatskikh & Olesen, 2007).

The timing of tillage influences its impact: in autumn soil tillage “often leads to enhanced losses of NO$_3$ through leaching, whereas tillage in spring can lead to enhanced uptake of N by the crop, but also to greater N$_2$O emissions” (Chatskikh & Olesen, 2007).

Different forms of conservation tillage, on the other hand aim to reduce the ploughing of the soil prior to planting and minimise pre- and post-planting cultivation for weed control. Different forms of conservation tillage can be applied, with variable effects on soil properties and nutrient cycling, which are also dependent on climatic and soil conditions. Conservation tillage includes, for example, reduced tillage (with harrowing to 8–10 cm) and zero tillage or direct drilling. Reducing tillage increases the presence of residues on soils which facilitates water infiltration, improves soil moisture, and soil biological activity (Gobin, et al., 2011). The objectives of reduced tillage focus primarily on reducing soil erosion, maintaining soil moisture, and increasing soil carbon sequestration. However, reduced/zero tillage can also have an effect on soil compaction and increased N$_2$O, in particular for water-saturated soil since water modifies the mechanical soil properties due to the solubilisation of soil elements (Ball, et al., 1999). Thus, minimum cultivation\textsuperscript{28} is not suitable for very wet autumns or light soils which are prone to capping or crusting (Cuttle, et al., 2007).

The effects of reduced tillage on NO$_3$ leaching are less clear. For example, the effects of zero tillage on nitrate leaching are variable, depending on soil and climatic conditions and N availability in soils (Constantin, et al., 2010). In the case of soil with good aeration, measures such as tillage will have a limited effect. Emissions can be even lower for direct sowing since carbon input can increase nitrogen immobilisation as mentioned above. Hence, considering the initial state of texture, saturation in water and structure, appropriate machinery use will have an important effect on GHG emissions (Rochette, 2011).

The effects of reduced soil erosion from conservation tillage can contribute to reduced phosphorus (P) losses from the field since P is mobilised primarily through the movement of soil particles, although in

\textsuperscript{28} Reduced tillage method using discs and tines to complete shallow cultivation rather than ploughing or inverting the soil, contributing to less mineralisation of soil organic matter and nitrogen through reduced soil disturbance (Cuttle et al., 2007).
the long term the dissolved P can be increased if the P content reaches the absorption capacity if the soil (Cuttle, et al., 2007).

**Soil coverage**

Maintenance of soil coverage means that soils are not left bare (in particular during fallow periods) and thus reduces susceptibility to erosion and surface run-off, as well as nutrient leaching. Plant cover acts as a physical barrier on fields, slowing down and thus reducing run-off of N and P into water bodies as well as preventing leaching into groundwater. Plant cover can also improve soil structure by increasing the amount of organic matter in the topsoil.

Plant cover also enriches the soil with carbon, which is interdependent with both the nitrogen and the phosphorus cycles. Thus, the carbon input results in nitrogen immobilisation by the plant material in the soil. The increased absorption and storage of nitrates in plant roots and incorporated residues is then available for future crops, which reduces the need for mineral fertiliser application, thereby further contributing to reduced leaching of N (Flynn, et al., 2007; Velthof & Kuikman, 2000)

Soil cover includes different types of crops (e.g., annual, perennial, permanent), which can have varying effects on the nutrient cycle. Upon harvest, the nutrients contained in the crop itself are removed from the field, but additional losses from the nutrient cycles can occur if the soil is left bare, such as leaching of nitrogen or soil erosion leading to run-off of N and P. Cover crops planted in the autumn can take up residual nitrates and other nutrients in the soil after the main crop has been harvested which reduces the risk of nitrate leaching over winter (Cuttle, et al., 2007). Soil vegetation cover is especially relevant during the winter months and rainy periods when the risk for erosion and run-off is higher, and especially in areas with excess precipitation and run-off during autumn, winter, and early spring. Fast-growing crops between successive plantings of a main crop can be under-sown below the main crop in spring, although they are more frequently sown in late summer or autumn, immediately following harvest with the purpose of providing soil cover during the winter.

The Codes of Good Agricultural Practice in the Nitrates Directive (Annex II B point 8) mention the need to maintain vegetation that will take up the nitrogen from the soil that would otherwise cause nitrate pollution of water during (rainy) periods.

**Adding legumes / N fixing crops**

Adding legumes or nitrogen-fixing crops into a farming system increases the amount of nitrogen in the farm nutrient balance through biological fixation (Jarvis, et al., 2011). Biological fixation, as part of the nitrogen cycle (see section 2.1.1), involves the fixing of atmospheric nitrogen into the plant roots by symbiotic bacteria or microorganisms so that the nitrogen is available for use by future crops (Galloway, et al., 2004). When the N-fixing plant dies, the fixed nitrogen is released and can be taken up by other crop roots, thereby reducing the need for additional nitrogen fertilisers for the growing crop. Optimally, the release of the N should be timed to coincide with the need for N of the following crop in order to avoid N losses. However, the release is controlled by the decomposition rate, which is influenced by weather conditions and field type, among other factors (Jarvis, et al., 2011).

N-fixing crops, such as clover or vetch, can be planted as a winter cover crop, intercropped with cereals, part of a grass-legume mixture, and so forth depending on the crop rotation (Flynn, et al., 2007). Mixing grass or catch crops with legumes can provide a more gradual release of nitrogen than the rapid release of stand-alone legume crops, which could reduce leaching (Hendrickson, 2009). Early sowing of winter grain legumes can also reduce nitrate leaching losses (Nemecek, et al., 2008). Adding legumes can
also increase the availability of phosphorus in the soil, while grasses can increase potassium (Uphoff, et al., 2006).

**Residue management**

Crop residue management represents another soil management practice which affects the nutrient cycles by incorporating N and P into the soil, as well as forms of carbon (Noack, et al., 2014). Following harvest, stubble or plant material from the previous crop is left on the soil instead of collected, for instance, for fodder or sale for other uses (e.g., maize stover for biofuels). Alternatively, residues from other crops, e.g., straw, can be applied to the soil.

Appropriate residue management can help maintain a balanced C/N ratio based on the various decomposition rates of the residues applied (Natural Resources Conservation Service, 2011). Residues with a wide C/N ratio, such as straw, can slow decomposition and help to immobilise mineral nitrogen (NO$_3^-$, NH$_4^+$) in the soil so that plant-available nutrients that were unused by the previous crop are harnessed and made available for future crops (Flynn, et al., 2007; Jarvis, et al., 2011). However, if the C/N is too high, too much N can be immobilised (as well as P and K) and made unavailable for plants, which induces the use of additional fertilisers. Crop residue application also recycles phosphorus back into the soil, which can affect the release of residual P in the soil and the bio-availability of phosphorus for plant uptake according to different plant types (Smit, et al., 2009; Varinderpal-Singh, et al., 2006). The C/P ratio of the residue influences the process, e.g., a narrow C/P ratio may cause better P uptake of plants, as well as the soil type, timing of incorporation, etc. (Varinderpal-Singh, et al., 2006). Removing plant material from the previous crop not only removes nitrogen and phosphorus, but also potassium which would be incorporated into the soil if the crop residue is maintained (Al-Kaisi, 2012). Potassium levels should be tested and managed appropriately. However, as K becomes fixed in clay soils, soil saturation leads to losses because the K is not released for plant uptake at a fast enough rate. Also, timing of residue application and soil type is important because K is a soluble nutrient that can leach quickly from the crop residue without plant uptake, e.g., in sandy soils (Beegle & Durst, 2001; Bohn, et al., 2002).

Therefore, residue management could reduce application of inorganic fertiliser, which poses potential reductions in direct nutrient losses through leaching, run-off, and denitrification. Recycling nutrients within the farming system rather than removing residues from the field contributes to a balanced nutrient cycle. Soil cover and building up soil organic matter through residue management can also prevent soil erosion, in which soil particles containing N, P, and K are washed away, especially P, as surface water run-off and contribute to eutrophication of water bodies.

On the other hand, depending on the type of residue, C/N ratio, and soil type, N$_2$O emissions could be increased or decreased (Carter, et al., 2012a). Residues which balance the C/N ratio and enhance immobilisation can decrease nitrogen emissions, but they also increase soil density and create anaerobic conditions favourable to N$_2$O emissions (INRA, 2012). Residues which are high in N could cause leakage in the nutrient cycle via atmospheric losses and water leaching, so replacing them with some other residue with a lower N content would be recommended (Flynn, et al., 2007). However, for soils which are well aerated or contain high clay contents which the residues will aerate, residue incorporation and management is mainly beneficial (INRA, 2012).

**Fallow land and crop rotation**

Crop rotation involves planting a sequence of specific crops on the same field, with succeeding crops being of a different genus, species, subspecies, or variety than the previous crop. Additionally, as covered above in the subsection on soil coverage, winter catch crops may be added to the crop rotation
in order to prevent erosion of bare soil exposed to weather (e.g., rain, snow, wind). Often the first crop sequence is used to prepare and regenerate the soil (e.g., legumes or grasslands), and a second sequence benefits from the fertility of the regenerated soil. Crop rotations can vary substantially in the number of crops and the sequence of cropping. In a diverse rotation, deep-rooted crops alternate with shallower, fibrous-rooted species to bring up nutrients from deeper in the soil. Extraction of nutrients from all soil layers is most effective when plant rooting patterns, including root density and root branching at different soil depth, are diverse (BIO Intelligence Service, 2010). This improves soil structure by creating soil pores; enabling the flow of gases, water nutrients and organic compounds in the soil; and allowing water storage and microbial activity.

Crop rotation can thereby contribute to improved soil organic matter by providing plant material to the soil (above-ground residues, roots, and root exudates). Temporary conversion to grassland in the rotation improves soil carbon levels and total soil nitrogen. Continuous leguminous cropping can increase soil carbon storage and total soil nitrogen by up to 20% in the 0-15 cm soil depth compared to rotations that include cereals, which reduces the need for mineral nitrogen fertiliser application as highlighted above (European Commission, 2011).

The quantity of N fertiliser required by and applied to crops affects N leakage in waters. Crops with a long vegetation period that develop a large mass of roots in periods critical for N losses have a higher capacity to use the nitrogen already in the soil and thus capture nutrients that might otherwise be lost from the system, e.g., brassica winter rape, mustard, cabbage, and turnip cover crops (Sustainable Agriculture Research and Education (SARE), 2012). This type of crop as part of a crop rotation would be recommended for reducing nutrient losses issues.

In contrast, monocultures, or producing a single crop or plant species over a large area and for a large number of consecutive years, often requires higher input of mineral N fertilisers (as well as pesticides) and tend to exploit the same root zone so that eventually the available nutrients for plant growth are diminished and root development is reduced (Thierfelder & Wall, n.d.). Moreover crops in monoculture are more vulnerable to pests and diseases and result in a decline in agro-biodiversity. Crop rotations can help to eliminate these issues arising from monocultures. Also, including a fallow period in the crop rotation during which the land is not planted with a nutrient demanding crop gives the land a resting period and opportunity to recharge soil fertility and structure. But if the soil is left bare, it increases the potential for soil erosion and continuing mineralisation could reduce soil organic matter content. Furthermore, no plant can catch the mineralised nutrients, which increases the potential for leaching.

**Perennial / permanent crops**

Establishing perennial/permanent crops in a farm system can reduce losses of nitrogen and phosphorus by maintaining roots in the soil and avoiding soil erosion and leaching (Jarvis, et al., 2011; Salomon & Sundberg, 2012). Livestock production systems often rely on perennial crops for grazing purposes, including leys of grass mixtures and grass-clover. Varying levels of production intensity may actually result in significant N losses (Jarvis, et al., 2011). For instance, adding increasing levels of nitrogen to grazing lands will not produce more outputs, such as milk, meat, and crops, after a certain point. Additionally, nitrogen losses from volatilisation, denitrification, run-off, and leaching continue to rise steadily as nitrogen inputs rise, so achieving an optimal input level so that outputs are maximised but few nutrients are lost would contribute to efficient resource use and closing the mineral cycles (see Figure 9).
Permanent crops in the form of trees or vines (e.g., citrus, olives, or grapes) maintain roots in the soil, but leaving the orchard or vineyard floor bare may lead to poor soil structure (O’Geen, et al., 2006). This could cause problems, such as crusting or compaction, leading to reduced water infiltration, and increased run-off of soil and nutrients. Planting a cover crop in orchards and vineyards helps to reduce soil erosion by providing protection from rain and slowing the flow of water over land, binding soil particles with a root system, contributing to a better soil structure, and increasing soil organic matter (O’Geen, et al., 2006). However, the cover crop may compete with the main permanent crop for nutrients and water, so appropriate crop choice would vary based on the agri-environmental conditions and other soil amendments. In certain situations, reduced tillage may be preferable to orchard and vineyard floor vegetation (Ingels, 1998).

Regarding grass-legume mixes, grass as a form of permanent (cover) crop alongside or within cereal crops, or in orchards/vineyards, increases the immobilisation of nitrogen in the soil. It also reduces soil erosion and organic matter decomposition with benefits to the nutrient cycle (Flynn, et al., 2007). A significant part of the N, P and K cycles involves livestock grazing and digesting then recycling phosphorus back onto the permanent grasslands or arable crops through manure (Smit, et al., 2009). However, some phosphorus contained in the manure is lost to discharge into surface waters or burning of the manure for bioenergy purposes. Additionally, nitrogen can leach or cause gaseous N\textsubscript{2}O emissions, and potassium can leach as well (Frank, 2000). Depending on the management of permanent grasslands, over-grazing may occur and increase the risk of soil erosion, which causes losses of N, P and K. Conversion of land from permanent crops to arable crops also results in phosphorus losses due to soil erosion (Smit, et al., 2009).

Establishing permanent crops, e.g., herbaceous mixture with 40 \% *Lotus corniculatus* L. and 60 \% *Bromus inermis*, L., has been found to be particularly effective for reducing losses of nutrients, as well as soil, organic matter and organic carbon, in mountainous areas. Caution should be exercised when first establishing permanent crops on slopes, however, because slopes are prone to soil erosion and crop failure may result if erosion occurs before the crop roots are firmly established (Flynn, et al., 2007).

### 2.2.2.3 Production of renewable energy on farm

The major link between the production of renewable energy and nutrient use efficiency on farms is the production of biomass for energy. Bioenergy can be produced from almost any kind of biomass: traditional commodities, such as maize, rape, soybeans and sugar crops; a variety of energy crops;
woody biomass from (agro-)forestry and short-rotation coppice (SRC); energy recovery from manure and other types of organic waste, etc.

Agricultural practices to produce many of these commodities that are suitable for bioenergy use are essentially similar to the production methods for the food and feed sector, e.g. maize, rapeseed, soy, sugar beet. Therefore, the potential impacts of nutrient saturation on nutrient transformation and transfers on the farm itself (direct impacts) are very similar. However, there are some specific impacts, some of them indirect, of biomass production for bioenergy:

**Energy crop mix**

According to many studies, including the EEA (EEA, 2013a), the current energy crop mix is not in line with EU objectives to use resources most efficiently. This is due the fact that so called 'first generation transport biofuel' - for example, biodiesel based on oilseed rape that requires comparably high fertiliser use or ethanol from wheat - are not efficient in their resource (and nutrient) use to produce each unit of bioenergy. According to EEA (EEA, 2013a) the most efficient biomass energy use is biomass for heating and electricity as well as advanced biofuels. The latter are also called “second generation” biofuels: cellulosic energy crops, including crop residues and perennial crops such as miscanthus. Impacts and efficiency of each of these feedstocks and their conversion to energy differs and needs to be evaluated on a case by case basis. Also, second generation biofuels still play a very minor role on the market yet. The EEA report therefore recommends a broader mix of crops to reduce environmental impacts. Specifically, this mix should include perennial crops, which are not harvested annually – for example energy grasses or short rotation willow plantations. Perennial crops generally reduce waterway pollution from leaching of nutrients and would enhance, rather than harm, ‘ecosystem services’ provided by farmland (EEA, 2013a).

**Impacts due to (indirect) land use change**

Biomass production for bioenergy often necessitates changes to land use, with significant implications for related systems, including nutrient cycles. Even while the effects of using biomass for energy will vary greatly from location to location, it often involves further intensification (including fertiliser use and N input) of existing land uses. It could mean converting directly or indirectly non-cropped land (e.g. pasture land) into cropped land with significant interference in nutrient cycles (EEA, 2013a).

First, previously unused land could be directly converted for energy crop production. Second, food crops may be used instead for biofuel production or land previously cropped with food crops may be directly converted to energy crops production, thereby posing threats to food security and potentially placing more pressure on land outside of the EU. For instance, the total area used for energy crop growth in the EU-27 in 2008 was 5.5 million ha, up from 4.0 million ha in 2007 and 3.7 million ha in 2006 (Panoutsou, et al., 2011). Expansion of the area covered by energy crop production (or food crops that are used for energy purposes) reduces the area covered by food crop production for food consumption.

Third, expanding areas of land used for crop production contributing to biofuels may cause indirect land use change through conversion of previously unused land to produce food crops. Displacement of crop production onto previously unused land can lead to the conversion of forests and savannah to agriculture etc. resulting in the release of CO₂ and N₂O-emissions (Panoutsou, et al., 2011). This type of indirect land use change can also reduce or negate greenhouse gas savings from bioenergy production based on energy crops (Stehfest, et al., 2010). Other severe indirect impacts include an increase of land prices and fees for the lease of land and impacts on food prices/availability.
Usage of cereal straw resources for bioenergy

Particularly for the generation of electricity, the usage of straw\(^{29}\) for (co-)firing of biomass is a bioenergy production pathway with significant impacts on agricultural soils and their nutrient availability. The collection of straw interferes with nutrient cycles as it removes potential nutrient inputs and thereby depletes the organic matter content that the soil otherwise would have. Also P and K removals are significant and can result in a potential decline in soil fertility (Joint Research Centre, 2007). It is important to bear in mind however, that the alternative non-bioenergy use of ploughing the straw back to the soil has an impact on soil nitrogen as the rotting straw consumes soil nitrogen. This can lead to the situation where farmers often remove the straw to save on N fertiliser, despite some (but very slow) return of organic N from the straw (about 5 kg N/t straw) (Joint Research Centre, 2007). If the development of second-generation biofuel technologies advances, straw as well as non-food plant biomass could also be a potential source for second generation biofuels (Naik, et al., 2010).

Use of residues from bioenergy processes (biogas slurry and biomass ash)

The reutilisation of residues from bioenergy processes can be an important factor in reducing the use of artificial fertilisers and in achieving nutrient recycling in agriculture, particularly phosphorus (Eichler-Löbermann, 2012). In Europe there are two bioenergy processes that are most widespread: combustion of solid biomass and anaerobic biogas production. The focus of the following paragraphs will therefore be put on fertiliser effects of biomass ash and biogas slurry.

- **Biomass ash**: residues of biomass combustion are the oldest mineral fertilisers in the world and contain nearly all nutrients plants require except nitrogen (Eichler-Löbermann, 2012). Depending on the original biomass, the phosphorus (P) content in ash varies widely – in woody biomass it is usually low, whereas the P content in ash based on cereals and oilseed crops or animal manure is higher (ibid). There is also a range of interaction of ash and crop cultivation on the uptake and utilisation of P from ashes. Due to these differences general conclusions regarding the fertilisation efficiency cannot be given. However, a positive effect on plant P nutrition can usually be expected (Eichler-Löbermann, 2012). There are also ranges for K depending on the biomass source (e.g. most concentrated in straw), whereas N is almost fully discharged with the smoke gases. Increased biomass combustion and biomass co-firing leads to questions as to how residues from the combustion can be utilised, what options there are in the future, and what role environmental considerations play. To discuss these questions, the International Energy Agencies “Bioenergy task 32” has written a report that includes case studies from different EU Member States (Eijk, 2012). Currently, ashes are often used in road construction and partly also as fertiliser (e.g. in Germany and Austria). In many countries however, most of the ashes are still discarded (ibid). The use of biomass ash as fertiliser depends on many issues: A main differentiation needs to be made with regard to the combustion technology and biomass sources, e.g. between dedicated biomass combustion plants and co-firing plants. However, even dedicated power plants use a wide range of biomass sources, including (waste) wood but also black liquor, and waste streams from municipal solid waste, sewage sludge, poultry litter, meat and bone meal, and paper sludge. The composition of the fuel has a strong impact on the utilisation options available for the ashes produced. Future uses also differ for the different ash fractions (bottom ash, fly ash) which have different nutrient concentrations. Regulations also apply when ashes are used as fertiliser, particularly with regard to their content of heavy metals. These regulations differ per country. In Austria, they are used as additives for compost production or as a liming agent for forest soils in Germany. In Finland, there are differentiations between “agroash” and “forest ash”. Limit values are usually set on minimum content and availability.

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\(^{29}\) Other crop residues are also used for energy production, such as corn stover, wood shavings, etc., though often below their production potential (Naik, et al., 2010).
of nutrients (N, P or K) or (Ca, Mg, or S). These ashes are usually not used as a source of nitrogen (N) since this element is missing in ash. In addition, a maximum content of heavy metals (e.g. Cd, Cr, Cu, Pb, Zn) may be set (ibid). Overall, there are still large uncertainties if and how the increasing amounts of biomass ash should be used as fertilisers and how the technological, organisational and regulatory challenges can be solved and how their use can be monitored.

- **Biogas slurry**: also, biogas slurry can be used as a fertiliser. While data is available regarding the effect of biogas slurries on nitrogen, the effect of biogas slurries on the soil P cycle has been investigated much less (Eichler-Löbermann, 2012). Preliminary results however show that good effects of biogas slurry can be expected regarding the P supply of crops (ibid). To further evaluate the benefits of biogas slurry, more knowledge is needed on its effects on nutrient availability. This, however, is complex as the transformation of organic compounds and nutrient release depends on many factors such as the stability of organic substances, climatic conditions, soil properties, type of cropping system and interaction with mineral fertilisers (ibid). With regards to the evaluation of P as a fertiliser, a comparison between digested (biogas) slurry and non-digested (animal manure) reveals that the chemical analysis of non-digested dairy slurry and biogas dairy slurry had almost the same nutrient composition (see Table 4). However, the growth of biogas plants has caused an excess supply of biogas slurry in many locations leading to over-fertilisation and negative impacts in water quality (e.g. in northern Germany) (Rolink, 2013; Fertmann, 2014; Widmann, 2014), as well as the above mentioned effects in terms of land use change and increased leaching potential.

### Table 4 – Nutrient content in non-digested dairy slurry and in different types of biogas slurry (% of fresh matter)

<table>
<thead>
<tr>
<th></th>
<th>DM</th>
<th>OM</th>
<th>N</th>
<th>NH\textsubscript{3}-N</th>
<th>P</th>
<th>K</th>
<th>Mg</th>
</tr>
</thead>
<tbody>
<tr>
<td>Non-digested dairy slurry</td>
<td>9.3</td>
<td>7.5</td>
<td>0.46</td>
<td>0.23</td>
<td>0.08</td>
<td>0.32</td>
<td>0.07</td>
</tr>
<tr>
<td>Biogas dairy slurry</td>
<td>8.1</td>
<td>6.3</td>
<td>0.50</td>
<td>0.25</td>
<td>0.08</td>
<td>0.33</td>
<td>0.07</td>
</tr>
<tr>
<td>Biogas pig slurry</td>
<td>4.2</td>
<td>3.1</td>
<td>0.46</td>
<td>0.35</td>
<td>0.07</td>
<td>0.20</td>
<td>0.05</td>
</tr>
<tr>
<td>Biogas maize slurry</td>
<td>11.3</td>
<td>8.5</td>
<td>0.64</td>
<td>0.3</td>
<td>0.16</td>
<td>0.47</td>
<td>0.09</td>
</tr>
</tbody>
</table>

DM-dry matter, OM-organic matter, N- nitrogen, NH\textsubscript{3}-N- ammoniacal nitrogen, P-phosphorus, K-potassium, Mg-magnesium

Source: (Bachmann & Eichler-Löbermann, 2009), cited in (Eichler-Löbermann, 2012)

### 2.2.2.4 Water management

**Focus on the effects of irrigation**

Soil water is essential for the uptake of nutrients by plants, which is only possible when the nutrients are in a dissolved form. By sustaining appropriate soil moisture conditions, irrigation may therefore contribute to nutrient availability for plants. Some irrigation practices, however, may contribute to run-off and leaching resulting in nutrient transfers to water bodies. Those negative impacts due to irrigation can result from both quantity and quality (e.g. nutrient and salt levels) of water applied, from the timing of irrigation and from the type of irrigation technologies. Implementing alternative irrigation practices that modulate these three factors may decrease nutrient content of surface and groundwaters (Barros, et al., 2012). For instance, a case study carried out in the Ebro River basin in Spain between 2000 and 2007 highlights that better irrigation efficiency may decrease nitrate amounts contained in soil and transferred to surface water by 24 % (García-Garizábal I. and Causape, 2010; García-Garizábal, et al., 2012).
While the quantity of water supplied is defined by the water manager, the type of irrigation technology mostly influences water efficiency. Flood irrigation systems, which aim to apply water to field’s full capacity, are likely to cause soil water logging in some part of the field and thus leaching since uniform flooding is not generally feasible. Flooding can also be performed with on-demand irrigation systems that better allow matching water supply to crop needs depending on their location in the field and therefore limiting the risks of local leaching and run-off (García-Garizábal I. and Causape, 2010). Regarding sprinkler irrigation, even though such a system is especially suitable for irrigating hilly terrains with undulating slope, the sprinkler pattern is easily distorted by wind which may not allow uniform irrigation and might result in locally excessive water input where run-off is thus likely to occur (TNAU, 2013), along with the transfer of nutrients. Run-off issues are also associated with certain types of sprinkler irrigation systems such as gun type sprinklers that can deliver water in excess (Evans & Sneed, 1996). Another irrigation technology is drip irrigation which consists in providing water drop by drop to the active plant root zone. Drip system is known as the most efficient way of supplying irrigation by significantly preventing from run-off and leaching (The Toro Company, 2013; Irrigationglobal, n.d.; Gardenas, et al., 2005). Furthermore, drip irrigation allows applying fertilisers to crops through the drip irrigation system at the same time as water. This method is known as “fertigation” and favours fertiliser inputs only where needed in the proper quantity, which limits nutrient surpluses and potential losses (Gardenas, et al., 2005).

The timing of irrigation may also influence the behaviour of water in the soil, and therefore the fate of nutrients. It depends on several factors including the crop water need, rainfall, evaporation, and the soil water retention. Multiphase or cyclic irrigation may have a positive effect on the quantity of soil water content and thus nutrient transfer as water is progressively bounded to soil and consumed by plants instead of being available at all time even though when it is not necessary (NRCS, 2012). Experiments conducted by Fare and Gilliam on Ilex crenata actually confirm that cyclic irrigation reduces nitrate leaching compared to continuous irrigation as 46 % of applied nitrogen was leached when 13 mm of water was applied in three cycles versus 63 % when the same amount of water was applied in a single cycle (Fare, et al., 1994).

Moreover, the timing of irrigation relative to the initial soil hydric conditions influences the risks of leaching within the root zone. Under wet conditions, nitrogen dissolved into irrigation water can be taken up by plants in the root zone. Under drier conditions, however, water may keep infiltrating into the soil below the root zone, thereby becoming unavailable to plants, which increases the risk of leaching if they supply of water continues. A study on strawberry production in Spain reveals, for instance, that nitrate uptake by crops was enhanced and thus nitrate leaching was reduced when irrigation regime accounts for soil hydric properties (Guimera, et al., 1995).

In this respect, it is important to keep in mind that the relevance of alternative irrigation practices depends on several parameters including climate, hydrogeology, and the agronomic characteristics of the territory. Thus far, most of findings obtained are site-specific, limiting the possibility of extrapolation of potential efficient irrigation practices (García-Garizábal I. and Causape, 2010).

The quality of irrigation water also represents an important issue regarding nutrient content of water bodies. First, the reuse of wastewater that is rich in nutrients for irrigation, for instance, contributes to the nutrient balance and is likely to cause nutrient excess and leaching. Second, poor quality irrigation water, like sodium-rich water, may enhance nutrient leaching and negatively impact groundwater quality, due to high content of sodium that contributes to the transfers of ions, including potassium and phosphorus nutrients, from soil to water (Jalali & Merrikhpour, 2008). Salinisation of soils prevents biological N transformations from occurring and thus hampers soil fertility (Curtin, et al., 1999; Velthof, et al., 2011). Legumes and nitrogen-fixing crops are also prevented from performing biological nitrogen
fixation processes (Delgado et al., 1993). In reducing plants’ nitrogen use efficiency, salinisation can subsequently lead to more leaching and gaseous losses from the soil (Velthof, et al., 2011).

**Focus on the drainage effect**

Drainage consists of removing excess water from the soil surface in order to optimise plant growth. Surface drainage refers to the removal of excess water at the soil surface whereas sub-surface drainage refers to the removal of excess water from beneath the soil surface. Both ways of collecting excess water simultaneously result in preventing nutrients dissolved or bounded to soil particles in suspension from reaching surface or groundwater bodies by run-off or leaching (FAO, 1992; García-Garizábal I. and Causape, 2010). However, in some cases drainage water can be directly discharged into streams leading to nutrient contamination. When it reaches wetlands, N is nonetheless partly removed via denitrification.

As put forward in Sims et al. research, surface drainage can effectively reduce phosphorus transfers from soil to surface water by decreasing run-off water volumes through channelling and collection of water. For instance, on clay-loam soils used for maize production, amounts of phosphorus lost in run-off water can decrease from 7.8 kg/ha per year to 5.0 kg/ha per year in undrained and drained soils respectively. This study also deals with the effect of drainage management on phosphorus transfers to water, pointing out that even though phosphorus concentration was higher in water collected by slow drainage, total phosphorus amount removed from soil was higher in water collected by fast drainage because of greater water volume removed (Sims, et al., 1998).

Another experiment carried out in Germany on different types of soil used for grass, maize and fodder beet establishes a link between sub-surface drainage and nitrogen transfers to groundwater, depending on drain depth. It appears that deeper drainage may enhance nitrogen leaching to groundwater, because it still allows high nitrogen microbial mineralisation to occur in saturated areas below the root zones, resulting in free nitrogen being potentially leached to groundwater (Behrendt, et al., 2004).

Drainage may also increase nitrogen emissions. In the case of drainage of organic soils, microorganisms start to decompose the organic matter. High concentrations of nitrogen are produced: though ammonification NH\textsubscript{3} is produced, subsequently NO\textsubscript{3} via nitrification, N\textsubscript{2}O is released as a by-product of these reactions.

**2.2.2.5 Agroforestry**

Agroforestry consists in the synergetic growing of agricultural (livestock, food crops) and tree products (fruits, timber, firewood, etc.). Trees can be grown inside parcels or on their boundaries (CIRCLE-2, 2013; IPCC, 2007a). Silvopastoral systems refer to woodlots with grass and pasture while agrisilvicultural systems refer to arable plots with crops and trees. Agroforestry practices can be implemented in all crop systems, provided that tree density is adjusted to crop shadow requirements, and that selected crop and tree species competition for water and nutrient is effectively reduced (i.e. crop roots in the topsoil and tree roots in the subsoil). This correct adjustment enhances deep rooting. Additionally, mixing tree species reduces the risk of diseases that could devastate crops (INRA, 2013b). Innovative agro-forestry systems are compatible with mechanisation constraints (Dupraz, et al., 2005).

Agroforestry is commonly credited with more efficient nutrient cycling (i.e. nutrient transfer from one ecosystem component to another) than many other systems. This comes notably from the combination of crop/grass superficial roots with trees’ deeper and more extensive root systems, enabling a higher nutrient capture and recycling potential. Woody perennials also have a greater contribution to soil surface enrichment thanks to litter fall. (Nair, 1993).
Due to this optimised nutrient cycling, several studies have shown the potential of agroforestry systems to reduce nutrient leaching losses. Lehmann et al. (1999) established that a sorghum tree based system enabled to mitigate nutrient leaching in general compared to sorghum monoculture in Kenya (nitrogen leaching decreasing of 53 %). Studies in Canada revealed that agroforestry nitrogen losses could be up to 50 % lower than in monoculture, trees taking up any N not utilised by alley crops (Briggs, 2012). In a Mediterranean context, López-Díaz et al. (2011) also highlighted that agroforestry could help reducing nitrate leaching from irrigated pastures on sandy soils.

At the European scale, the potential spread of agroforestry systems could reach up 90 million of hectares, i.e. 55 % of arable lands in Europe, and silvoarable plantations could help mitigate key environmental problems such as soil erosion and nitrate leaching in 65 million hectares. Assuming that 20 % of farmers implement agroforestry measures on 20 % of their lands, agroforestry systems would cover about 2 to 3 million of hectares in Europe (Dupraz, et al., 2005).

2.3 Potential impacts of nutrient losses on water, air, soil, biodiversity, climate and human health

Nutrient losses can cause impacts on water, air, soil, biodiversity, climate and human health. Impacts are due to the nutrients, in particular nitrogen, that are released in the environment at different stage of the agricultural activity as gases or soluble ions.

2.3.1 Impacts on climate

Impacts of nutrients on climate change

Agricultural activities strongly impact climate. Between 10-12 % of global anthropogenic GHG emissions are attributed to agriculture, which correspond to 5.2-5.8 Gt CO₂eq/yr (IPCC, 2014). In the EU, agriculture also accounted for approximately 10 % of GHG emissions in 2010 but this figure does not include emissions associated with the production of agricultural inputs and certain imports (Underwood, et al., 2013). The largest share of the EU’s agricultural GHG emissions stems from nitrous oxide (N₂O) and methane (CH₄), including manure and inorganic fertiliser application to agricultural soils, manure management, and cultivation of organic soils. International estimates show that the share of the agricultural GHG emissions directly caused by the disruption of mineral cycles due to fertilisation and manure management accounted for 51.5 % of the total.

Nitrous oxide (N₂O) is a highly potent greenhouse gas and is involved in destruction of the stratospheric ozone-layer, nitric oxide contributes to formation of tropospheric ozone, a plant toxin and greenhouse gas. While the global warming potential (GWP100) of carbon dioxide is 1, the GWP100 value for

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30 Annual total non-CO₂ GHG emissions such as N₂O and CH₄, not including emissions from e.g. fertiliser production, land use change and forestry, use of fossil fuels

31 464.3 million tonnes of CO₂ equivalent in 2011 (Eurostat 2013), emissions from e.g. fertiliser production, land use change and forestry, use of fossil fuels are not included

32 N₂O from agricultural soils (32.5 %) and CH₄, N₂O from manure management (9 %) for 2005 and international GHG emissions (EPA, 2011)
methane is 25 and 298 for nitrous oxide (IPCC, 2007b). In Europe approximately 80 % of all anthropic N\textsubscript{2}O emissions stem from agriculture. N\textsubscript{2}O emissions in EU dropped by around 20 % between 1990 and 2010 largely due to a reduced use of organic and mineral nitrogen fertilisers as well as indirectly due to the existence of obligatory set aside (until its abolition in 2008), and improved protection of permanent pasture (Eurostat, 2014a) (European Commission 2009). Moreover, the development and implementation of agricultural and environmental policies (e.g. the Nitrates Directive and the “cross compliance” provisions within the Common Agricultural Policy) played a significant role.

Another share of N\textsubscript{2}O, CH\textsubscript{4}, and CO\textsubscript{2} emissions stems from soil carbon stock changes including conversion of grasslands to arable land, the cultivation of organic soils and drainage of wetlands for agricultural use, which are accounted for under LULUCF\textsuperscript{34}. Although a minor threat to soils with locally negative climate change impacts in Europe, deforestation in order to convert land to cropland or pasture affects the soil’s physical, chemical and biological characteristics, which are determining factors affecting the extent of the reactions occurring in the soil such as nitrification and denitrification. It has been shown that the conversion from forest to pasture leads to change in the soil organic carbon and nitrogen contents and on the rates at which soil organic matter are turned over and during which N is formed (Nyamadzawo, 2012). These changes can lead to an increase of N\textsubscript{2}O emissions. The extent to which the conversion of the land leads to increased N\textsubscript{2}O emissions depends on the use of fertilisers and crop residues. The climatic conditions are also important as the generation of N\textsubscript{2}O is increased in wet conditions (Smith, et al., 2007). High emissions are observed with the increase in the soil of mineral N concentrations from the use of fertilisers containing N (Bouwman, 1990). The conversion of arable cropland to grassland (land use change) results in the increase of soil carbon content because of the lower disturbance of the soil and the reduced carbon removed through harvested crops. In addition, grasslands have lower N\textsubscript{2}O emissions than cultivated lands, resulting from lower nitrogen inputs (Smith, et al., 2007).

The consequences of climate change on agriculture and nutrient cycles

Climate change will also have an important effect on agricultural production systems. With the forecasted increase in temperature, evapotranspiration will increase and thus water availability will decline in some regions. Therefore plant water demand and the amount of irrigated water will rise. Temperature rise will also affect the use of different crop species. Agricultural production systems will have to be adapted for example by using species that demand less water and can withstand extreme temperatures or change sowing dates. Furthermore, agriculture will face more extreme events such as floods, droughts, hail, storms, but also fire risks, which are not predictable (EEA, 2012b). In the light of the increasing demand for water, droughts pose a threat on water resources, in particular in the southern EU Member States and water-intensive agricultural production systems. Heavy rainfall events and storms can lead to a higher soil erosion rate that also eliminates nutrients from cropland. With heavier rainfall, the occurrence of floods will increase, resulting in loss of harvest and/or arable land in areas of high flood risk. Climate change impacts such as longer growing seasons due to warmer temperatures, may also promote better conditions for the growth and development of organisms such as insect pests.

\textsuperscript{33} Methane emissions stemming from agriculture, also primarily related to livestock, have a share of c. 35 % of all anthropogenic CH\textsubscript{4}-emissions at global level (FAO 2014)\textsuperscript{33} and c. 50 % of all anthropic CH\textsubscript{4}-emissions in the EU (EEA 2013). \url{http://www.eea.europa.eu/data-and-maps/indicators/greenhouse-gas-emission-trends/greenhouse-gas-emission-trends-assessment-5} (Fig 4 on the webpage); corresponds to 270,8 million tonnes of CO\textsubscript{2}-equivalent (Eurostat 2013)

\textsuperscript{34} Land use, land-use change and Forestry
and crop diseases (EEA, 2012b). Such developments will in particular result in declining crops yields and overall yield losses.

However, climate change can also favour agricultural productivity through lengthening of the growing cycle, reduced days of frost, and more favourable overall climatic conditions in northern Europe, but the adverse effects relating to extreme weather events (e.g. storms, hail, pests) may be of a similar magnitude (Underwood, et al., 2013).

Animal husbandry will also have to cope with the higher temperatures in adapting their stables or trucks e.g. with climate control systems.

### 2.3.2 Impacts on air

Emissions to air arise from livestock rearing, manure storage and the application of N-fertilisers and manure to the soil. The main air pollutants emitted are:

- $\text{NH}_3$ that results from $\text{NH}_4^+$ volatilisation;
- PM emissions from agricultural activities and facilitated by $\text{NH}_3$ emissions;
- NO emitted from denitrification which contributed to the formation of O$_3$; and
- Climate change gases :methane and N$_2$O (see section 2.3.1)

**Ammonia emissions from agriculture**

Ammonia ($\text{NH}_3$) is released to air (volatilisation) from all ammonium ($\text{NH}_4^+$) containing products. The emission and deposition of ammonia are harmful to ecosystems as it causes acidification and disrupts plant communities. Furthermore, $\text{NH}_3$ is a precursor for the formation of particulate matters (PM) which has adverse effects on human health affecting the respiratory and cardiovascular systems and causing premature death. Ammonia is also a precursor of nitrogen oxides and can be, in certain situations, a source of nitrous oxide (N$_2$O) which is a greenhouse gas (see section 2.3.1).

**Nitrogen volatilisation**

Direct emissions to air following the application of fertiliser containing nitrogen are one of the most important sources of ammonia ($\text{NH}_3$) emissions from agriculture. Nitrogen volatilisation is a reaction due to a physic-chemical process between ammonium ($\text{NH}_4^+$) and ammonia ($\text{NH}_3$) which is influenced by the pH of the soil. In natural ecosystems there is very little ammonia volatilisation as ammonium is transformed in NO$_3^-$ and is absorbed by plants. In agricultural soils, volatilisation happens when fertilisers containing nitrogen as ammonium NH$_4^+$ are applied to the soil surface layer. The natural mechanisms for absorbing ammonium (i.e. nitrification, absorption by crops, adsorption by soil particles and immobilisation in organic matters) require several days to process and eliminate the excess of ammonium. Consequently, it is during this period (1 to 5 days) that emissions to air happen, as a result of volatilisation. They decrease over time as the NH$_4^+$ is absorbed onto soil colloids, nitrified or absorbed by plants (Rochette, 2008).

The quantity of ammonia volatilised from fertiliser depends on several factors such as composition of the fertiliser, the temperature of the soil, the surface area exposed to the atmosphere and the resistance to ammonia transport in the atmosphere (Hutchings, et al., 2009). This resistance depends on factors such as the meteorological conditions and the wind speed and direction. The quantity of ammonia volatilised is linked to the ammonia concentration at the surface of the fertiliser applied, and the concentration of ammonia contained in the air above the surface (Huijsmans, 2003). Other factors such as the pH of the soil, the meteorological conditions and the properties of the soil or crops the fertilisers
are applied to are relevant. The factors influencing ammonia volatilisation are presented in further details in Annex 3.

**Contribution of the agriculture sector**

Whilst the contribution of the agriculture sector has been decreasing over the last years, due to the reduction in the total number of livestock (especially cattle) (see section 2.2.1), at EU28 level, agriculture still generates the overwhelming majority (93%) of the total ammonia emissions to air (EEA, 2013c). Other contributing sectors are road transport, waste treatment and some industrial processes (responsible together for around 5% of the total EU28 ammonia emissions) (see Figure 10).

![Figure 10 – Emissions of ammonia in EU-28 from agriculture (2011)](source: EEA, 2013c)

The main agricultural sources of ammonia emissions are livestock rearing and the use of inorganic fertilisers. Figure 11 presents the partition of emissions of ammonia from agriculture by livestock type and agricultural activity. It shows that in 2011 the three main sources of emissions are from cattle (dairy and non-dairy) with more than 1 400 kt of ammonia emitted, the use of inorganic fertilisers (744 kt), and emissions from swine rearing (553 kt). Moreover, it is estimated that crop production and agricultural soils contribute to 10% of the total source for emissions of ammonia with noticeable variation between Member States (Hutchings, et al., 2009).

![Figure 11 – Emissions of ammonia in EU-28 per agricultural livestock and activity (in kt, 2011)](source: EEA, 2013c)

There are some variations to the contributions of emissions of ammonia from agriculture between Member States. Figure 12 presents the quantity of emissions per Member State and per main type of
livestock. It shows that France, Germany, Italy, Spain, Poland and the UK are the main emitters of ammonia, followed by Romania, the Netherlands and Ireland.

Ammonia emissions from livestock rearing depend on several factors including for example the quantity and the N content of feed consumed; the efficiency of the conversion of the N contained in the feed and the amount of N deposited in excreta; the proportion of time spent by animals indoors and outside (e.g. in pasture); the housing system of the animal (i.e. the floor area per animal); the storage conditions of manure; and the climatic conditions in the building and the ventilation system. More information about these specific points and their influence on nutrient losses is presented in section 2.2.1.

The EMEP guidebook provides guidance for Member States reporting under the Convention on Long Range Transboundary Air Pollution. It includes a methodology and emission factors to use when determining emissions of ammonia from agriculture. The specific emission factors for ammonia emissions from housing, manure storage, spreading or outdoor grazing of livestock are presented in Annex 4. Similarly, for emissions arising from the use of N fertiliser, the EMEP guidance considers that there is not enough evidence to differentiate between crops. Consequently, the default emission factor (tier 1) for ammonia emissions from crop is used for all types of crop production and is 0.084 kg of NH₃ per kg of fertiliser-N applied. In addition, more refined emission factors (tier 2) are defined in relation to the type of fertiliser used and take into account the temperature and the pH of the soil. The emission factors are presented in Annex 4.

**Emissions of particulate matter (PM) from agriculture**

In 2008, agriculture contributed to around 4% of PM2.5 emissions (particulate matter below 2.5 micrometres) and around 12% of PM10 emissions (particulate matter below 10 micrometres) in the EU (EEA, 2010). The latest emissions data from 2011 reported by Member States under the LRTAP Convention shows a similar trend with 11% of PM10 and 3% of PM2.5 emissions in the EU emitted by the agriculture sector. Figure 13 presents the split of emissions in the agriculture sector for PM10 and PM2.5.
For PM10, the highest emitting categories are farm level agricultural operations, which include emissions from crop production and agricultural soils. It is estimated that soil cultivation and crop harvesting account for more than 80% of the total PM10 emissions from tillage land (Hutchings, et al., 2009). Another significant source of PM10 emissions is the field burning of agricultural wastes. This agricultural practices consisting in burning crop residues to clear land, but it does not include emissions from the burning of crops products, such as straw used on farm. It is interesting to note that under cross compliance’s Good Agricultural and Environmental Condition (GAEC) standards, most Member States ban the burning of straw and stubbles, the notable exceptions being Cyprus, France, Ireland and Slovenia. For livestock rearing, poultry and swine are the main sources of PM10 emissions.

PM2.5 arises mostly from livestock rearing, with cattle, poultry and swine representing 60% of the total emissions. Field burning of agricultural wastes and crop cultivation are other important sources of PM 2.5 emissions.

**Emissions of NO and NO$_2$ from agriculture**

Nitric oxide (NO) is emitted to air by agricultural soil through nitrification and denitrification processes (see section 2.1.1). The emissions follow the application of fertiliser and vary according to the following elements: the concentration of nitrogen contained in the fertiliser, the temperature, the soil carbon concentration and the soil moisture. The type of fertiliser used and the type of crops cultivated do not appear to have a specific influence in the quantity of NO emitted (Hutchings, et al., 2009).

For soil with a pH above 5, the nitrification process is considered to be the main source of NO emissions. NO emissions can also result from denitrification process, but it is considered to be less common and less significant (Hutchings, et al., 2009). Furthermore, it is considered that soil with organic carbon content over 3% emits more NO than those with -content below 3%. In addition, drainage, coarse texture of the soil and neutral pH are factors that can lead to an increase of NO emissions (Hutchings, et al., 2009).

The method used to apply the fertiliser to the soil is influential in the quantity of NO emissions. Broadcasting technique results in higher NO emissions than incorporation. It is estimated that in Europe, 1.2% of the nitrogen applied on cropland and grassland is emitted to air as NO (Hutchings, et al., 2013).
Nitric oxide is a short lived gas that once emitted to air, combines with oxygen to form nitrogen dioxide (NO₂). NO and NO₂ are not reported individually by Member States, however NOₓ (of which both NO and NO₂ are compounds) is. It is considered that soils contribute between 4 to 8 % of the total NOₓ emissions in the EU (EEA, 2014). However, emissions are sensitive to climate, and in hot weather conditions this share can increase to 27 %. Between 1990 and 2010, the EEA estimated that emissions of NOₓ originating from agriculture has reduced by 24 %, and in 2011, these represented 1.4 % of the total NOₓ emissions of the EU (EEA, 2014). Figure 14 presents the emissions of NOₓ reported by Member States under the CLRTAP for agricultural activities.

![Figure 14 – Emissions of NOₓ in EU-28 per agricultural livestock and activity (in kt, 2011)](source: EEA, 2013c)

The majority of emissions of NOₓ are from crop cultivation and activities related to agricultural soils. Emissions from livestock are limited and represent around 10 % of the total NOₓ emitted by agriculture sector. Emission factors for calculation of NO emissions from livestock are presented in Annex 5.

**Impacts from acidification**

The pollutants mentioned above (NH₃, NOₓ, and NO₂) impact the environment through the acidification of the air and the facilitation of the formation of ozone. They also have impacts on human health due to the role of NH₃ and NOₓ in the formation of particle matter (PM2.5) and ozone and the overall degradation of air quality.

Acidification occurs when airborne particles are deposited to the ground by acid rain, snow, fog, and are transformed into nitric acid in the soil. The nitric acid is then emitted to the air through nitrification and is transported over relatively long distances (EEA, 2007b). Other gases such as sulphur dioxide (SO₂) and nitrogen oxides (NOₓ), mostly produced by industrial activities and transport, contribute to acid rain formation and subsequently acidification. However, ammonia is an important source of acidification, and it has been estimated that ammonia is responsible for 24 % of the acidification (Sensi, 1994).

Acidification and acid rain cause damage to forests, crops and vegetation. Acid rain damages forests and other vegetation both directly and indirectly. Directly, when dissolved chemicals in the acid rain damage the leaves/needles and bark of trees and other vegetation, leaving them more vulnerable to disease and insect damage. Indirectly, when the chemicals in acid rain impact soils by changing its pH, killing soil micro-organisms and reacting with nutrients in the soil, causing some nutrients to dissolve and be washed away by rain before they can be absorbed (leading for example to magnesium deficiency). Another indirect effect of acid rain is the mobilisation of harmful chemicals such as...
aluminium, which are then released into the soil and harm trees/vegetation. The impact of acid rain on food crops can be minimised by the application of lime and fertiliser to correct (i.e. raise) the pH levels of the soil and replace lost nutrients, however this is generally not an option for non-cultivated areas e.g. forests. In its 2007 assessment of the environment, the EEA estimated that the forest areas receiving acid deposition above their critical load were projected to decrease from 23 % in 2000 to 13 % in 2020. However, for the forest areas that are still above the critical loads ammonia is considered to be the main source of acidification (EEA, 2007b).

The acidification of water bodies and soil can cause harmful effects on ecosystems. More details are available on these effects in section 2.3.3 and section 2.3.4. Acidification also has impacts on biodiversity which are further described in section 2.3.5.

The EEA conducts a regular assessment of the exposure of ecosystems to acidification, eutrophication and ozone. Critical loads of acidity and of nutrient nitrogen are used in order to describe the exposure to acidification. The assessment consists in analysing the change of exceedance over time, this allows to identify the magnitude of the exceedance (deposition minus critical load) but also the areas where critical loads are no longer exceeded.

The 2013 assessment concluded that the areas where the critical load is exceeded have declined by more than 80 % in 2010 in comparison with 1990 (EEA, 2013d). The EEA has estimated that by 2020, the risk of ecosystem acidification is likely to be a much localised issue (i.e. hot spots) concerning in particular the border area between the Netherlands and Germany (EEA, 2013d).
Figure 15 presents the results for 2000 and 2010 and shows the improvement that has been achieved.

**Impacts from emissions of NO\textsubscript{x} (NO and NO\textsubscript{2})**

Emissions of NO\textsubscript{x} contribute to acid deposition, which affect air quality, water quality and soil. NO\textsubscript{x} emissions contribute to the formation of secondary particulate aerosols and, by reaction with sunlight, tropospheric ozone (O\textsubscript{3}) in the atmosphere\textsuperscript{35}.

Tropospheric ozone is a secondary pollutant that can have adverse effects on ecosystems especially during the summer months. A high concentration of ozone in the air is harmful to crops leading to decreasing yields and reduced disease resistance. The main precursor for ozone formation is NO\textsubscript{x} and to a lesser extent carbon monoxide and methane\textsuperscript{36}.


In addition to environmental impacts, NO\textsubscript{2} emissions can have adverse impacts on human health with high concentrations causing respiratory problems (i.e. inflammation of airways and lungs). Similarly tropospheric ozone has adverse effects on the human respiratory system.

Figure 16 presents the share of arable land that has been exposed to ozone. The Air Quality Directive set a target value for the protection of vegetation of 18 000 \(\mu\)g/m\(^3\) with a long-term objective set at 6 000 \(\mu\)g/m\(^3\). The figure shows that in 2010, for nearly 25% of the arable land, the limit for the protection of the vegetation set under the Directive was breached.

**Impacts from emissions of particulate matter (PM)**

Ammonia is a precursor of particulate matter. Once emitted, ammonia can react with sulphuric and nitric acids to form ammonium sulphate and nitrate aerosols, respectively. There is not a lot of information available on the share of ammonium and nitrate contained in emissions of PM2.5 but research carried out in 2004 in the USA estimated that these substances could contribute to 13% of the PM2.5 concentration levels each (from 3 to 20% for ammonium and 4 to 37% for nitrate) (Hodan & Barnard, 2004). There are no similar data available for PM10 concentrations. Depending on climatic conditions such as insulation, temperature, humidity and presence of other constituents in the atmosphere, ammonium nitrate can contribute to high PM2.5 and PM10 concentrations (Hamaoui-Laguel, et al., 2009), the latter being higher in case of high temperature and humidity, inducing the formation of more particles (Walker, et al., 2004; Hodan & Barnard, 2004).

PM impacts are more severe for human health than the environment as PM are inhaled by farm workers and local residents and can cause respiratory disease (Takai, et al., 2002). Primary PM is likely to have more localised impacts on health and the environment, however the formation of secondary PM from ammonia can contribute to health and environmental impacts on a wider geographical scale due to their increased transportation over longer distances. A 2003 article indicated that in Europe secondary particles constitutes a large part of PM10 emissions (from 50% to 90%) and around 50% or more of PM2.5 emissions (Erisman & Schaap, 2003).

Concerning environmental impacts, deposition of acidic compounds contained in PM contributes to acidification of soil and eventually water pollution (see respective sections below).
**Impacts from odour**

Emissions of odour are related to many different compounds, such as mercaptans, \( \text{H}_2\text{S} \), skatole, thiocresol, thiophenol and ammonia, although not all compounds that are involved have been identified yet (Joint Research Centre, 2013a). Most odours originate during the anaerobic decomposition of wet organic matter such as manure, feed or silage. The process is increased in the presence of warm temperatures. During this decomposition process, organic (volatile organic compounds) and inorganic gases (e.g. ammonia and hydrogen sulphide) are produced which are then transported through air. The link between ammonia concentration and odour is unclear. Some research has found a relationship between high levels of ammonia emissions and odour. However, this could not be confirmed conclusively, and it seems to be considered that ammonia emissions and odour from manure storage units cannot always be correlated (Atia, et al., 2004). The odours will dissipate on dry and windy days, but will tend to linger if the weather is humid and windless (Toombs, 2013). Strong odours can be transported over a large area and be disagreeable for the neighbourhood (Environmental Protection Agency, 2001). This can in turn lead to local opposition to some agricultural activities.

### 2.3.3 Impacts on water

An excess of nutrients may have three major potential effects on water bodies:

- pollution of surface and groundwater bodies beyond authorised thresholds for drinking purposes and for achieving “good status” of aquatic ecosystems;
- eutrophication; and
- acidification.

The latter two aspects mainly concern surface waters.

**Pollution of water beyond authorised thresholds**

The presence of excess nutrients in water bodies represents a risk of pollution. Thresholds are set by the Water Framework Directive for the quality of aquatic ecosystems and the quality of water for drinking purposes (European Commission, 2000).

According to the water framework directive, both surface water (fresh and saline) and groundwater are expected to reach good status by 2015 depending on biological, chemical and physical parameters\(^{37}\). Excess in nutrients (in particular nitrate and phosphorus) may namely affect the good ecological status of surface waters and the good chemical status of groundwater (nitrates).

For a water source to be used for drinking purposes, the Drinking Water Directive\(^ {38}\) sets a threshold of nitrate content of 50 mg/L, both in surface and groundwater, to ensure it does not impact human activity or health in accordance with the World Health Organisation’s “Guidelines for drinking water quality” (World Health Organisation, 2011). This nitrate content threshold is supposed to be sufficiently

---

\(^{37}\) For surface water, good status is characterised by good chemical status, which consists in respecting quantitative environmental quality standards, and by good ecological status, which consists in good biological status (assessment of animal and plant species’ structure and functioning), and good physic-chemical status. Both can be affected by high concentrations in nutrients. For groundwater, good status is characterised by good chemical status but also good quantitative status, which consists in ensuring sustainable levels of aquifer replenishment and preventing from groundwater resource depletion.

conservative so as not to endanger the health of consumers, particularly the most vulnerable population including infants and pregnant women. Where concentration of nitrate in water bodies exceeds or is likely to exceed 50 mg/L but remains lower than 100 mg/L, the local authority in charge of supplying drinking water has to inform consumers, pregnant women and children should not drink this water and catchment points used for drinking water purposes can be closed. Where concentration of nitrate exceeds 100 mg/L, water cannot be drunk or used for any eating purposes, catchment points must be closed and alternative sources of water must be found.

Every four years, the Member States must provide data about nitrate status of water in monitored stations. The main findings for the reference periods 2004-2007 and 2008-2011 are presented hereafter (European Commission, 2010; European Commission, 2013b).

From a chemical and health perspective, as opposed to an ecological perspective as mentioned later, nitrate content represents a higher risk of pollution in groundwater than in surface water, due to the dilution factor. On average in the EU, about 15 % of groundwater monitoring stations exceed 50 mg/L whereas less than 5 % of surface water monitoring stations exceed this threshold. About 70 % of groundwater monitoring stations are below 25 mg/L, whereas 85 % of surface water monitoring stations are below this concentration. The largest shares of reported stations with high nitrate contents are in Malta and Germany for groundwater and Malta, Belgium and United Kingdom for surface water.

Regarding groundwater, nitrate content seems higher in unconfined aquifers than in confined aquifers (Foster, et al., 2002). Regarding surface water, nitrate content is higher in rivers than in lakes and saline water bodies.

Positive trends in water quality, especially for surface water, have been registered on average over the EU-27 from 2004-2007 to 2008-2011. Table 5 highlights this improvement in quality of fresh surface water and groundwater. However, water quality highly varies across different countries and regions in the EU and several regions with significant nitrate pollution problems are still present.

Table 5 – Trends in nitrate content of fresh surface water and groundwater monitoring stations in EU-27

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrate content</td>
<td></td>
<td></td>
</tr>
<tr>
<td>&lt; 10 mg/L</td>
<td>21 %</td>
<td>62.5 %</td>
</tr>
<tr>
<td>&gt; 50 mg/L</td>
<td>3 %</td>
<td>2.4 %</td>
</tr>
<tr>
<td>Fresh surface water</td>
<td></td>
<td></td>
</tr>
<tr>
<td>&lt; 10 mg/L</td>
<td>15 %</td>
<td>14.4 %</td>
</tr>
<tr>
<td>&gt; 50 mg/L</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Source: (European Commission, 2010; European Commission, 2013b)

Note: The fact that detection methods have been improved and that the number of stations monitored increased reinforces this positive signal at the EU level\(^\text{39, 40}\), although it is important to keep in mind that monitoring has a number of shortcomings such as a limited number of monitoring stations and differences in methodology for data collection.

Table 6 outlines the trends of average nitrate levels from the monitoring period 2004-2007 to the monitoring period 2008-2011.

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\(^{39}\) Increase by about 9 % and 10 % of the number of monitoring stations for fresh surface water and groundwater respectively.

\(^{40}\) Average density of monitoring stations during the period 2008-2011: 6.9 and 8 stations per 1 000 km\(^2\) of land area for fresh surface water and groundwater respectively.
Table 6 – Water quality trends (in percentage of the total monitored stations) from the period 2004-2007 to the period 2008-2011

<table>
<thead>
<tr>
<th></th>
<th>Decreasing trends</th>
<th>Stable trends</th>
<th>Increasing trends</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fresh surface water</td>
<td>42.1 %</td>
<td>38.7 %</td>
<td>19.1 %</td>
</tr>
<tr>
<td>Groundwater</td>
<td>30.7 %</td>
<td>42.7 %</td>
<td>26.6 %</td>
</tr>
</tbody>
</table>

Source: (European Commission, 2010; European Commission, 2013b)

In the EU, areas of land draining into waters affected by nitrate pollution or that could be affected by pollution are designated as "Nitrate Vulnerable Zones" (NVZs). In accordance with the Nitrates Directive, Member States have to identify these zones, where they are expected to take measures aiming at reducing water pollution caused by nitrate from agricultural sources or preventing such pollution (European Commission, 1991). NVZs shall be reviewed and if necessary revised, as appropriate, at least every four years based on the results of the monitoring exercises carried out by the MS. The map of Figure 4 in Annex 6 shows the distribution of NVZs in EU-27. Some Member States have opted to apply mandatory measures to their whole territory\(^{41}\), or regions of their territory\(^{42}\), as NVZ. From 2008 to 2013, the percentage of territory to which mandatory measures applied notably increased in Romania, Belgium, Spain, Sweden and UK (European Commission, 2013b).

**Eutrophication**

Eutrophication consists in a progressive over-enrichment of water by nutrients, resulting in an increasing biological production leading to excessive algae and plant growth (e.g. microscopic algae (phytoplankton and diatoms), filamentous algae, macroscopic algae and higher plants). Eutrophication naturally occurs in water bodies filled in with nutrients over centuries but is also reinforced by excessive fertilisation or wastewater discharge into aquatic ecosystems. In potentially affected areas, the risk of eutrophication increases with high temperature, high light availability, and low water flow. This explains why eutrophication impacts surface waters, from lakes and rivers to saline lagoons and coastal water.

High density of these algae and plants can have dramatic consequences. Blocking sunlight and consuming dissolved oxygen, algae and plant bloom leads to low-clarity and anoxic bottom waters that disturb aquatic ecosystems and reduce the quality of the water environment. After their growth, the decay of algae continues depleting dissolved oxygen due to microbial decomposition but also releases toxic compounds in the water and more generally in coastal areas (e.g. deposition on beaches). In addition, some algae, especially the type of phytoplankton known as cyanobacteria, are of unwanted nature due to their toxicity and smell (Leng, 2009). Through water contamination and disruption of the balance between biotic communities, eutrophication represents a direct threat for public health (e.g. by making water non-potable) and biodiversity (e.g. by leading to the extinction of certain populations or through the development of invasive species). It also affects key economic sectors, such as fishery and tourism because of the inconvenient smells and colours, spoil landscapes, and restriction of economic and recreational (e.g. fishing) activities with high socio-economic costs (Chislock, et al., 2013; Bomsans, et al., 2005).

The major nutrients that cause eutrophication are nitrogen and phosphorus. Depending on the nutrient-demand of the algae and plants, the amounts of nutrients available in the water, and the climate and season, either nitrogen or phosphorus is expected to be the growth-limiting nutrient (Table 7). In the EU,

\(^{41}\) Austria, Denmark, Finland, Germany, Ireland, Lithuania, Luxembourg, Malta, The Netherlands, Slovenia

\(^{42}\) Wallonia (Belgium) and North Ireland (United Kingdom)
while nitrogen is mostly related to the eutrophication of estuaries and coastal marine waters, phosphorus is often found to be the growth-limiting nutrient in fresh surface waters (Bomans, et al., 2005).

**Table 7 – Nitrogen/phosphorus ratios (expressed in weight) for various limiting conditions in freshwater and estuarine/coastal waters**

<table>
<thead>
<tr>
<th></th>
<th>N-limiting (Ratio N/P)</th>
<th>Intermediate (Ratio N/P)</th>
<th>P-limiting (Ratio N/P)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freshwater</td>
<td>≤ 4.5</td>
<td>4.5-6</td>
<td>≥ 6</td>
</tr>
<tr>
<td>Estuarine/coastal</td>
<td>≤ 5</td>
<td>5-10</td>
<td>≥10</td>
</tr>
</tbody>
</table>

Source: (World Health Organisation Regional Office for Europe, 2002)

The phosphorus concentration in water is a relevant indicator of the degree of eutrophication (mainly for inland waters) and can be used to categorise waters among different classes of their trophic status (an oligotrophic system refers to low nutrient content, contrary to a eutrophic system that is particularly rich in nutrients). Thresholds above which phosphorus triggers eutrophication have been set, depending on water bodies characterised by different temperature, light availability, and flow. In Table 8, thresholds are expressed in phosphorus concentration and also in phosphate concentration because phosphate is the only dissolved form of phosphorus available for algae and plants.

**Table 8 – Thresholds of phosphorus causing eutrophication in lakes and rivers**

<table>
<thead>
<tr>
<th></th>
<th>Lakes</th>
<th>Rivers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phosphorus (P)</td>
<td>0.03 mg/L</td>
<td>0.06 mg/L</td>
</tr>
<tr>
<td>Phosphate (PO$_4^{3-}$)</td>
<td>0.1 mg/L</td>
<td>0.2 mg/L</td>
</tr>
</tbody>
</table>

Source: (Comité de Bassin Rhône-Méditerranée-Corse, 1996)

The trophic status of aquatic ecosystems substantially varies across EU. Differences in assessment methods (e.g. measurement parameters, class definitions) and data availability (e.g. lack or incompleteness of data) exacerbate the heterogeneity of trophic status between Member States. Table 9 reports a classification of the trophic status or rivers and lakes in EU 27, however harmonisation of methodologies and improvement in the coherence of monitoring results remains a challenge in this regard. According to the data reported under the implementation of the Nitrates Directive, high levels of eutrophication were observed in rivers and streams in Belgium and the Netherlands, followed by Czech Republic and Finland. High levels of eutrophication were also found in Lithuania and Luxembourg. In lakes, the highest percentage of eutrophic or hypertrophic stations was found in the Netherlands, followed by Denmark, Slovakia, Poland, Bulgaria and Belgium. For saline water, data is lacking for several Member States. Based on the available information, Belgium reported all its saline waters as hypertrophic while Bulgaria, Latvia, Lithuania and the Netherlands reported all saline stations as eutrophic. At the regional level, the Baltic Sea, the Black Sea, the coastal region of Brittany (France), and some Mediterranean coastal zones are particularly concerned (Schroder, et al., 2010; European Commission, 2013b)

**Table 9 – Breakdown of EU-27 monitoring stations by degree of eutrophication for the period 2008-2011**

<table>
<thead>
<tr>
<th>Trophic status</th>
<th>Ultra-oligotrophic</th>
<th>Oligotrophic</th>
<th>Eutrophic</th>
<th>Hypertrophic</th>
</tr>
</thead>
<tbody>
<tr>
<td>% of monitoring stations</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rivers and streams</td>
<td>20.6</td>
<td>35.4</td>
<td>16.3</td>
<td>6.3</td>
</tr>
<tr>
<td>Lakes</td>
<td>2.4</td>
<td>36.6</td>
<td>24.1</td>
<td>12.7</td>
</tr>
</tbody>
</table>

Source: (European Commission, 2013)
**Acidification**

Water acidification results from the dissolution of acids in water that release protons and lower the pH of water (Reuss, et al., 1987; Barker & Ridgwell, 2012). Two phenomena mainly contribute to such acidification: deposition of acids (especially sulphur dioxide $\text{SO}_2$ and nitrogen oxide $\text{NO}_x$) from the atmosphere to fresh surface water and the uptake of carbon dioxide from the atmosphere by ocean waters.

To a lesser but increasing extent, run-off and leaching from nitrate-saturated soils may significantly enhance water content in nitrates as acid anions contributing to water acidification. Excess nitrate especially leads to fresh surface water acidification during high-flow periods, while sulphate acid effect decreases with increased stream flow (Bernhard, 2010; Murdoch & Stoddard, 2010; Curtis, et al., 2005).

Water acidification potentially affects aquatic ecosystem balance as organisms require a specific pH to ensure efficient biochemical processes and metabolism rates. For instance, the density and the reproductive success of amphibian population are reduced in low-pH waters (Leuven, et al., 1986). The acidification reaction also produces mineral carbonate that can result in organism calcification (Barker & Ridgwell, 2012).

Whereas key chemical issues of nutrient excess primarily concerns groundwater bodies, key ecological issues concern surface water bodies.

**2.3.4 Impacts on soil**

Agricultural practices affect nutrient cycles, resulting in nutrient excess which in turn leads to impacts on soil, such as a decrease in soil organic matter, acidification, eutrophication, and loss of soil biodiversity. Various agricultural management practices described in section 2.2 contribute to nutrient excess by affecting the different pools of organically bound nitrogen, phosphorus and potassium within the soil. Section 2.1.2.1 describes soil conditions that may interact with agricultural practices and act as drivers of nutrient excess in soils, water, and air.

Nitrogen inputs affect soil organic matter levels and may affect soil fertility. Low SOM can cause more water and air losses of nitrogen from the nutrient cycle (Velthof, et al., 2011). When there are high levels of nitrates in the soil, $\text{NO}_3^-$ may likely be leached due to its solubility or denitrification may cause gaseous emissions of $\text{N}_2\text{O}$ and other reactive nitrogen into the atmosphere. The extent of these effects depends upon a range of factors, such as soil type, water levels and drainage (leaching occurs more frequently in soils with adequate self-drainage whereas denitrification happens more in water-logged soils), and the timing and type of fertiliser application (Jarvis, et al., 2011).

As for water, acidification may affect soil as well, leading to reduced soil fertility and microbial transformations in the soil, and potentially affecting crop growth (Marschner, 1995; Bolan, et al., 2003; Velthof, et al., 2011). Indeed, acidification can negatively influence phosphorus availability and restrict plant access to subsoil moisture; although crop type, soil type, weather and additional nitrogen sources in addition to the type of nitrogen fertiliser influence the occurrence of acidification (Bolan, et al., 2003). Light-textured sandy soils, loams with low soil organic matter content, soils with low clay content, and naturally acidic soils are most at risk to acidification, which if used for intensive agriculture, can rapidly occur when nitrogen fertilisers are applied (Davies, et al., 2006; Queensland Government, 2009). For example, ammonium-based nitrogen fertilisers applied beyond the plant requirements may contribute to acidification in soils that are naturally acidic, as well as through nitrate leaching out of the root zone (Fenton & Helyar, 2002). Removal of crop residues can increase acidification as well.

Additionally, excessive nitrogen inputs can cause eutrophication due to low C/N ratios within the soils’ organic matter (Velthof, et al., 2011).
Low SOM levels, acidification and eutrophication also decrease soil biodiversity (Velthof, et al., 2011). Negative impacts on soil organic matter in agricultural soils through excessive nitrogen levels can also inhibit their carbon sequestration capacity (Smith, et al., 2008). Cultivation of organic soils has strong environmental consequences such as contribution to climate change and eutrophication of waters.

Soil functions include filtering, buffering, and regulating services for water and air. Thus, poor nutrient management and subsequent effects on soil transformation processes can affect these soil functions and thereby also human health. For instance, nitrate run-off through soil erosion into water bodies from which drinking water is extracted can cause “blue baby syndrome”, a blood disorder in infants wherein the oxygen-carrying capacity of the blood is reduced (World Health Organisation, 2014).

2.3.5 Impacts on biodiversity

The impacts of nutrient surpluses on biodiversity and ecosystem services are mostly indirect, through the positive or negative impacts that are described above. It is important to underline that the impacts outlined below are often due to a combination of factors and nutrient surpluses cannot be made responsible of the total impact, though it contributes to them.

Impacts on microorganisms, insects, birds (including habitats)

In soil, both crops and wild plants generally benefit from additional nutrients, in particular from manure application. Nutrient inputs, in particular those from organic origins, enhance the abundance of biological regulators (for example bacterivorous nematodes), ecosystem engineers (earthworms) in the soil, and generalist predators above ground (Vickery, et al., 2004; Birkhofer, et al., 2008). High nitrogen and phosphorus levels from high use of fertiliser and manure lead to increasing mineralisation of organic matter in the soil. In particular, high fertiliser inputs increase bacterial decomposition and thus the amount of nutrients available for plants. However, high inputs of organic fertilisers provide more diverse soil microorganisms that are able to destroy organic matter and positively affect microhabitats necessary for microbes’ activity (Turbé, et al., 2010; Underwood, et al., 2013). By enhancing the microbial activity, it can increase the mineralisation of nutrients and thus potentially affect the nutrient assimilation of some plants.

Moreover, nutrient surpluses can cause changes in species composition, in particular in natural and semi-natural habitats that have generally evolved in low nutrient levels. High nutrient levels also tend to encourage aggressive plant species (e.g. Italian ryegrass) to outcompete other species which thrive in nutrient poor conditions. It can also create dense fast growing homogeneous grassland swards and crops, which are too tall and dense for birds to nest and feed in, e.g. skylark (Aluda arvensis) and lapwing (Vanellus vanellus) (Donald, et al., 2001; Underwood, et al., 2013).

In water, free ammonia has significant impacts, more than ammonium ion, as it is toxic to many fish even at very low concentrations (Martinez & Burton, 2009). Hence, nitrogen surplus may damage biodiversity even the in case of a low surplus. However, nitrogen in water is more present as ammonium form than ammonia, and acidification also moves equilibrium towards the ammonium form. In addition, an excess of nutrients from organic sources in surface water results in reduced oxygen dissolved in water, as the decomposition of organic matter uses oxygen.

An excess of nutrients, phosphorus in surface water and nitrogen in seas, can lead to phytoplankton proliferation that reduces available oxygen and prevents sunlight from entering water. Such phenomena (eutrophication) can affect biodiversity (water fauna and flora) and the decaying process of dead phytoplankton may lead to deoxygenated dead zones in which only a few bacterial species may survive in extreme case (Brandjes, et al., 1996; EEA, 2012a). Indeed, the decrease of oxygen levels, especially at depth, drives deep-spawning species into shallower water where they spawn with other species to
form hybrids. The prey for fish also becomes less diverse. Furthermore, the decrease of water clarity and the possible habitat change may disturb the species reproduction and nesting and induces a demographic decline. The selective pressure is not sufficient to maintain distinct species characteristics, leading to a reduced divergent selection. As a consequence, the deeper-spawning species become extinct, with the remaining species being hybrids. This may modify the balance of vegetation and animal species, facilitating the development of invasive species (McKinnon & Taylor, 2012).

Acidification caused, for instance, by nitrogen emissions has an effect on the solubility of soil components. Thus, it modifies soil structure and increases the availability of some toxic elements, especially aluminium and manganese (Slattery, et al., 1999). Excessive concentration of aluminium in soil water causes crop and tree failure, yield losses by interfering with plant capacity to absorb nutrients and water. Manganese has an effect on plant and tree growth, resulting in stunted, discoloured crops, reduced growth and poor yields (Johnson & Zhang, 1988). Other negative impacts may occur to animals feeding on plants, when plants absorb the heavy metals contained in soils (a characteristic that is used in phytoremediation). Although examples of poisoning of animals can be found in the literature (Henkin, 1974), there is no agreement on the extent of this impact.

**Impacts on ecosystem services**

Ecosystem services are the outputs from natural processes that benefit humans. The Millennium Ecosystem Assessment classified them into four categories (World Resources Institute, 2005):

- provisioning, such as the production of food and water;
- regulating, such as the control of climate and disease;
- supporting, such as nutrient cycles and crop pollination; and
- cultural, such as spiritual and recreational benefits.

The environmental issues due to nutrient excess (e.g. eutrophication, acidification, etc.) may negatively affect ecosystem services. For example, eutrophication leads to reduced possibilities of using water for drinking (a provisioning service), bathing (a cultural/recreational service in rivers, lakes or coastal areas), or fishing (economic or recreational activities) purposes. The attractiveness of eco-touristic regions can also decrease in case of excessive algal growth that prevents tourists from bathing, releases foul-smelling odours, and visually pollutes aesthetic landscapes. In case of acidification of soils, *Rhizobia*’s survival and persistence are affected, leading to reduced fixation of nutrients and water by plants and hence yield (a provisioning and supporting service). Also, protecting biodiversity allows maintaining natural pest control services. It avoid the decrease of yields due to animals and pathogens' damages and also reduces the dependency and costs associated with biocides (World Resources Institute, 2005).

### 2.3.6 Impacts on human health

Nutrient emissions, leaching, and run-off can lead to air and water pollution, which can come with substantial risks on human health and nuisances. Risks and nuisances vary with nutrient availability, volatility and solubility. While nitrogen easily circulates between air, water and soil compartments, phosphorus and potassium movements are rather limited. Hence, the probability of occurrence of potential impacts on human health and well-being may be higher for nitrogen.

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43 Bacteria from soil that can fix the nitrogen
Nitrogen

Nitrogen application to soils induces the formation of nitrites, nitrates and ammonium ions as well as nitrogen oxides (NO\textsubscript{x}) and dinitrogen oxide; as many molecules susceptible to affect – directly or indirectly – human health and well-being.

**NO\textsubscript{x} and ammoniac emissions: respiratory and heart problems**

NO\textsubscript{x} released into the atmosphere can impair human health due to potential reaction with ammonia, moisture, and other compounds to form particulate matter or with volatile organic compounds in the presence of heat and sunlight to form ground-level ozone.

Particulate matter can get deep into the lungs and cause serious respiratory and heart diseases, especially among fragile populations (children, elderly). Several problems have been related to exposure to particle pollution, such as: premature death in people with heart or lung disease, nonfatal heart attacks, irregular heartbeat, aggravated asthma, decreased lung function, and increased respiratory symptoms (irritation of the airways, coughing or difficulty breathing chronic bronchitis) (US EPA, 2013). These effects are in line with those assessed in the framework of the Cost Benefit Analysis for the Revision of the National Emission Ceilings Directive (AEA, 2010b). Fine particles are also responsible of reduced visibility (haze) in high concentration areas.

On the other hand, excessive concentrations of ground level ozone will also lead to respiratory problems (breathing difficulties, coughing, inflammation, lung diseases, etc.) (US EPA, 2013).

However, it should be stressed that main sources of NO\textsubscript{x} are the transport and energy sector (47 % and 21 % in 2011, (EEA, 2013e). While emissions from agriculture are not negligible, they are mostly due to petrochemical combustion (mechanisation, fertiliser production); emissions from organic and mineral fertiliser spreading remain rather low especially with regards to the vastness of agricultural surface area. For instance French NO\textsubscript{x} emissions from breeding and fertilisers application account for about 6 % of total emissions (CITEPA, 2013).

**Nitrites and nitrates: adverse effects in infants and indirect toxic effects**

Dietary nitrates and nitrites mostly come from vegetables and water, since nitrate has been accumulating in our water resources over the last half century, mostly due to fertiliser application.

In Europe, the Drinking Water Directive (98/83/EC) sets a maximum allowable concentration for nitrates of 50 mg/L (in accordance with World Health Organisation guidelines). This limit was set to prevent methemoglobinemia, in infants in particular. Methemoglobinemia is a disease affecting haemoglobin capacity to release oxygen and potentially leading to hypoxia, i.e. body whole or partial deprivation of oxygen (Fewtrell, 2004).

Ingested nitrates and nitrites undergo several chemical reactions in human organism, leading to the formation of N-nitroso compounds (NOC) in the stomach. These NOC are potential causative agents of specific cancers, diabetes, hypertension, increased infant mortality, central nervous system birth defects, spontaneous abortions, respiratory tract infections, and changes to the immune system (Ward et al., 2006; Fewtrell, 2004). However, since NOC formation in the stomach can be inhibited by dietary antioxidants, the direct relationship between nitrate intake and cancers is still debated (Ward et al., 2005).

Lastly, nitrates are also responsible for water eutrophication, in marine water especially, generating algae blooms. While living algae are harmless for humans, they can release disagreeable and even toxic substances when decomposing in anaerobic conditions. And when large amount of algae desiccate, they create sediment and the algae deposit can be several meter high, covered with a white
Dinitrogen oxide: indirect health effects due to climate change

Dinitrogen oxide being a gas with a high global warming potential (nearly 300 times higher than CO\textsubscript{2} potential), nitrogen is also responsible for determinants such as clean air, safe drinking water, sufficient food and secure shelter that may result in climate-change-related health diseases in the event of the disruption of these factors.

Phosphorus

High phosphorus dietary intake and serum phosphate concentration can cause damage on the cardiovascular system and induce higher mortality risk, especially for people with renal and cardiovascular diseases, as well as for the general public (Ritz et al., 2012). However, most of ingested phosphates come from protein-rich foods and industrially processed foods since polyphosphates are commonly used as additives. As an example, a 150g portion of meat contains 200 to 300 mg of phosphate while drinking water concentration hardly exceeds 1 mg/L. Excessive phosphorus supply on agricultural land cannot thus be charged with adverse health effects. Especially as phosphate can be added to domestic drinking water to reduce lead and copper contents and help meet EU Drinking Water Directive standards (Comber, 2011). Neither the World Health Organisation nor the European Union has set a limit for phosphorus concentration in water. In addition, phosphorus may be responsible for eutrophication of fresh surface waters. The resulting algae growth and then decomposition can affect the quality of freshwater for drinking purposes and release toxic gas likely to cause respiratory concerns (World Health Organisation Regional Office for Europe, 2002).

Potassium

Excessive potassium intake can lead to hyperkalaemia, causing nausea and/or heart dysfunction (irregular pulse and even heartbeat stops). However, with proper kidney functions, hyperkalaemia is unlikely to happen since kidneys normally remove excess potassium from the body (A.D.A.M. Medical Encyclopedia, 2011). Additionally, it is improbable that excessive potassium supply on agricultural lands leads to such a disease since potassium is a rather stable element. Potassium is mostly fixed in soil or plants, the fraction available for run-off being very low (about 0.1 to 0.2 % of potassium supply) (Faculty of Land and Food Systems, 2004). Soil potassium supply is thus unlikely to induce adverse effects on human health.

Trace elements

Finally, it should be mentioned that inorganic fertilisers may also contain hazardous trace elements, such as toxic heavy metals or radioactive compounds. These trace elements can enter human body after having been fixed by plants and then ingested, or leaching into water resources. A high
concentration of these elements can induce a wide range of diseases in human beings, such as poisoning, gastrointestinal effects, reduced immune function, Parkinson-type syndrome, etc. (Fraga, 2005).
2.3.7 Synthesis of the impacts (environmental consequences of nutrient excess)

Figure 17 – Synthesis of the impacts related to nutrient excess

Source: Own compilation
3. Solutions to reduce the impacts of nutrient losses

3.1 Review of possible solutions

The previous chapters highlighted how the inadequate management of mineral cycles may cause detrimental impacts on human health and the environment, leading to reduced revenues from various economic sectors and/or increased expenditures for public authorities and other sectors. These findings call for the implementation of concrete actions for a more efficient use of nutrients and for closing mineral cycles. Key strategies in this respect include:

- Reducing the sources of contamination, i.e. reducing the use of nutrients;
- Increasing the productivity of nutrients\textsuperscript{44}, i.e. recycling or reusing nutrients; and
- Better controlling contamination pathways, i.e. limiting the transfer of nutrients between environmental compartments.

A range of actions to reduce nutrient surpluses and its impacts could be identified throughout the nutrient life cycle, in crop, livestock, and mixed farming systems. Two key sources of nutrients in the environment are mineral and organic fertilisers, including manure. A number of actions can target the nutrient production (e.g. manure appropriate breeding systems and feeding techniques), processing, storage, application (e.g. manure spreading, manure transfer), and/or waste collection and recycling (e.g. crop residues incorporation). Other measures aim to create favourable conditions for reducing risks of contamination, for instance through appropriate soil (e.g. conservation tillage, soil overage), crop (e.g. crop rotation, residue management) and water management (e.g. irrigation systems and drainage). The synthesis of the analysis of the good practices is presented in Annex 15.

3.2 Solutions for livestock production

3.2.1 Reducing the sources of contamination

Reducing the source of contamination implies the reduction of manure production, decreasing the amount of nutrient excreted (for the same amount of manure produced), or the transfer of manure to other locations.

Reduce manure quantity

Manure production can be reduced either by limiting / decreasing the number of livestock per unit area or using appropriate feeding practices.

\textsuperscript{44} Productivity is here related to the amount of goods and services such as biomass or energy that can be produced for a given amount of nutrients
**Decrease livestock density or select livestock with high genetic merit**

Reducing the number of heads per unit area by a certain percentage or select livestock that produce more milk or milk for the same quantity of feed decrease the amount of manure produced and nutrient excreted by the same percentage. Therefore, it reduces NH$_3$ and N$_2$O emissions and the possible risk of NO$_3$ and PO$_4^{3-}$ run-off and leaching and related impacts on air, water, soil, biodiversity, and human health. CH$_4$ emissions are also reduced. Indirectly, the measure also reduces the emissions of N$_2$O due to the reduction from unnecessary fertilisers as less feed and forage are needed.

The limitation or reduction of the agricultural pressures at source is particularly relevant for farms in nutrient saturated areas or for farms in NVZ where the amount of manure produced exceeds the amount of manure that can be applied in the zone according to the threshold(s) established in the regional nitrate action plan. Notwithstanding that several approaches can be in place to manage the surplus of nutrients, agricultural pressures far beyond the carrying capacity of the environment might result in being unable to cope with the adverse environmental effects and in the impossibility of closing the mineral cycles.

On the other hand, limiting or reducing the number of livestock can have a serious impact on the profitability of the farms as it would lead to a reduction of the quantity of meat and milk produced. In particular, this could jeopardize the viability of small holdings considering the necessary material and infrastructure investments while larger holdings benefit from economies of scale. Consequently, decreasing the number of heads of livestock is likely to be very challenging for a number of farmers. In addition, the decreased production can affect the economic performance of the region if agriculture is an important base of the local economic activities. This may also cause food security issues as it may require the importation of agricultural products from other regions. The selection of animals that produce more milk or meat than others, which is commonly done by farmers, can help reducing the density of cattle while preserving the economic viability of the farm.

**Use appropriate feeding practices**

Feeding practices can influence the amount of manure produced. Adjusting the amount of feed according to the growth stage and possibly reducing the quantity of feed provided, decrease the amount of manure produced (Sutton & Lander, 2003). In intensive indoor livestock rearing, it is common practice that animals are provided with excessive fodder, which leads to overfeeding and increases the amount of manure produced. Low protein intake decreases the water requirement and hence, the volume of slurry to manage (Ward & McKague, 2007; Rasby & Walz, 2011): for finisher pigs a one percentage point reduction in protein intake results in a decrease of water consumption by 2 to 3 % and of slurry volume by 3 to 5 % (Kaasik, 2012).

Implementing such measures requires farmers to group the animals according to their sex, age and production stage as feed requirements differ accordingly (Carter, et al., 2012b; Spiehs, 2005). When keeping animals separated, feed can be better adjusted. The farmers will need to regularly adjust feeders and provide regular maintenance of the feeders, bunks and drinking troughs (Spiehs, 2005).

This measure requires a moderate level of know-how from the farmers. Firstly, an appropriate feeder design for age and type of animal has to be set up (Spiehs, 2005). This also requires knowledge on the availability of nutrients in feed ingredients on the base of which diets have to be formulated (Sutton & Lander, 2003). For the farmers, the measure will potentially lead to an increase in revenues thanks to the achieved savings (reduced need for feed and fodder), although the costs for the regular analyses of fodder may partly reduce the expected economic gain.

The selection of livestock with high genetic merit also allows obtaining the same yield for a lower amount of feed provided. For dairy cattle for instance, manure production increases as the milk production increases. However, the relationship is not very strong which means that it is possible to have high milk
production without significant change in the amount of manure produced. Therefore, the amount of nutrient excreted per litre of milk is lower for highly productive animals than for conventional cows (Cellier & Peyraud, 2012).

Reducing nutrients in manure

If the amount of manure produced on farm cannot be reduced, another solution is to decrease the amount of nutrients contained in manure. To do so, farmers can use appropriate feeding practices or remove nutrients through specific manure processing.

Use appropriate feeding practices

Feeding practice allows decreasing the amount of nutrient excreted by (1) adjusting the feed (and hence the nutrients) to the animal’s needs and (2) increasing the assimilation of nutrients by animals.

In intensive agriculture systems, livestock feed is often provided in excess of the animal needs. The adjustment of the quantity of animal feed to livestock’s needs at each growth stage helps to prevent losses of N and P by decreasing excretion and by limiting unconsumed feed (Sutton & Beede, 2003). Diet formulation closer to animals’ requirements can reduce both N and P excretion by 10 to 15% (Ferket, et al., 2002; FASS Environment, Waste Management and Ecosystems Committee, 2001). Multiphase feeding helps to reduce nitrogen excretion by up to 15% for growing pigs and up to 50% when combined with other feeding practices such as the adjustment of protein content and the use of amino acids (Ferket, et al., 2002; Dourmad, et al., 2009).

Farmers can also increase the digestibility of feed by decreasing the forage to concentrate ratio, adjusting the amount of protein, sugar and P in the feed provided, adding additives such as amino acids (required for pigs in case of crude protein reduction in the diet), phytase or antibiotics and processing feed by decreasing the feed particle size or ensiling forage for instance. For pigs, reducing crude protein by 3-4.5% can lower excreted N by 30-40% and ammonia emissions by 40-60% (Sutton & Beede, 2003; Carter, et al., 2012b). The addition of phytase can decrease P excretion by 20 to 60%, depending on the animal type and its growth stage (Kaasik, 2012). Processing feed can reduce the amount of nitrogen in manure by up to 20-25% (Carter, et al., 2012b). However, in practice the reduction potential is lower since these experimental results are often obtained from a highly unbalanced diet (Martin & Mathias, 2013). In particular, the protein content should be carefully adjusted to the animal needs to avoid any decrease of the yield (e.g. the quantity of meat, milk, eggs produced) if the quantity of ingested protein is not sufficient to allow the animal metabolism to attain the usual yield.

The basic requirements for the implementation of this measure are similar to the ones for the feeding practices that aim at decreasing the amount of manure produced. In addition, sufficient knowledge is needed for the application of the measure in order to ensure that livestock production will remain constant (Knowlton, et al., 2004). For example due to variable compositions of feed, routine laboratory feed analyses and a ration formulation program are necessary to adjust diets and maintain minimum nutrient excretions (Sutton & Lander, 2003). For the farmers, the measure will potentially lead to an increase in revenues due to the savings achieved from reduced feed. The costs for the regular analysis of feed and the purchase of additives may decrease this gain.

Remove nutrients from manure

Nutrients can be removed from liquid manure by a nitrification/denitrification process that transforms the NH$_4^+$ to inert gas (N$_2$). The maximum N-removal in this process can attain up to 70% of the initial amount of N (Belin & Guiziou, 2004). While this technique is very effective, it requires a high-level of know-how from farmers since numerous parameters need to be controlled (temperature, bacterial loads, aeration, and manure composition). The economic investment and operating costs are high (partly due to high
energy consumption), therefore this technique is more easily implemented in farms with high income (Institut de l'élevage, IFIP, ITAVI, 2010) or through collective initiatives. However, even for individual small/medium farms, it can be worth studying the profitability of such investment compared to the cost of manure transportation over long distances (Lyngsø Foged, et al., 2011).

### 3.2.2 Improving nutrient efficiency

When animals are kept indoors, manure can be easily collected and then used as a fertiliser or for energy production. The following measures allow increasing the efficiency of the manure produced and collected.

**Processing manure to facilitate its use**

While theoretical manure composition is often known by farmers, in practice it can vary according to many factors such as animal type, bedding, ration, storage/handling, environmental conditions, field application method, and age of manure (Peters, et al., 2003). The nutrient content and form in manure can also evolve during storage due to the microbial activity. Thus, manure can be less easy to use than mineral fertilisers that provide a stable quantity and quality of nutrients. Processing manure can help to concentrate the nutrients or transform them into a stable form that can be directly assimilated by plants. It can also decrease the volume of manure that needs to be stored and thus reduce potential ammonia emissions during storage. For all processing measures, the volume and weight of manure are decreased, which reduces costs in case transportation is needed. In addition, the application of high-quality manure-based fertilisers reduces the needs of chemical fertilisers as well as the consumption of fossil fuels used to synthesise them. Considering the high nutrient content of the processed products, careful field application is required to avoid over-fertilisation.

Separating the solid and liquid fractions of manure is used to remove nutrients from the liquid (clarified) fraction and export the solid fraction outside the farm, and to ease manure management during storage and during spreading by avoiding the risk of clogging. The liquid part contains nitrogen and potassium and the solid part contains phosphorus and nitrogen. Nitrogen from the liquid fraction is in the form of urea/ammonium salt and is rapidly available to plants. The N/P ratio of this fraction is particularly well-suited for forage crops. The nitrogen in the solid fraction is in the form of macromolecular compounds, available after mineralisation. The solid fraction can be directly applied to soil or composted while the liquid fraction can be used in irrigation systems. Nonetheless, particular attention is required while applying the liquid fraction to avoid an over-application of ammonium and possible leaching. Several separation technologies are possible: sedimentation by gravity, coagulation-flocculation, filtration, etc. They require the purchase of treatment machinery whose investment and operational costs vary according to the technology used (Lyngsø Foged, et al., 2011).

Composting solid manure transforms nitrogen into an ammonia form while reducing the volume of the original material (Barrington, et al., 2002; Agro Business Park, 2010; Baltic deal, 2013). Composting can be easily implemented on-farm and generates a product that is easily exportable off-farm (Beline & Guiziou, 2004; Lyngsø Foged, et al., 2011). Manure can be composted in open-air piles or in containers. It only requires regular aeration by pushing or pulling air through compost piles. The use of closed systems helps to reduce ammonia emissions (up to 40 % N lost through ammonia form) but can favour nitrous oxide emissions.

Drying and pelletising solid manure improves manure storage by producing a stable and very concentrated fertiliser. Since solid manure is quickly dried, it avoids the ammonia emissions and odour related to manure storage. The equipment from other technology such as feed production can be adapted to dry and pelletise manure, avoiding investment in new equipment. The operational costs are
important, in particular due to the high energy consumption, which furthermore generates CO$_2$ emissions (Lyngsø Foged, et al., 2011).

The addition of magnesium ions to liquid manure with a basic pH results in the precipitation of nitrogen and phosphorus in struvite that is a very stable crystal. The struvite can then be used as a fertiliser that is easily stored. The slow release of nutrients from struvite prevents from the excess supply of nutrients that crops cannot assimilate (Baltic Manure , 2011). This operation requires a reactor, implying a collective initiative to purchase the reactor or the use of an external provider (Lyngsø Foged, et al., 2011).

**Use of manure to produce energy**

Manure can also be transformed into energy. Manure is digested by microorganisms in anaerobic conditions in a digester – a sealed tank or container with the appropriate biological conditions to speed up this digestion process. The digestion results in emissions of biogas that include methane. The latter can be used for energy production. The heat and electricity produced can be used on the farm and may reduce the expenditure related to energy on the long term while saving GHG emissions.

The digested manure (“digestate”) produced by the biogas reactor can be spread to the soil. The digested manure contains highly concentrated nutrients. Nitrogen is more easily assimilated by crops than undigested manure because the anaerobic digestion (AD) mineralises the organic, bound nitrogen into inorganic nitrogen. As a consequence, the digested manure has a high risk of leaching in the short term in case of over-fertilisation. On the other hand, the digestate has a lower risk of leaching in the long term than for untreated manure due to the lower share of nitrogen bounded to soil that can be progressively released (Børgesen & al, 2013; Homan, 2014). However, as for other processes, careful use is required while applying the treated product into field to avoid over-fertilisation. Furthermore, since the digested manure is stable with a lower volume than before digestion, it also indirectly avoids ammonia emissions during storage.

This measure is not applicable directly for individual farms mostly due to the investment and operational costs involved. Indeed, small reactors are very costly and need sufficient substrate (manure, slurry or other organic waste) to be profitable. Farmers can choose to team up to benefit from a community reactor which would guaranty a sufficient amount of manure and divide the investment costs. However, it results in additional transport of the manure from the farm to the plant. It should be noted that slurry requires large digester volumes and the biogas production per unit of digester volume is low (Baltic Manure, 2013). Indeed, these systems need other substrates, apart from the pig slurry, to obtain a significant methane production because of the low organic charge of the pig slurry. This may result in the conversion of grassland to arable land for the production of these substrates such as maize, potentially leading to environmental impacts due to a higher use of fertilisers and pesticides, to a loss of soil carbon and to a loss of biodiversity. In addition, the substrate combination should be thoroughly studied in order to avoid inhibitions in the gas production which could result from the mix of substrates. Lastly, one of the main obstacles to the uptake of biogas is also public opposition to the opening of new plants. Biogas plants can generate odour and increase road traffic, which are considered a nuisance for the neighbouring areas.
3.2.3 Controlling the contamination pathways

Measures related to housing

Choose adequate bedding systems

For cattle, the litter management and the choice of the bedding material influence the nutrient rate. The use of litter provides a barrier between urine and air and thus decreases NH\textsubscript{3} emissions. In addition, the choice of the bedding material influences the C/N ratio and the mineralisation rate of nitrogen. Increasing the C/N ratio limits nutrient losses during manure application.

Frequent removal of the litter and limiting turning and mixing of manure also decrease the ammonia emissions. Limiting turning and mixing can reduce ammonia emissions by 40-60 % (Joint Research Centre, 2013a). However, this induces anaerobic conditions that increase N\textsubscript{2}O emissions and increase the temperature of manure and thus methane emissions (Joint Research Centre, 2013a). This measure can also be difficult to implement for animals that naturally turn the litter such as pigs.

A deep litter system also decreases ammonia emissions due to the compaction of the litter that limits the contact of manure with air. In addition, when the manure is applied it increases nutrient assimilation by crops by increasing N mineralisation and thus reducing the risk of nutrient losses. However, this system also results in high N\textsubscript{2}O (and CH\textsubscript{4}) emissions due to the anaerobic conditions in the compacted litter (Joint Research Centre, 2013a).

Moreover, the litter management and the choice of the bedding material significantly influence animal welfare and hygiene. Deep litter systems were widely used in the past for dairy cows, but most farms have moved to cubicles to improve health and milk quality (bacterial count).

Lastly, it is also essential to improve the construction designs to minimise the consumption of water when cleaning the buildings and, as a consequence, the final volume of slurries. In this sense, it is important that the buildings have moats with a suitable slope to easily carry the slurry from the buildings to the storage vessels using a lower quantity of water. The use of high pressure cleaning reduces the amount of water needed.

Reduce ammonia volatilisation during manure collection

The installation of a partly slatted floor facilitates the collection of manure and reduces the loss of nutrients via volatilisation. Covering 50 % of the floor area generally results in 15-20 % lower ammonia emissions (ADAS, 2011). In addition, the N content of the collected manure is increased which reduces the need for manufactured fertilisers (when the manure is used as a fertiliser). Lastly, this measure has positive effects on odours by limiting the residence time of manure in barns and improves the sanitary conditions and comfort in the building.

The main obstacle to the implementation of this measure at farm level is the amount of structural work required, it is most appropriate for new, purpose-built installations as the retro-fitting of existing housing units can be costly (Martin & Mathias, 2013). The implementation of this measure can be enhanced in case of a ban on fully slatted floors for animal welfare. Animal welfare is an important aspect to consider in relation to housing measures. Often better welfare conditions are associated with an increase in emissions (e.g. increased soiled walking area per animal or changing from tied stall to cubicle) (Gothenburg Protocol Guidance, 2011). Conversely, changes in building design in order to reduce...
emissions can be an important opportunity to introduce better animal welfare requirements at the same time, thereby reducing the costs of these measures.

**Measures related to pasture**

**Optimise grazing management**

When livestock graze, manure is deposited on the soil which leads to a localised increased in nitrogen surplus. In addition, outdoor livestock have a tendency to leave their excrements in areas where the animals concentrate, leading to very low nutrient utilisation and crop uptake and increased risk of run-off and leaching (EEA, 2000). Emissions of ammonia from livestock can be greater when grazing than when kept indoors as excreta deposited are not incorporated. The use of woodchip pads in case of outdoor wintering of cattle can help decreasing the ammonia emissions due to the local excess of manure deposition by modifying the C/N ratio (ADAS, 2011). The biological nitrogen fixation could also be increased by having more legumes in the pastures, which thus would reduce the level of fertilisation with manure and chemical fertilisers. As the biological nitrogen fixation in grassland as opposed to fertilisation does not involve direct emissions of nitrous oxide, this could reduce nitrous oxide emissions from grasslands.

More indoor housing and less grazing could reduce the volatilisation of ammonia through deposition in the pasture and the leaching of nutrients contained in the urine directly deposited in the pasture. It also allows better control of feed as well as better collection of manure for future use as a fertiliser. In addition, pastures that are not used for grazing are cut. It was estimated that a shift from grazing to cutting reduces leaching from the root zone by 0 - 25 kg N / ha on clay soils and by 12-70 kg N / ha on sandy soils (Børgesen & al, 2013). However, keeping animals indoors would result in some adverse impacts. Emissions of CH₄ would be increased as more manure is stored (ADAS, 2011). This practice is also not compatible with practices such as organic farming. Moreover, when pasture is cut, nitrogen contained in the sward is removed from the field (Børgesen & al, 2013) which may then require the application of fertiliser. Lastly, keeping animals indoor would be associated with additional costs of housing, manure management, and forage production. Farmers would incur costs of additional floor scraping and slurry handling, together with additional silage production to feed the housed livestock.

**Measures related to storage**

**Increasing storage capacity**

A high storage capacity allows for the manure produced during the year to be kept until it is necessary (i.e. until the appropriate period in spring and summer). If the amount of manure produced exceeds the storage capacity, farmers may have to apply manure in the field, by doing so outside of the appropriate periods, the crops may not be able to assimilate the manure applied, leading to an excess of nutrients in soil that could be easily leached. The Nitrates Directive requires that farmers in NVZ comply with a minimum storage capacity, defined by each Member State, that “exceed the capacity required for storage throughout the longest period during which land application is prohibited”. This storage capacity is not required for areas other than the NVZ. The farmers can also store the solid manure directly on field. In that case, a concrete floor can help avoiding losses from leaching while covering heaps reduces ammonia emissions and avoids losses from run-off and leaching due to precipitation.

**Making storage facilities impermeable**

In case of temporary storage of manure on field, preventive measures (waterproof layers, covers, requirements on dry matter content, distance from watercourses, etc.) should be put in place to minimise losses from leaching. Covering heaps also reduces ammonia emissions and avoids losses from run-off and leaching due to precipitation.
The building of impermeable floors (e.g. concrete) in case of temporary storage of solid manure in field can decrease losses of nutrients via run-off to the soil by up to 5% (ADAS, 2011). As a result, the nutrient content of the manure is improved and, when used as a fertiliser, the need for additional nutrients (e.g. via inorganic fertilisers) is reduced. However, a concrete floor may be difficult to build in field. In addition, the renovation or building of manure storage equipment requires high investment for limited nitrogen leaching savings (< 5%) (ADAS, 2011). Unless this is coupled with an increase of the board or tank capacity, farmers are not likely to implement this measure, in particular for small holdings with limited income. Other types of waterproof floors such as a floor covered by plastic liner may be a more affordable option.

The storage of liquid manure in proper tanks is necessary to avoid nutrient leaching. When stored in simple ponds, nutrients in slurry are easily leached. Covering the slurry reservoir limits the contact between manure and the atmosphere. This way, ammonia volatilisation is partly controlled by creating a buffer zone above the reservoir. Overall, covering slurry storages can reduce GHG and ammonia emissions to air from 40% (floating cover) to 80% (rigid store cover). favouring the formation of natural crust can, to a lesser extent, also reduce volatilisation. In addition, the crust favours methane oxidation as microorganisms in the crust convert methane to CO₂, thereby reducing methane emissions (Martin & Mathias, 2013). Additionally, tank covers prevent rain from entering outdoor tanks and diluting manure. By reducing the dilution, covers also provide longer storage capacity and the quantity of liquid manure is not uselessly increased by rainwater. However, implementing an artificial cover may imply changing the filling system of an entire tank which is expensive and economic issues are often an important barrier to the installation of new systems and equipment.

Inhibiting volatilisation

Reducing the temperature and/or pH of the stored liquid manure can reduce the transformation rate of organic nitrogen to mineral nitrogen. The volatilisation rate of ammonia increases from 15% up to 38% when temperatures increase from 17°C to 28°C. Cooling of the slurry also contributes to a reduction in methane emissions (Dourmad, et al., 2008). For each unit decrease of the pH of pig slurry, the ammonia emissions decrease by 45% (Canh, et al., 1998). In addition, experiments at both laboratory scale and under more realistic storage conditions have documented a long-lasting inhibition of methane production by up to 90% depending on the acid used (Olesen & al, 2013; Fangueiro, et al., 2013). As a result, the N content of manure is higher as less is lost through volatilisation. This parameter must be taken into account in order to avoid overdosing while spreading, which otherwise can lead to leaching.

While the reduction of pH is easy when using sulphuric acid, the decrease of the temperature may require infrastructure changes at farm level, for example if the slurry tanks are located under the housing units they may be warmed by the proximity of the livestock. This measure is applicable to indoor pigs and cattle farms producing slurry (i.e. where livestock is not kept on a deep litter system). In addition, the storage units need to be properly insulated so that the temperature does not vary in accordance with the exterior temperature.

3.3 Solutions for crop production

3.3.1 Reducing the sources of contamination

Fertilisation aims at providing crops with the quantity and quality of nutrients they need in order to grow properly and ensure a good yield. The first lever to avoid or reduce nutrient loss is to only apply what crops need. The following measures contribute to the achievement of this objective.
Develop and improve a fertiliser plan for N and P

In order to adjust the nutrient quantity to match crop needs and to decrease the amount of fertiliser applied, farmers should develop and/or improve their fertilisation plans for both N and P, including in areas outside of NVZ.

To do so, the farmer must calculate the nutrient balance of the farm, taking into account (Latvian Rural Advisory and Training Centre, et al., 2013):

- The type of crops and related needs for each plot according to the growth stage;
- The nutrient content in the soil by considering the previous balance and/or doing soil sample analysis in autumn after harvest or in early spring before fertilisation, with the analysis being performed on-farm or in a laboratory;
- The organic fertiliser/mineral fertiliser ratio that should be applied, considering the crop needs, the carbon content of the soil and the organic fertiliser applied, manure availability, the environmental conditions, the farmers financial capacity and the farmer’s preferences to use organic and/or chemical fertilisers (e.g. organic farming). The dose of the two types of fertilisers should be estimated in parallel to make sure that the total amount of nutrients provided fulfils the crop needs, taking into account the C/N ratio. In some circumstances such as a lack of availability of manure, it may be more convenient and/or economical to purchase inorganic fertilisers.
- The nutrient content of the organic and mineral fertilisers applied.

This measure is essential to avoid or reduce nutrient excesses and to close the nutrient cycles. Its implementation involves a certain level of technical understanding to conduct the analysis. Farmers can easily be helped by advisory services to improve their fertilisation plan. The measure is a source of costs, for example, for the analysis of soil samples. More broadly, as long as fertiliser prices are high, there are incentives for farmers to optimize input of organic fertilisers: as a reduction of mineral fertiliser purchased is expected, the costs tend to level out for the farmer (Newell Price, et al., 2011). In the event of excessive manure that cannot be applied on the field as the nutrient demand is already covered, additional revenue might be achieved through the trade of this manure in manure stock exchange (see above). As long as fertiliser prices are high, there are incentives for farmers to optimise input of organic fertilisers. However, in case of a lack of demand for manure from other farms, farmers may have to pay to get rid of this manure.

Precision farming can also be used to elaborate a very accurate fertiliser plan, by providing data on the amount of fertiliser that should be applied that may vary within a single plot. This technology allows decreasing the amount of fertilisers required and decrease the impacts related to the excess of nutrient compared to crop needs. More and more precision tools are proposed by private companies but this technology is still not accessible to all farmers considering the high investment costs.

Apply fertilisers at the right time

Applying fertilisers at the right time, in particular organic fertilisers, is essential to avoid any nutrient losses. When choosing the application timing, several parameters should be taken into account.

The nutrients needs of a crop vary according to the crop growth stages. Thus the quantity of fertilisers applied should be adjusted at each stage to avoid any over-fertilisation.

Generally, fertilisation with solid manure is only possible before sowing or after harvest. Slurry can also be applied in later stages, when crop demand should be considered. This also opts for split application that can reduce nitrate losses by up to 20 % for slurry application (Czymmek, et al., 2005) and P losses by 10 % (Bruulsema, 2005). Environmental conditions also need to be taken into account when applying organic fertiliser. For instance, ammonia emissions are enhanced due to high temperatures, wind and
solar radiation, and leaching takes place during precipitation events or in coarse-pored soils (high sand content). In particular, avoiding the period with the higher risk of leaching has led to the Nitrate Directive requirements regarding closed period in winter.

Lastly, the spreading of manure may also be linked to the speed at which the manure storage capacities are filled up. For example, farmers spread manure in autumn in order to make room in the storage units for the period of the manure application ban in winter. This type of manure management leads to unused nutrients in the soil that are leached or washed out and should be avoided.

The use of ATMS\textsuperscript{45} is recommended. The implementation of the measures requires knowledge of crop growth stages and the respective nutrient requirements. Furthermore, the farmers need to have good knowledge on the environmental conditions to evaluate the right timing for manure application. The use of ATMS requires substantial knowledge of environmental data. For example, the ALFAM model requires input of soil water content, slurry type, slurry dry matter content, total ammoniacal nitrogen (TAN) content of slurry, and application rate (Lalor & Lanigan, 2010). The vast technical requirements build a potential barrier to usage of such systems. However, most farmers already have knowledge on these issues. A co-benefit is realised in the smaller amount of fertiliser used.

**Manure transfer**

An option for farms with a surplus of nutrients is to transfer manure to other farms with a capacity to accept it. This allows optimising manure use, substituting chemical fertilisation (in farms receiving manure) and decreasing environmental problems by preventing excessive manure (in farms providing manure). Compared to chemical fertilisers, manure is less efficient and has a different agronomic behaviour, but also has the advantage of containing organic matter. Thus, transferring manure to regions with low soil organic matter content will improve their organic matter balance (better structure and the water holding capacity of soil).

The success of this measure depends on a number of factors.

- The key aspect is the availability of land for the surplus manure to be applied. This depends on legal (in NVZ the application of nitrogen from manure is limited) and agronomic (nutrient need depending on the type of crops, etc.) considerations as regards application thresholds.
- The existing legal framework for manure transfer differs across Member States. According to the EU regulation 1069/2009/EC, raw or processed manure exported to other Member States, they must have been treated at least 70°C for 60 minutes in an approved biogas, composting, or technical plant. In addition, EU Member States have specific additional regulations. For instance, in some states in Germany, poultry manure has to be sterilised before transport. In France, imported manure products are only accepted if they have been composted.

Manure and chemical fertilisers are substitutes to a large extent, however they have different characteristics in terms of efficiency and consequent leaching and runoff potential. The farmers’ preferences for a type of nutrient source or another will also be driven by the price. The price of manure including transportation costs should be more attractive than the price of a chemical fertiliser. Transportation costs increase with distance. A possible cost reduction of transportation costs of manure transfer can be the transport of the solid fraction after separation of the slurry. Thus, the transport cost per unit of nutrient content decreases significantly. In the Netherlands, there are currently negative

\textsuperscript{45} ATMS are mainly computer based models which calculate for the average regional environmental conditions (rainfall, wind speed, radiation, soil conditions etc.) the amount of nutrients lost during and following application.
manure prices, which mean that a farmer with a manure surplus pays manure-deficient farmers for the application of its manure.

The future of this measure is tightly linked to the evolution of the price of mineral fertilisers, transportation costs, quality of manure, processing techniques and legal framework in the MS. Manure transfers would require an important organization of the sector in terms of collecting manure, processing, transport, sanitary controls, etc.

3.3.2 Improving nutrient efficiency

Incorporate the residues

Incorporating crop residues enriches soils in organic matter. In general, most farms do not need to apply P because of the high P levels from crop residues that remain in the soil (BERAS International, 2013). Removing the residues from soil may require the addition of fertilisers to compensate the missing nutrients, depending on the soil’s nutrient content. However, if the residues are legumes crops, it is important to ensure that the ground is not left fallow following the incorporation of legume residues since it is high source of nitrogen that can result in rapid losses in case of excess of N. Residues such as straw have high C/N ratio and help to immobilise nitrogen that is then slowly released. It can also prevent soil from erosion and thus in particular P losses. In addition, incorporating crop residues has a positive impact on soil structure and soil water retention capacity, which in turn reduce the risk of nutrient leaching.

Incorporating the residues can be beneficial for soils which are well aerated or which contain a high proportion of clay (INRA, 2012). Tillage can help to increase the decomposition of residues and release the mineralised nitrogen. Farmers should take into the carbon content of the residues. Indeed, if the C/N ratio is too high, too much nitrogen is immobilised and not available for plant uptake. The reduced need of fertilisers for the next crops reduces the costs for farmers, although the residues could have been sold for energy production.

Grow perennial energy crops

Perennial bio-energy crops such as willow, poplar, miscanthus and other grasses with a long cultivation period have a permanent, deep root system and a long growing season, which ensures an efficient use of nutrients (Børgesen & al, 2013). The effect on nitrogen leaching when converting an area to perennial bio-energy crops depends on the cropping system that the energy crops replace. It has been estimated that for upland soils, N leaching would be reduced by 15-35 kg N per hectare for clay soils and by 40-60 kg N per hectare for sandy soils (Knudsen & Lemming, 2012). However, N leaching would increase in case of conversion of pasture to arable land for energy crop production.

In addition, extending the perennial phase of crop rotation reduces N₂O emissions since these crops require less fertiliser. Their extensive root system also helps to reduce erosion and thus nutrient loss, and phosphorus in particular. However, some species can require significant water quantities. Furthermore, the switch back from perennial to annual crops needs attention with regards to the soil CO₂ emissions. Indeed, the soil structure and soil water content is modified. Less intensive tillage, such as no-till or strip tillage, along with careful irrigation management will reduce soil CO₂ emissions in transitional land (Jabro, et al., 2008). Moreover, the production of energy crops can have consequences on the food supply due to the increased use of land for the production of energy crops (ADAS, 2011).
Use catch crops

Catch crops are types of cover crops which have a positive impact on soil quality. Integrating catch crops and legumes in crop rotation allows increasing the biological nitrogen fixation. Thus it reduces the nutrient losses through leaching, decreases the need for organic and mineral fertilisers and reduces erosion. It is estimated that N leaching on average can be reduced by 25 kg N per ha when cultivating well-established non-nitrogen-fixing catch crops after common agricultural crops fertilised with recommended amounts of mineral fertiliser or manure (Børgesen & al, 2013). However, this can limit the possibility to apply manure which can be an issue in saturated areas.

Legumes can also be used as catch crops. Legumes management is a bit more difficult. Indeed, legumes also fix nitrogen from the atmosphere. In case of incorporation, legumes restore to the soil the nutrients caught during its growth and the nitrogen fixed from the atmosphere. Both inputs should be taken into account in the budget of the following to avoid over-fertilisation. Hence, the input of nutrients from legumes should be carefully analysed and taken into account in the fertilisation management to avoid over-fertilisation. On the other hand, the use of legumes can be appropriate as an additional source of nutrients, replacing mineral fertilisers for soils with poor nutrient content or subject to high nutrient losses.

Develop agroforestry

Developing agroforestry allows increasing the capture of nutrient by root systems and thus reduces nutrient leaching and avoids nutrient loss by erosion. Moreover, nutrients are more efficiently taken up by the trees and crop due to a high C/N ratio. This measure also contributes to the increase in the C/N ratio which enhances N immobilisation. Woody perennials also increase carbon sequestration, which is a benefit to biodiversity by creating habitats. However, some species such as coniferous trees can also lead to soil acidification. Farmers can also use or sell the woody biomass.

The system should be carefully watched in order to avoid resource competition between trees and crops (light, nutrients, water) which would negatively impact the yields. Mixing trees can also reduce the risk of disease that could devastate the crops. Farmers can also potentially profit from the production of trees (fruits and valuable wood). A potential barrier for the implementation of this measure is that agroforestry is a long-term investment. The willingness to participate for farmers may be compromised by the medium term perception of the benefits of this measure. Moreover, the risk of yield decrease may also cause economic issues.

Established paludiculture on rewetted peat lands

The measure of establishing paludiculture on rewetted soils promises a new form of incentive for the farmer as the land can still generate revenues from the land. Paludiculture is the production of species that can be established in wetlands. Depending on the water level that is established, usage for extensive grazing, grassland usage in line with nature conservation requirements, ribbon grass culture, alder tree or reed cultivation is possible (sorted according to low to high water tables) (Wichtmann, et al., 2010). After the harvest, reed and wood can be used as construction material; but this does not represent the main economic option. Indeed, harvested materials are mostly used for thermal combustion in combined heat and power plants, as “biomass to liquid” (reed) or in biogas digesters. Furthermore, subsidies for the compliance with nature conservation programmes are a substantial gain in revenues for the farmer which compensates the losses of the conventional production on the land.

In addition, the measure has co-benefits for nature conservation through the restoration of habitat for rare bird and plant species that occur in the threatened ecosystem.
3.3.3 Controlling the contamination pathways

Use additives that inhibit nutrients transfers

It is estimated that 60% of ammonia emissions are reduced when slurry is acidified during field application (Joint Research Centre, 2013a). The acidification of slurry can also decrease methane emissions. The greatest effect is achieved by acidifying the slurry already in the stable, whereas there will be no effect on CH$_4$ emissions if acidification takes place immediately before slurry application in the field. Sulphuric acid is used to acidify slurry during storage or before application. The draft BREF on intensive rearing of pigs and poultry noted that acidification of the slurry could lead to marginal soil acidification, increasing the need for liming (Joint Research Centre, 2013a). Danish data found that this is a minor issue which corresponds to less than 10 percent of the additional liming needed. Theoretically, 1.4 kg lime should be spread to neutralise each kg of sulphuric acid used for slurry acidification (Joint Research Centre, 2013a).

A range of products is available on the market in order to inhibit the nitrification of ammonium-based fertilisers. This reduces both the risk of nitrogen leaching and the potential for denitrification and consequently the formation of nitrous oxide (Olesen & al, 2013). Nitrate losses are reduced by 35% and nitrous oxide emissions by 30-70% (ADAS, 2011). The inhibitors are more effective if the soil water content is high (University of Illinois Extension, 1993).

The use of urease inhibitors restricts the transformation of urea into ammonia during seven to 14 days. Many inorganic fertilisers already include inhibitors, so this measure is more applicable to organic fertilisers. Urease inhibitors reduce NH$_3$ emissions by 50 to 90% (International Plant Nutrition Institute, 2014). The inhibitors are more effective with low water and SOM contents in the soil, as well as with a basic pH (Laboski, 2006).

These techniques require a medium know-how from farmers on the conditions in which to apply the inhibitors since the effectiveness of the additives depends on manure pH and water content. The inhibitors can be costly but it can be compensated by the fertilisers’ savings.

Increase water retention and avoid land degradation

Adapt tillage techniques

Soil tillage increases the aeration of compacted and waterlogged soils as well as soils with a fine texture. It increases water and nutrient infiltration in soil, decreases the N$_2$O emissions and decreases run-off (Rochette, 2011). In addition to tillage techniques, limiting soil compaction can also be obtained by avoiding the use of heavy machinery on wet soils or by reducing the number of livestock in order to decrease animal trampling. Tillage is also a common strategy to fight weeds in a crop field as it buries weed seeds in the deeper layers of the soils thus limiting the germination.

On the other hand, soil tillage is a disturbance of the soil. It prevents the development of the litter layer and related fertility improvement (especially in semi-arid regions) and reduces SOC sequestration. It is also a source of soil water erosion (up to 25%) and wind erosion, increasing the risk of phosphorus losses. Conservation tillage decreases the decomposition of residues, constituting a “sink” of nutrients bounded to soil elements that can be slowly released for plant uptake. It can improve the soil water retention capacity and thus decrease the risk of leaching. No tillage may only concern large holdings with few or no livestock. Indeed, this measure cannot be implemented in holdings with livestock production since farmers that use manure as a fertiliser need to plough the manure down into the soil.

Another solution could be to restrict tillage in areas at risks through contour tillage in hilly or mountainous areas or no tillage during a defined period (e.g. in autumn). Contour tillage mostly reduces run-off and
soil erosion. It can decrease the total nitrogen losses by 60 % while phosphorus losses can be reduced by 80 % (Gye-Jun Lee, et al., 2010). Restricting tillage during a period of time reduces the risk of nutrient run-off and/or leaching (Key, et al., 2013).

No significant one-off costs would arise for reduced tillage, contour tillage or no tillage during a period of time. No tillage may require the purchase of specialist machinery. Implementation of the measure may also result in some labour cost savings. On the other hand, reduced tillage or no tillage may result in an increased use of herbicides and pesticides or potential yield losses due to weed invasions leading to further costs (Frelilh-Larsen, A. et al., forthcoming).

**Application**

**Consider climatic and geographic conditions**

Consideration of climatic and geographic conditions including avoiding application of organic and chemical fertilisers on land that is waterlogged, flooded, snow-covered or frozen, steeply sloped, before a heavy rain or near watercourses or water abstraction points contributes to reduction of nutrient losses by run-off and/or leaching. Farmers should also pay close attention to the weather forecast on the short term before fertiliser application. Indeed, precipitation after fertiliser application can have negative consequences on nutrient transport by water flows (run-off and leaching) especially with mineral fertilisers which are particularly sensitive to leaching.

In addition to the environmental benefits of this measure, in particular on water pollution, this measure also allows for savings on fertilisers. It is relatively simple to implement although it requires an organisation of the field work. As required by the Nitrates Directive, it also implies the availability of sufficient storage capacities to keep manure during the period when no fertilisation is authorised and manure accumulates. The “closed periods” and their duration are established by each Member States (see 2.2.2.1).

**Choose the best suited application techniques**

Fertilisers must be evenly applied to avoid localised excesses of nutrients that could lead to consequent nutrient losses. Farmers should use machinery that allows a high level of precision of the application and should drive the tractor at a constant speed.

For solid manure and mineral fertilisers, a rear discharge spreader proposes the highest level of precision of the application.

For liquid manure, farmers should use techniques that decrease the surface of fertilisers in contact with air and thus decrease potential for volatilisation and subsequent NH$_3$ emissions. Band spreading decreases the NH$_3$ emissions by 30 % (trailing hose and trailing shoe) on short grass and by 60 % (trailing shoe) on long grass (>10 cm), depending on environmental conditions (variations possible from 0-90 %) (Newell Price, et al., 2011). Deep and shallow injections decrease NH$_3$ emissions by more than 90 % and 70 % respectively (Newell Price, et al., 2011). As shallow injection increases the potential for N$_2$O emissions, only deep injection should be promoted as it lowers also the N$_2$O emissions in comparison to broadcast spreading and other superficial application methods.

As a result, the amount of nitrogen in soil may be higher, which must be taken into account in the elaboration of a fertilisation plan to avoid excessive fertilisation and possible additional losses (Flessa, et al., 2012). The measure further results in a reduced need to use chemical fertilisers and reduced odour emissions while even application ensures better yields as all crops are fertilised (ADAS, 2011). For all these possible techniques, specific equipment is necessary and needs to be rented or purchased. Changing the application technique may induce significant modifications in the farm management,
possibly requiring a machinery change with high investment costs which may discourage farmers from changing their existing practices.

**Incorporation**

To avoid nutrient run-off and ammonia emissions during manure application, manure landfill through injection or ploughing after application, seems to be an efficient option. Indeed, when applied directly in the soil, manure has less contact with air thus reducing ammonia volatilisation. Moreover, manure is trapped in the soil structure and less likely to be transported out of the field by water flows in case of rain episodes. Injection of manure in the soil reduces 90% of ammonia emissions when incorporated in deep layers of the soil and 70% when incorporated in shallow layers of the soil (ADAS, 2011). This measure also reduces considerably the odours during manure application and offers a more regular and controlled quantity of manure to be applied.

However, the incorporation of manure can damage the soil structure due to the use of plough, discs or tines and affect crop yields. If that is the case, the fertiliser applied is not fully utilised by crops, which increases the risks of nitrate leaching over the next winter drainage period. This technique is best applied to tilled arable land and reseeded grassland.

Incorporation can be costly. It has been estimated that the additional plough-based cultivation costs can be €60 per hectare (ADAS, 2011). The incorporation can be performed as part of normal field preparation, however, rescheduling might be required to synchronise spreading and rapid incorporation.

**Catch nutrients by managing landscape and land use**

**Use cover crops and reduce bare fallow**

Cover crops are used to reduce nutrient run-off and leaching, in particular for nitrogen, during the fallow period. Cover crops are more effective if cover crops are catch crops. The cover crops roots also limit the risk of soil erosion and thus additional losses of phosphorus. Moreover, cover crops also benefit to biodiversity, organic matter and are a way to fight against weeds. This measure requires time for seeding and destruction of the cover and additional costs for machinery operations. However, these cover crops allow saving nutrients from soil that can be used by the next crop if they are incorporated.

**Create buffer zones**

Buffer zones are used as a management tool to prevent nutrient access to waterways, filter sediments and other particulates (including nutrients), and provide habitats. A buffer zone can be an area around a cropped field or along a shoreline or a water stream (buffer strips) where the use for agricultural production is restricted or prohibited and the zone is left as a grassy sward or planted with vegetation or woodlands (‘edge-of-field’ buffer strips). ‘In-field’ buffer strips may also be positioned within cropped fields. A buffer zone can also be a constructed wetland or a sedimentation pond. The constructed wetlands (CWs) are large free water surface wetlands are designed and constructed primarily for removal of nutrients. The nutrients are retained by the soil particles, absorbed by the plants or emitted into the air in the case of nitrogen. The sedimentation pond is composed of a small, shallow basin and a wetland filter covered with typical wetland plants. It is designed primarily to retain phosphorus (Baltic Compass, 2012).

In a short period of time, concentration of nitrate (N) can be reduced from between 40 and 94% in a buffer or wetland before entering a stream. With buffer strips and constructed wetlands, phosphorus losses are reduced as well but to a lesser extent. In some cases, the buffer strip soil can even be saturated in P due to its ability to be bounded to soil particles (SAI Platform, 2010). Hence it cannot
catch phosphorus ions anymore; they are thus lost to run-off or leaching. By contrast, sedimentation ponds are particularly suited to capture phosphorus bounded with soil particles.

Once established, the buffer zones require little maintenance. They could even be utilised as extensive grasslands or in some cases as short rotations for energy wood biomass energy to use or sell for bioenergy generation, leading to lower electricity costs or revenues from the sale of excess electricity. However, depending on the land price for the area taken out of production and the resulting income loss from not harvesting crops there, costs may vary. In the case of constructed wetlands and sedimentation ponds, the accumulated sediments need to be regularly removed. They can be applied to field as fertilisers.

**Protect and restore organic lands**

Wetlands are often called green filters in reference to their capacity to reduce pollution in water (Alvarez-Rogel, et al., 2005). Indeed, wetlands have the capacity to filter wastewater and nutrients from arable land. Wetlands which have been drained in order to allow for agricultural production are major contributors to global GHG emissions through N₂O as well as CO₂, particularly if N fertiliser is added (Frelih-Larsen, A. et al., forthcoming).

The level of greenhouse gas emissions emitted by moors can be mitigated through the elevation of the water table. Established together with nature conservation attempts, this measure includes the dismantling of drainage mechanisms and allowing the water table to rise close to the soil surface. Hence the area would not be usable for agriculture anymore. The effectiveness for the reduction of greenhouse gas emissions and nutrient leaching and run-off is proportional to the increase of the water table. When a former moor is used today as an extensive grassland with high water table, greenhouse gas emissions decrease from 33.8 t CO₂ eq./ha/yr to 10.3 t CO₂ eq./ha/yr (Flessa, et al., 2012). The building of artificial ponds can also reduce nitrate losses.

When restoring the water table to a natural level, the use of the rewetted ecosystem is very limited. Therefore, acceptance by farmers is very low when no compensation is offered. This practice is proposed with the additional implementation of establishing paludiculture on rewetted sites to compensate the farmers for their lost revenue.

**Promote the conversion or maintenance of grassland, in particular in areas of high soil risk**

Land use change from arable production to grassland converts the land to a less intensive use and reduces the potential of nutrient losses. This measure could be very effective in areas with high loss risk. This is also the case for the maintenance of grassland. To reduce the leaching and erosion risk, the grassland sward needs to be seeded, or re-seeded if needed, at an appropriate density rate (20 kg per ha generally recommended (RSBP, 2008) and rolled afterward in order to ensure productivity, prevent weed establishment and encourage consolidation of seeds (Hybu Cig Cymru, 2008). The costs for the farmer are high due to the loss of production. Therefore, acceptance without compensation is unlikely.

**Choose adequate irrigation and drainage systems**

**Increase water efficiency**

Excessive water content in the soil can simply flush the nutrients to water bodies or saturate the soils’ water retention capacity. Soil saturation results in high emissions of N₂O (and methane) due to the anaerobic conditions and high risks for run-off and/or leaching. This is particularly the case for flood irrigation or for submersion irrigation. Conversely, measures that help to increase the water efficiency
contribute to the reduction of the pressure of agriculture on water and limiting the over-abstraction of water.

Multiphase irrigation and irrigating at appropriate times allow the water supply to be adapted to the plant needs. It therefore reduces the water losses and related nutrient losses through leaching and run-offs. This measure requires frequent measurement of soil moisture and that the local soil and climatic conditions be taken into account.

Drip irrigation prevents water excess by adapting the irrigation to the plant’s need and uptake capacity. Water use efficiency is 70 to 95% higher than in open systems. It thereby reduces run-off or leaching of potential nutrient surpluses. Thus it also allows saving on fertilisers by mixing an appropriate amount with the irrigation water (“fertigation”). Drip irrigation also minimises the risk of fungal contamination, reduces soil crusting and compaction, and allows for a better control of weed growth.

Encouraging farmers to use an irrigation device is a necessary and effective system to control the water provided and to limit the losses of both water and nutrients. The implementation of this type of system may face technical and acceptance barriers from farmers. Indeed, the use of drip irrigation is not common for cereals, although it is possible, including for maize. This is not the case for rice. The measure requires a high accuracy given the small amounts of water applied, which may lead to under-irrigation and reductions in crop yield and quality, or over-irrigation and possible yield losses. In particular, drip irrigation generates a restricted root system that requires frequent water (and nutrient) supply which implies a greater attention from farmers, and related working time, as compared to other irrigation systems. Some herbicides and top dressed fertilisers require sprinkling and thus cannot be used through the drip irrigation system. Moreover, the success of this measure also depends on how the farmers are willing to change their crop management. Indeed, the implementation of this measure requires the placement of drip lines that may limit tillage operation.

In the case of the fertigation practice, the farmers must couple this practice with a fertilisation plan to avoid over-fertilisation as well as under-fertilisation. It means that a detailed knowledge of crop need and nutrient concentration in soil solutions is necessary. This practice can easily be combined with the use of organic fertiliser that had been previously processed to separate the liquid from the solid phase which is already sometimes done in the region (e.g. in some cases up to 40% of pig manure).

**Improve the drainage systems**

Making sure that the drainage systems and the materials used are in good shape allows for N\textsubscript{2}O emissions from water logged soils to be reduced and to avoid nutrient run-offs. However, it could also increase nutrient leaching. Hence the drainage systems should be adapted to the local geographical, soil and climate conditions. It can be minimised in areas with a high risk of leaching and enhanced in areas with low risk of leaching and subject to waterlogging (e.g. in soils with fine texture). The acceptance among farmers is expected to be high if water saturation affects the crop yields. Minimising drainage can induce additional costs if there is a need to adapt the existing drainage systems. The costs can be minimised if farmers only allow existing (old) drainage systems to naturally deteriorate but it can possibly induce loss of production (ADAS, 2011).

The reuse of agricultural drainage water for irrigation by pumping lost irrigation water from drainage channels and re-injecting this water into the irrigation cycle can also be a solution to reuse the lost nutrients. However, this is still experimental and could require significant investments.
3.4 Focus on manure transfer

3.4.1 Principles of manure transfer

Rationale for transferring manure

Ruminants (grazing animals) are mainly fed with roughage that is generally produced on the farm where the animals are raised (Luesink, et al., 2014). The manure produced on these farms can be applied on the cropland of the farm. In a balanced system, there is enough cropland and pasture surface to apply the produced manure. However, there are legal thresholds for the maximum nitrogen amount and, in some countries phosphorus amount, from livestock manure that can be applied (see section 3.4.2). Intensive livestock farms, especially in pig and poultry farms, often generate more manure than can be applied on croplands and pastures according to the legal and agronomic manure application thresholds. Some intensive livestock farms with pigs and poultry do not even have any land attached to agricultural buildings. Farms with a manure surplus have to transport this to other farms where there is a need for manure to be applied.

Main constraints related to manure transfer

Transferring manure induces significant costs for farms that produced the manure. Costs are related to the volumes transported and transportation distances. Generally, manure from grazing animals – i.e. liquid manure with low mineral content – is applied nearby, while manure from poultry – i.e. solid manure with high mineral content – is transported over longer distances (see section 3.4.4).

Manure processing such as the separation of cattle and pig manure into its liquid and solid fractions is an option to reduce the overall costs of manure transport. The transportation costs per unit of minerals for processed manure products is cheaper than for raw manure due to the higher concentrations of minerals in processed manure. In addition, when the solid fraction is transported to other MS, sanitary measures are required (hygienisation of manure and cleaning of the trucks) by the European Waste Shipment Regulation\(^{46}\) (EWSR) to prevent spreading animal diseases (Luesink, et al., 2013).

Transport of manure over long distances consists of several steps, as presented in Figure 18. Storage facilities for manure are required both in the farms producing the manure and in the farm receiving the manure. Indeed, manure production takes place all year round whereas crops only need minerals in their growing season. When manure is stored, then it can be used in appropriate times of the crop growing season. The Nitrates Directive has rules on the capacity of manure storage in Nitrate Vulnerable Zones, to prevent the application of manure in periods when crops are absent or do not need the nutrients.

The transportation of manure involves additional (non-renewable) fuel use and, as a consequence, additional air pollution and greenhouse gas emissions.

**Situation in the EU**

As shown in Figure 19, EU regions with extremely high manure production are the south-eastern part of the Netherlands and west Flanders (Belgium). Other regions with relatively high manure production are the northern part of Italy and Spain, as well as Brittany (France), West Germany, Ireland and Denmark. In all regions (except Ireland), manure production exceeds the level of legal application (Grizetti, et al., 2007) and thus those regions have to manage their manure surplus.

Source: Calculations LEI Wageningen UR based on FADN 2010
Table 10 shows the production of manure in ton per 100 ha of agricultural land (utilised agricultural area). The different types of animals have been aggregated into three groups: grazing animals, pigs and poultry. In all of the eight selected NUTS 2 regions (see chapter 4 for details on how these regions were selected), the largest share of manure comes from grazing animals. The production of pig manure ranges from 2 % (South and eastern Ireland) to 36 % (North-Brabant). Even though manure production is relatively high in all of these regions, there are significant differences. For instance, the total manure production in North-Brabant is six times higher than in Murcia. The manure production is calculated with the number of animals from FADN data of the year 2010 and the excretion factors from the Dutch manure laws (Netherlands Enterprise Agency, 2014). In the present study, it was not possible to gather the excretion factors for each Member State. The variation margin of the excretion factor is set at ±10 %, because of differences in milk yields of dairy cows in some Member States, lower feed intensity in pig production, differences in production systems and differences in shares between beef and dairy cattle, amongst others (Lyngsø Foged, et al., 2011).

Table 10 – Production of manure in ton per 100 ha of agricultural land (UAA) per selected NUTS 2 region based on FADN data (NUTS 3 level) for the year 2010

<table>
<thead>
<tr>
<th>Region</th>
<th>Grazing animals</th>
<th>Pigs</th>
<th>Poultry</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brittany, France</td>
<td>1 937</td>
<td>508</td>
<td>93</td>
</tr>
<tr>
<td>Lombardy, Italy</td>
<td>2 342</td>
<td>555</td>
<td>43</td>
</tr>
<tr>
<td>Mit Jutland, Denmark</td>
<td>942</td>
<td>759</td>
<td>8</td>
</tr>
<tr>
<td>Murcia, Spain</td>
<td>502</td>
<td>528</td>
<td>17</td>
</tr>
<tr>
<td>North-Brabant, the Netherlands</td>
<td>3 743</td>
<td>2 174</td>
<td>167</td>
</tr>
<tr>
<td>South and eastern Ireland, Ireland</td>
<td>2 211</td>
<td>36</td>
<td>2</td>
</tr>
<tr>
<td>Weser-Ems, Germany</td>
<td>2 044</td>
<td>661</td>
<td>71</td>
</tr>
<tr>
<td>Wielkopolska, Poland</td>
<td>727</td>
<td>259</td>
<td>29</td>
</tr>
</tbody>
</table>

1 Including liquid fraction after separation manure from grazing animals and pigs
2 Including solid fraction after separation manure from grazing animals and pigs

Source: Calculations by (Karaczun, 2010).

For North-Brabant, the difference between the manure production figures of Table 10 and the official sources of the Dutch manure production (Luesink, et al., 2014) are less than 1 %.

47 Farm Accountancy Data Network
48 Other studies about manure production in the EU encountered the same issue. For instance Lyngsø Foged, et al. (2011) wrote: “The decision to make our own, simplified estimates of the livestock manure production followed an unsuccessful search for published, uniform and updated statistics or estimates covering all MS.”
3.4.2 Regulatory framework

In order to minimise and/or mitigate the negative effects of nutrient surpluses on the environment, European regulations regarding the use and handling of animal manure have been implemented, among which:

- The EU Nitrates Directive (Council Directive 91/676/EEC). The Nitrates Directive aims at reducing water pollution caused or induced by nitrates from agricultural sources and preventing further such pollution, and
- The EU Water Framework Directive (2000/60/EC). The Water Framework Directive aims at preserving the aquatic ecosystems and the unique and valuable habitats, as well as drinking water resources and bathing waters.
- EU regulations 1069/2009/EC and 1774/2002/EC. These regulations focus on animal by-products, including animal manure.

EU-regulation 2003/2003/EC relating to fertilisers (in a revision process at the time of publication of this report) addresses chemical and organic fertilisers from non-animal origin and does not include requirements about animal manure.

**EU Nitrates Directive**

Pursuant to the Nitrates Directive, all EU Member States have drawn up action programmes for areas which are subject to nitrate leaching or run-off (areas known as Nitrate-Vulnerable Zones – NVZ). Pursuant to article 3.5 of the Directive, Member States can also opt to establish and apply action programmes throughout their whole national territory. In 2014, action programmes had been implemented in almost 50 % of the European territory in order to limit the potential impacts caused by nitrates. Eleven Member States decided to protect their whole territory (Austria, Denmark, Finland, Germany, Ireland, Lithuania, Luxembourg, Malta, Netherlands, Romania, Slovenia), as well as Flanders and Northern Ireland. The other Member States designated parts of their territory as NVZ. In all those areas, the application of nitrogen from livestock manure is limited to 170 kg nitrogen per ha per year (Council Directive 91/676/EEC). The action programmes are reviewed and if necessary revised every four years and include at least the measures in Annex III of the Directive, including the 170 kg N/ha/year obligation. In certain cases, Member States can benefit from a derogation allowing application of a higher limit. This derogation must be justified on the basis of objective criteria, for example long growing seasons, crops with high nitrogen uptake, high net precipitation in the vulnerable zone and soils with exceptionally high denitrification capacity. The allowed amount must be fixed so as not to prejudice the achievement of the objectives of the Directive. At the end of 2014, six countries had a derogation, namely Belgium (Flanders), Ireland, Denmark, the Netherlands, the United Kingdom and Italy (Lombardy, Piedmont, Veneto, and Emilia Romagna). Farms with derogation may use from 230 to 250 kg of nitrogen from animal manure per ha.

Due to the limitations of the national laws based on the nitrate action programmes, there is more animal manure produced than can be applied in certain regions. There is a need to transport this manure surpluses to regions with manure and/or nutrient deficits.

**EU Water Framework Directive**

Currently, the share of water bodies with good quality status in the MS range from less than 10 % to about 50 % (European Commission, 2012a), one major issue being the high concentrations of nutrient in the surface and ground waters due to the use of fertilisation in agriculture. In application of the Water Framework Directive, River Basin Management Plans (RBMPs) had to be drawn up for all River Basin District (RBD) in order to ensure that all surface waters reach a good ecological status by 2015 (or at
the latest in 2027 in case of exemptions). For diffuse impacts of pollution (including manure) the article 10 of the Directive indicates that “Member States shall ensure the establishment and/or implementation of [...] best environmental practices”. The Directive had to be integrated in national laws, including practices such as balanced fertilisation. Many countries such as Sweden, Denmark, Germany, Belgium and the Netherlands, already have manure-related laws aiming at implementing balanced fertilisation.

### 3.4.3 Market for manure and economic opportunities for farmers through manure processing

#### Market for manure

The market for manure consists of the farmers (or regions) with manure surpluses as suppliers and the farmers (or regions) with manure deficits as demanders. The demand for manure is essentially derived from the demand for nutrients. However, nutrients can be obtained from animal manure and alternatives such as (chemical) fertilisers and compost. In this respect, manure or manure products, fertiliser and compost are substitutes for each other to a large extent (Luesink, et al., 2013). As a result, given their needs for nutrients (depending on the type of crop and the area), arable farmers can be expected to choose the cheapest alternative.

Compared to chemical fertilisers, manure has the advantage of containing organic matter and micro-minerals. It also has disadvantages: manure is voluminous, its nitrogen content is uncertain as it depends on the composition of the manure and thus, it is more difficult to dose with respect to nutrient demands. In addition, the costs for spreading animal manure are higher than for chemical fertiliser application, due to the greater volume. On the whole, the costs of nutrient application coming from animal manure or fertiliser are economically balanced (Uenk, 2012). The acceptance and price of phosphate and potassium in manure does not differ from chemical fertilisers. Thus, the price of manure that an arable farmer would be willing to pay will be about the same price as phosphorus fertiliser. Since nitrogen availability for plants is lower in manure than in chemical fertilisers, arable farmers in the EU will not be inclined to pay more than 60% of the price of chemical nitrogen fertiliser for manure (Uenk, 2012).

Profitability of their activities is a key aspect for farmers. While farmers with animal manure surpluses would like to sell their manure surplus at the highest possible prices, farmers who buy manure want to buy it at the lowest possible prices. If there is a situation where the manure demand is lower than the manure supply, then the price of manure for recipients could even become negative, as is the case at this moment in the Netherlands and Flanders.

#### Box 2 – Focus on Dutch manure market

Pig manure from fatteners in the Netherlands contains 9.0 kg of nitrogen, 3.9 kg of phosphate and 6.8 kg of potassium per ton of manure. In 2013, chemical fertiliser prices were € 1.02, € 0.93 and € 0.61 per kg of nitrogen, phosphate and potassium, respectively (LEI, 2014). Thus, an arable farmer would be willing to pay a maximum of € 13 per tonne of pig manure from fatteners. The transportation costs are about € 2 per tonne and € 0.8 per tonne per km (Kruiseman, et al., 2008; Kruiseman, et al., 2012). The costs of GPS systems in transportation trucks, of weighing, and of manure analyses of mineral contents are included and are about € 1 per tonne (Horne, et al., 2013). Government costs for a control system and the administrative costs for the farmer are not included. Based on these costs, in this case study pig manure from fatteners can be transported over 135 km before a pig farmer has to pay for the removal of his surplus manure (Luesink, et al., 2013). The transportation cost will then cancel out the selling price of manure. However, similarly to other markets, the price will drop when supply is higher than demand. This is the case in all European regions with manure
surplus. The Netherlands experiences the worse situation, since the manure surplus concerns the whole country. Considering phosphate, for instance, the supply is about 85 million kg while the demand is between 50 and 55 million kg (Luesink, et al., 2014).

Due to the higher supply than demand, manure tends to be seen as a waste product rather than a valuable fertilising product. As a consequence, Dutch arable farmers in the northern and southwestern parts of the country get paid when they accept pig manure. Pig farmers from the southeastern Netherlands had to pay approximately €18 per tonne to get rid of their manure surplus. Although there are price incentives for farmers that can apply manure, a part of the manure produced within the Netherlands has to be eventually exported (Luesink, et al., 2014). The Netherlands and Flanders (Belgium) are the only EU regions that produces so much manure that it is not legally possible to apply the surplus anywhere in the country.

**Manure processing**

In that context, manure suppliers try to minimise transportation costs of manure by processing it. Lyngsø Foged, et al. (2011) carried out an inventory of the processed manure for all EU Member States. The authors distinguished separation of manure (solid and liquid manure), pre-treatment, anaerobic treatment, and treatment of the solid/fibre fraction.

With the separation of the solid and liquid fractions of manure, the minerals in solid manure are more concentrated. The volume of the obtained manure products is much lower than that of raw manure and therefore the transport cost per kg of minerals for manure products are lower than for raw manure.

With anaerobic treatment of animal manure, farmers can generate energy for electricity production. The higher the price of electricity, the higher the number of installations is in a country. Note that in anaerobic treatment, animal manure is combined with other biomass inputs such as maize silage. Along with the generation of biogas, digestate is produced as a result of the anaerobic treatment process, which results in 50 to 60% higher outputs of nitrogen and phosphate than the input by manure. More than 100 coproducts are used. The three most important are: maize silage, glycerine and vegetable oils (Peene, et al., 2011). The amount of nutrient in the digestate is higher than the amount of nutrient in the animal manure used as an input. For instance, the digestate will have a higher concentration of phosphates which are not converted into the biogas. In the Netherlands, the digestate is classified as manure if more than half of the inputs consist of animal manure, which would imply that the same legal application norms for manure must be applied for digestate. Other MS have different approaches.

The Figure 20 presents the costs related to manure processing per NUTS3 region in the EU. Only data on manure separation and anaerobic treatment are taken into account. According to Schroder, et al., (2010), the costs of manure separation range from €0.80 to €1.00 per m³ of manure. In the modelling presented in Figure 20, an estimated cost of €0.90 per m³ was used. In addition, similarly to Lyngsø Foged, et al. (2011), the amount of coproducts and their nitrogen and phosphate content are considered as manure products. The revenues of biogas production are ignored because the revenues for electricity production are approximately equal to the production cost of anaerobic treatment (Peene, et al., 2011).
3.4.4 Transportation of manure

Transportation distances per type of manure

Most European countries do not have statistical information about manure transport and distances (EAGER, 2008). European studies about nutrient balances are all based on models, for instance CAPRI and Miterra Europe, (Bouraoui, et al., 2011; Velthof, et al., 2007; Grizetti, et al., 2007). Due to lack of data, manure transport is not included in the models used for those studies (Oudendag, 2014). In all those studies, the manure production in an area is divided based on land use and crop uptake (Bouraoui, et al., 2011; Grizetti, et al., 2007).

As discussed previously, in some regions of Europe (e.g. in the Netherlands), the supply of manure is higher than what can be legally applied. Therefore, the manure excess has to be exported with or without processing. The last few years about 14 million kg of phosphate from solid poultry manure has been exported annually to Germany without processing (Luesink, et al., 2014). About 8 million kg of phosphate in cattle, pig and poultry manure is exported annually to Germany and France after hygienisation or composting. About 400 000 tonnes of solid poultry manure is burned annually and the ash, together with about 8 million kg of phosphate, is used as fertiliser in France and Wallonia (Horne, et al., 2013). At least about 100 000 tonnes of solid poultry manure is dried, pelletized, and exported annually all over the world.

For the Netherlands, manure transportation distances depend on the manure product (Luesink, et al., 2013; Luesink, et al., 2014) as follows:

- Slurry of cattle manure: less than 50 km;
- Slurry of pig manure: approximately 50-150 km;
- Solid cattle manure and solid products of separated pig and cattle manure: 100-250 km;
• Solid poultry manure: 150–500 km;
• Ash of burned chicken manure: more than 250 km;
• Pellets of processed manure: more than 500 km.

Some information about manure transport was found for the Flanders region (Van der Straeten, et al., 2011). In Flanders, the distances over which manure is transported are approximately two third of the distances in the Netherlands for slurry and solid manure (Van der Straeten, et al., 2011).

**Transportation costs**

Although manure transfer has been in place for several years in certain parts of EU such as Netherlands, Flanders (Belgium) and some parts of Germany, none of the EU Member States have statistics based on field data on the transportation (costs) of manure for direct use on the crop fields. All estimations of manure transportation costs are estimated by models.

The Table 11 presents for each NUTS 3 region in the EU, transportation costs that have been calculated with LEI model. For the calculation of manure transportation costs (processing and transfer), it was assumed that the transportation costs per m$^3$ per km are uniform across the EU, because the market for transportation is an international market with a high degree of competition (Kruseman, et al., 2008; Kruseman, et al., 2012), although it should be acknowledged that legislation for transportation of manure differs across Member States. For manure transportation within a region, the costs are € 4.50 per tonne (including transportation up to 100 km). For transportation outside a region or abroad, the costs are € 2.00 per tonne and € 0.08 per tonne per km (Horne, et al., 2013).

For all surplus NUTS 3-regions which have to transport manure out of their territory, the closest driving distances to other regions (i.e. regions with deficit that can accept manure) was calculated. The driving distance between two NUTS 3-regions is based on the distance on the map (between the 2 centres) times a factor 1.3 (Kruseman, et al., 2012). All driving distances were sorted; the highest priority was given to the nearest region (shortest distance).

For each region the costs were calculated from the cost per ton, the volume in tonnes and the transport distance (see results in Table 11).
Table 11 – Transport costs in euro per 100 ha of agricultural land (utilised agricultural area) per selected NUTS 2 regions based on FADN data (NUTS 3 level) for the year 2010

<table>
<thead>
<tr>
<th>Region</th>
<th>Grazing animals&lt;sup&gt;1&lt;/sup&gt;</th>
<th>Pigs</th>
<th>Poultry&lt;sup&gt;2&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brittany, France</td>
<td>872</td>
<td>1,813</td>
<td>1,295</td>
</tr>
<tr>
<td>Lombardy, Italy</td>
<td>3,699</td>
<td>4,953</td>
<td>788</td>
</tr>
<tr>
<td>Mit Jutland, Denmark</td>
<td>451</td>
<td>2,300</td>
<td>57</td>
</tr>
<tr>
<td>Murcia, Spain</td>
<td>227</td>
<td>1,551</td>
<td>61</td>
</tr>
<tr>
<td>North-Brabant, the Netherlands</td>
<td>19,687</td>
<td>60,611</td>
<td>5,307</td>
</tr>
<tr>
<td>South and eastern Ireland, Ireland</td>
<td>995</td>
<td>106</td>
<td>6</td>
</tr>
<tr>
<td>Weser-Ems, Germany</td>
<td>988</td>
<td>3,875</td>
<td>1,250</td>
</tr>
<tr>
<td>Wielkopolska, Poland</td>
<td>327</td>
<td>757</td>
<td>106</td>
</tr>
</tbody>
</table>

1) Including liquid fraction after separation manure from grazing animals and pigs
2) Including solid fraction after separation manure from grazing animals and pigs

Source: Calculations by LEI Wageningen UR based on FADN 2010.

Figure 21 – Transport costs in euro per 100 ha of agricultural land (utilised agricultural area) per NUTS 3 region in the European Union

Source: Calculations by LEI Wageningen UR based on FADN 2010.
3.4.5 Organic matter balance

The use of livestock manure is one of the main methods (along with crop rotation and green manure) used throughout history to maintain soil fertility. Manure spreading not only brings nitrogen, phosphorus, potassium, and micro-nutrients that are essential for crops in a form that can be directly assimilated by the plant, but also improves soil quality (European Commission, 2014a). In particular, it allows the organic matter content of soils with lower organic matter content and low fertility to be increased (Schoenau, et al., 2004).

The organic matter balance of the soils is important as it improves the structure and the water holding capacity of soil. These aspects are essential for optimal crop growth and for optimal nutrient efficiency.

One of the advantages of manure transfer to regions with low soil organic matter content is the improvement of the organic matter balance. Regions with a low level of manure application are susceptible to unbalanced organic matter. Therefore, regions with a low manure application will have a large deficit of phosphates and will have soils with unbalanced organic matter. Regions with low level of manure application (with phosphates deficit and unbalanced organic matter) can be found in eastern Germany and in some regions of Spain (see Figure 22).

![Figure 22 – Space for manure application (kg P2O5/ NUTS 3 region) in the European Union](source: Calculations by LEI Wageningen UR based on FADN 2010)

3.4.6 Animal health aspects

Manure, especially slurry from cattle and pigs, has a potential for spreading infectious diseases (Baguer, et al., 2000; Betner & Belsham, 2012). Some outbreaks have been traced to animal manure (e.g. E. coli or protozoa) (Bicudo & Goyal, 2003). An assessment of hygienic and ecological risks of the utilisation of animal manure is complex (Venglovsky, et al., 2009). A wide variety of pathogens can be found in the faeces of humans and wild and domestic animals.

Risks from manure application (i.e. after transport) relate to the following factors:
Storage – Conditions during storage affect the survival of pathogens in stored manure, but this is difficult to predict because many factors affect the survival, e.g. type of slurry, pH, temperature, concentration of pathogens, ability to form spores, ability to survive long term (oocytes) (Pell, 1997).

A disease can be present in the herd before it is diagnosed. In the case of foot-and-mouth disease the incubation period is 2-3 weeks (Botner & Belsham, 2012). Therefore, the storage and transport manure or transport of manure of affected animals within this incubation period risks spreading the disease.

Manure processing – (Lyngsø Foged, et al., 2011) reports the results of an extensive study on manure processing in Europe. There is a wide variety of animal manures. According to (Venglovsky, et al., 2009), solid livestock manure is relatively safe when it is composted (aerobic digestion). Untreated or improperly treated manure is, however, a source of zoonotic pathogens that contaminate soil and water for many months (Venglovsky, et al., 2009). Traditional dung pits generate temperatures sufficiently high to destroy pathogens (Baguer, et al., 2000) but the absence of bedding material nowadays lack self-heating processes. Inactivation or disinfection can be reached with long-term storage, the addition of granulated quicklime, chemicals, and/or physical or biological means (Baguer, et al., 2000; Bicudo & Goyal, 2003; Haas, et al., 1995). Note however that the variety of conditions yields different results. Some measures to process manure have unfavourable effects like toxicity of chemicals or high energy costs of electrolytic treatment (Bicudo & Goyal, 2003). If manure is treated in installations that collect manure from different farms, then it should be kept in mind that this is a route for cross contamination, of manure but especially vehicles (Botner & Belsham, 2012).

Multiplication – With the right conditions low concentrations of pathogens can be increased by multiplication (e.g. bacteria) and this may cause high risk (Venglovsky, et al., 2009). Climate change can reinforce this effect. Viruses are unable to replicate outside their hosts and therefore their numbers never increase (Pell, 1997).

Survival on land – According to (Venglovsky, et al., 2009), parasites, spore-forming bacteria, and some types of viruses can persist for a long period. Survival increases when manure is incorporated in the soil (compared to surface application) due to reduced UV radiation, temperature, and drying. Survival also depends on the manure type, time of the year, presence of plants, etc. (Bicudo & Goyal, 2003). Therefore the evaluation of the relative risk posed by different pathogens is difficult.

Water contamination – There is not only a significant microbiological risk for crops or grazing by livestock, but also the likelihood of pathogens reaching aquifers or surface waters when used for irrigation or consumption (Venglovsky, et al., 2009). Drinking water with protozoa, which cysts or oocysts which remained viable for at least a year (Current, 1988), has led to disease outbreaks (Pell, 1997). Groundwater can be contaminated with viruses by means of wastewater irrigation (Pell, 1997). Water can lead to the movement of pathogens (Venglovsky, et al., 2009), though the distance is influenced by many factors, e.g. soil moisture and adsorptive capacity. Periods of rainfall can cause run-off from fields ending in streams which may subsequently be used for water supply (Bicudo & Goyal, 2003).

Antimicrobial resistance – Antimicrobial agents risk producing antimicrobial-resistant bacteria. Most of the applied antibiotics end up in manure. Antibiotics that bind strongly to the soil have long half-lives and there is concern that they may be taken up by plants (Venglovsky, et al., 2009). (Baguer, et al., 2000) claim that manure appears to be the main pathway for the release of antibiotics in the terrestrial environment. However, the effect of this on human health is difficult to determine. Much work is needed to fully address this aspect, also with respect to the transmission of (non-)zoonotic pathogens (Venglovsky, et al., 2009).

Risk for human health – The potential of transmission to animals and humans via land application of manure is difficult to assess due to a large number of factors and conditions. An increased survival of
zoonotic pathogens is a concern (Venglovsky, et al., 2009). Environmental contamination can exist for many months. Several studies indicate that land-applied manure from Q-fever positive dairy goat farms may be an important additional source of human Q-fever (Herman, et al., 2014). In their study, Hermans et al. concluded a “high percentage of notified human cases geographically associated with contaminated land parcels”. In the Netherlands, which had an outbreak from 2007 to 2010, manure is transported to arable farms in other regions and thus played a significant role in the transmission of Q-fever, especially applied during or shortly after the lambing season. A hygiene protocol became mandatory (coverage during storage and transport and immediate ploughing).

**Hygienic measures in EU** – Transport of manure between Member States of the EU must follow strict procedures and regulations such as Regulation (1069/2009/EC). According to this Regulation, if livestock manure or processed manure are exported to other Member States, they must have been treated at least 70°C for 60 minutes in an approved biogas, composting, or technical plant. EU Member States have specific additional regulations. In some states in Germany, poultry manure has to be hygienised before transport. In France, imported manure products are only accepted if they have been composted.

**Management** – The farmer should be aware of the risk of pathogens in manure. Decisions for storage (time and conditions), processing, transport and application on land should be taken with this risk in mind. Recommendations should be communicated and followed, e.g. not to apply next to a watercourse or open tile inlet, or when the soil is wet or water-logged, or when heavy rainfall is forecasted (Current, 1988). A number of recommendations are also given by (Haas, et al., 1995).

**Manure transport** – Although the probability of an outbreak caused by manure transport might be considerably less than by animal transport, by wild animals, or by wind, there still is a probability that an outbreak might occur in near future with high impact. However, the risk of spreading infected slurry cannot be clearly quantified.
4. Identification of nutrient saturated and nutrient scarce areas

The previous chapters provided an overview of the major potential effects of agriculture on nutrient cycles and the possible solutions to prevent, limit or reduce the excess of nutrients and related impacts. However, as mentioned above, nutrient fate is closely related to the local farming system and environmental conditions. Consequently, the good practices to reduce excess nutrients at farm level can differ from one European region to another. The present chapter aims at identifying key saturated areas by analysing nutrient budgets at the EU level. This will allow for relevant regions to be identified for which notable farming systems and practices leading to nutrient excess as well as costs generated by such excess will be studied in more detail. Such analysis will allow the proposal of specific good practices that farmers could implement at farm level to reduce local excesses of nutrients and related impacts.

4.1 Nutrient budget in agriculture

4.1.1 Defining nutrient budget

Nutrient budget, formerly called Gross Nutrient Balance, is the quantification of all major nutrient flows within a given geographical area and period. In other words, nutrient budget is calculated as the difference between all nutrient inputs and all nutrient outputs within a determined system. It can be calculated at a European or national level, as well as the watershed or farm levels.

Three approaches can be used to estimate nutrient budgets (Eurostat, 2013b), varying according to their boundaries:

- Farm budget: refers to all nutrients that enter and leave the “farm gate”;
- Soil budget: refers to all nutrient inputs to soil and nutrient outputs from the soil;
- Land budget: refers to all nutrient inputs and outputs in a determined area, taking into account flows from/to air, soil and water;
- In the case of phosphorus, land and soil budgets are comparable since phosphorus is not emitted to air. For nitrogen, the land budget considers emissions, leaching and run-off and N stock changes in soil (similar to soil budget) but also in the farm during housing and the storage of manure.

In 2007, the OECD and Eurostat published two handbooks defining guidelines for the calculation of gross nitrogen and phosphorus balances. A few years later, the term nitrogen “budget” was introduced by Leip A. et al (2011) and used in the Expert Panel on Nitrogen Budgets (EPNB) to elaborate guidance for emissions accounting policies (Economic Commission for Europe, Executive Body for the Convention on Long-range Transboundary Air Pollution, 2012). In 2013, an updated handbook for the calculation of nutrient budgets including both nitrogen and phosphorus was published by Eurostat and the OECD (Eurostat, 2013b).
In this handbook, nitrogen and phosphorus budgets are defined as follows:

**Nitrogen budget**

\[
\text{Nitrogen budget} = \text{nitrogen inputs} - \text{nitrogen outputs} \\
= (\text{mineral fertilisers} + \text{manure production} + \text{net manure import/export}, \text{withdrawals} + \text{other organic fertilisers} + \text{biological N fixation} + \text{atmospheric N deposition} + \text{seed and planting materials}) \\
- (\text{crop production} + \text{fodder production} + \text{residues removed/burnt})
\]

**Phosphorus budget**

\[
\text{Phosphorus budget} = \text{phosphorus inputs} - \text{phosphorus outputs} \\
= (\text{mineral fertilisers} + \text{manure production} + \text{net manure import/export, withdrawals} + \text{other organic fertilisers} + \text{seed and planting materials}) \\
- (\text{crop production} + \text{fodder production} + \text{crop residues removed})
\]

Potassium budget is not defined in the OECD/Eurostat Nutrient Budget Handbook. However, potassium budget can be calculated similarly to phosphorus (Berry, et al., 2003; Garcia-Ruiz, et al., 2012; Korsaeth, 2012; Kristaponyte, 2005; Pathak, et al., 2010).

**Box 3 – Definitions of the nutrient budget equation’s terms (Eurostat, 2013b)**

**Nutrient inputs:**

**Mineral fertilisers or inorganic fertilisers** refer to fertilisers whose nutrients are in the forms of minerals obtained by extraction or chemical processes.

**Manure production** refers to total livestock excreta during housing. It can be slurry (urine and faeces, often mixed with bedding elements) and solid manure. Livestock includes cattle, pigs, poultry, horses, sheep, goats and rabbits. Each type of manure from each type of livestock contains a certain quantity of nutrients (N and P) expressed through the “excretion factor”.

**Manure import/export withdrawals** refer to the manure trade with other countries and manure treatments. It reflects the quantity of manure effectively used on agricultural land.

**Other organic fertilisers** are organic fertilisers other than livestock excretion, i.e. organic products such as green manure (compost, crop residues, and grassland silage), sewage sludge or industrial waste that contain nutrients and that is applied on agricultural land.

**Biological nitrogen fixation (for N only)** refers to the nitrogen fixed in the soil by:

- leguminous crops such as soya bean, fava bean, peas etc., through the action of bacteria that live symbiotically in root nodules of the leguminous crops;
- grass-legume mixtures, in the case of temporary grassland mixed with leguminous forage crops;
- free-living N-fixing soil bacteria

**Atmospheric nitrogen deposition** is the deposition of nitrogen particles on land surfaces, through precipitation or dry deposition. Atmospheric phosphorus deposition also exists but is not taken into account in practice in phosphorus budget calculations.
Seed and planting materials contain nutrients that have to be taken into account. It refers to the sum of the nutrient content of all crop seeds, taking into account the amount sown.

**Nutrient outputs:**

- **Crop production** is the nutrient content of the total harvested crops for a defined area.
- **Fodder production** is the nutrient content in the fodder, which is harvested or grazed.
- **Crop residue outputs** are the nutrient content of the total amount of residues that are removed from land. In an ideal nutrient budget, the inputs from crop residues on the field should also be taken into account. However, due to the difficult data gathering this is not the case in practice.

The calculation of nutrient budgets is an important tool to help farmers to balance their fertilisation in order to optimise plant growth and fertilisation costs as well as to limit environmental impacts of the production and use of fertilisers. Indeed, inefficient use may occur through over-fertilisation, when the amount of fertiliser applied or the level of N, P and K already present in the soil exceeds crops’ nutritional needs.

A positive nutrient budget (**nutrient surplus**) indicates that nutrient losses may occur from the soil to water and/or air, with the associated risk of pollution for these compartments, while a negative nutrient budget (**nutrient deficiency**) designates soils that may lose their fertility in the long term. In cases where nutrient inputs equal nutrient outputs, the “closed nutrient budget” is called **nutrient balance**.

Nutrient surplus results in different potential pollution risks, i.e. emissions to air, leaching and run-off to ground and surface waters, and accumulation or depletion in soil (the latter if the losses to water and air are too high) (Eurostat, 2013b). The degree of a surplus only provides an indication of a potential pollution risk since the actual pollution may also depend on climatic conditions, soil type, and soil water saturation.

**Nutrient content** refers to the amount of nutrient in the soil compartment measured by local sampling at a given time. It is different from nutrient budget, which is calculated and based on existing agricultural models and takes into account the type of production and the agricultural practices. Thus, nutrient content of soil, as collected in the LUCAS database (Joint Research Centre, European Commission, 2013), does not reflect the level of surplus in nutrients.

### 4.1.2 Mapping nutrient budgets

Mapping nutrient budgets allows visualising the surplus and deficiencies within EU-28 according to their geographic location. To do so, a nutrient budget value is calculated for geographically located area units (whose size corresponds to a determined resolution such as one hectare or 10km²). This implies to find data (at the smallest geographical level possible) for each term in the budget equation and to create spatial representation of this data using GIS software (obtaining an “information layer”).

To obtain spatial information, data has to be available at the chosen resolution. If data is only available at a larger scale (e.g. regional or national level), data can be equally distributed between every area unit at the smaller scale or allocated by using additional information such as land use or yield.

**Land use** represents the human use of a specific area of land for a specific purpose, i.e. in the case of agriculture, the type of crop production, permanent and temporary pasture, vineyards and orchards. Land use is different from **land cover** that provides information on the observed physical occupation of land. It distinguishes artificial surfaces (urban areas, industrial areas, roads, etc.), agricultural territory (arable land, permanent crops, grassland, etc.), and forest or other natural surfaces. Land cover
corresponds to information observed from the ground or by remote sensing. For land use, observation data must be completed by additional information such as surveys to obtain information on the land use for the entire year. Both land use and land cover are determined for each geographically located area unit.

Thus, if fertiliser application data is only available at national scale, then the amount of fertiliser applied per NUTS$^{49}$ 2 region can be calculated considering the local surface area of the different crop types in agricultural land for each NUTS 2 region within the country. Hence, knowing the yield of each crop and the need for fertilisation, the user can calculate the average amount of fertiliser applied for each region.

The information needed to map nutrient budgets and the possible sources of information are described in Table 12.

| Table 12 – Parameters needed to calculate and to create spatial representation nutrient budget input and output values and possible sources of this information |
| --- | --- |
| **Needed parameters** | **Possible sources** |
| **Land use** | Land cover, in particular surface area and location of agricultural land (arable area, pasture, etc.) | Corine Land Cover dataset and maps |
| | Share of each crop type | Farm Structure Survey (FSS) and statistics from Eurostat |
| | Share of each livestock type |  |
| **Crop uptake** | Crop yield | Statistics from Eurostat, Food and Agriculture Organisation (FAO) |
| | Uptake coefficient per crop type | Data calculated by the Organisation for Economic Co-operation and Development (OECD), Eurostat |
| **Mineral fertilisers** | Application rate for each crop | Statistics from International Fertiliser Industry Association (IFA) / FAO and FSS (N, P; K) |
| **Livestock manure** | Livestock number per animal category | Statistics from FSS, Eurostat, FAO |
| | Excretion coefficient per head | Data calculated by OECD (N), FAO (N, P) and United Nations Framework Convention on Climate Change (UNFCCC) GHG inventory (N, P) |
| **N - atmospheric deposition** | Nitrogen atmospheric deposition | Estimated data from the European Monitoring and Evaluation Programme (EMEP) |
| **Biological N fixation** | Nitrogen fixation coefficient per crop | Data calculated by the OECD |

$^{49}$ NUTS: Nomenclature of Territorial Units for Statistics
4.2 Nutrient saturated regions in EU-28

The visual inspection of available maps of nutrient budgets allows the identification of areas that are saturated in nitrogen and/or phosphorus (see section 3.2.1). The magnitude of the surplus is the main criteria used to select the regions that will be studied in detail during the project.

Potassium is not considered in the selection of the saturated regions because no map of EU potassium budget is available. Furthermore, little data is available regarding potassium budgets at both farm and regional scale (Popp & Gransee, 2005; Cermak, et al., 2007; Nikolova & Popp, 2007; Oborn, et al., 2009).

Considering that the data used to elaborate the maps were produced some years ago, an analysis of the potential development in the nutrient surpluses between the data period and present is necessary to ensure that the observed saturated regions are still currently valid. Therefore, the past trends in nitrogen surplus (no data were available for phosphorus) and the potential development for parameters importantly influencing nitrogen surplus (such as manure input and livestock density for which more recent data was available – see section 3.2.2) were analysed.

This first analysis of saturated areas is complemented by a study of the fate of nutrient surpluses (see section 3.2.3). The nutrient fate provides information on the risk of pollution due to nutrient excess in one or several environmental compartments (soil, water, air) and consequently potential negative impacts on climate, biodiversity and human health. The nutrient fate depends on the soil type, the water system, the environmental conditions and the agricultural practices, and is a key aspect to consider while selecting the study areas. Indeed, an area can show a nutrient surplus without being susceptible to strong nutrient leaching, run-off or emissions in the short to medium term. For instance, while nitrogen load to water is generally closed to the agricultural N surplus (Windolf, et al., 2012), the load may be reduced or delayed depending on the soil type and the water system for a same N surplus, leading to additional emissions (Blicher-Mathiesen, et al., 2007).

In addition, in order to address the various situations that can occur in the EU, other criteria such as the type of production, the size of holdings, the production intensity and the soil and climate context, are taken into account while selecting the regions.

In short, by providing information on the potential type and the potential intensity of pollution that could affect the area (e.g. N leaching or concentration in groundwater and surface water) as well as on the farming systems (in particular livestock) and on the soil and climate conditions, the approach ensures the relevance and the variety of the set of selected regions.

4.2.1 Analysis of the available maps of nutrient budgets

To date, four maps of nutrient budgets based on the equations defined by the OECD/Eurostat in the 2007 handbooks were identified (see Table 13, Figure 23 to Figure 26). A detailed comparison of the maps characteristics is presented in Annex 7.
<table>
<thead>
<tr>
<th>Nutrient(s)</th>
<th>Year of data</th>
<th>Perimeter</th>
<th>Resolution</th>
<th>Nutrient balance formula used</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grizzetti (2007), Spatialised European Nutrient Balance</td>
<td>N &amp; P</td>
<td>2000</td>
<td>EU 15 100x100m, recalculated at 10 km² grid</td>
<td>Gross nitrogen balance = (mineral fertilisers + livestock manure + biological fixation + atmospheric deposition) – crop uptake</td>
</tr>
<tr>
<td>Leip (2011), Integrating nitrogen fluxes at the European scale</td>
<td>N</td>
<td>2002</td>
<td>EU 27</td>
<td>N surplus = input (mineral fertilisers + livestock manure excretion corrected for transport + other organic sources + left crop residues + biological fixation + atmospheric deposition) – total crop removal – total forage uptake</td>
</tr>
</tbody>
</table>
Nitrogen budget considered as very high: > 120 kg N/ha; high: 91 to 120 kg N/ha; moderate: 61 to 90 kg N/ha; low: 30 to 60 kg N/ha; very low: < 30 kg N/ha

Nitrogen budget considered as high to very high: > 100 kg N/ha; moderate: 50 to 100 kg N/ha; low: 25 to 50 kg N/ha; very low: < 25 kg N/ha

Nitrogen budget considered as high to very high: > 81 kg N/ha; moderate: 41 to 81 kg N/ha; low: 31 to 40 kg N/ha; very low: < 30 kg N/ha
The calculation and mapping methods used to create these maps are broadly similar. However, the nitrogen surplus maps cannot be directly compared. The data used differs, as does the graphical presentation. In particular, the resolution is not the same for all the maps. Moreover, the scale and related the colour codes differ and do not refer to the same amount of nitrogen surplus. Nevertheless, these maps present close results that enable an identification of the main saturated areas for the period 2000-2005. Table 14 qualitatively classifies the regions and countries according to the magnitude of the surplus.

Table 14 – European areas saturated in nutrients, as identified in Figure 23 to Figure 26

<table>
<thead>
<tr>
<th>Surplus areas</th>
<th>Level of N surplus</th>
<th>Level of P surplus</th>
</tr>
</thead>
<tbody>
<tr>
<td>Areas with a moderate and high surplus for one or two nutrients</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td></td>
<td></td>
</tr>
<tr>
<td>South</td>
<td>++++</td>
<td>++++</td>
</tr>
<tr>
<td>Belgium</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Flanders</td>
<td>++++</td>
<td>+++</td>
</tr>
<tr>
<td>Italy</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Po Valley</td>
<td>++</td>
<td>++</td>
</tr>
<tr>
<td>Lombardy(^\text{54})</td>
<td>++++</td>
<td>++++</td>
</tr>
<tr>
<td>Denmark</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mid Jutland</td>
<td>+++</td>
<td>+++</td>
</tr>
<tr>
<td>South Jutland</td>
<td>+++</td>
<td>++</td>
</tr>
</tbody>
</table>

\(^{53}\) Phosphorus budget considered as very high: > 25 kg P/ha; high: 16 to 25 kg P/ha; moderate: 11 to 15 kg P/ha; low: 6 to 10 kg P/ha; very low: < 5 kg P/ha

\(^{54}\) Part of the Po Valley region
As presented in Table 14, several regions can be identified as saturated for nitrogen, phosphorus or both, with different level of saturation. In the next section, the highly saturated regions are explored as a priority for the selection of the regions. However, in order to ensure a diversity of different situations and contexts among the selected regions, moderately saturated areas are also investigated.

Regarding information on potassium, it appears that potassium budgets are poorly studied and no map of potassium budgets exists for Europe. Some maps exist for China and Brazil (Naumov & Prado, 2008).

4.2.2 Analysis of the applicability of map results for 2000-2005 to the current period

The most recent map uses data from 2005. While the comparison of maps from 2000 and 2005 does not show evidence of significant change during that period, it is important to ensure that the main saturated areas previously identified can still be considered as saturated areas. Indeed, it would mean that the issues related to the excess of nutrients have not been solved and hence it is still relevant to propose good practices to reduce the excess of nutrients in these regions.

Past trends of nutrient surplus

Nitrogen surplus

The analysis of nitrogen surplus every five years between 1985 and 2005 shows a global decrease (Bouraoui, et al., 2011). For some regions such as Lithuania, Bulgaria and Greece, the surplus has even become very low or null (Figure 27).
Between 1985 and 1990, several Member States such as Bulgaria, Czech Republic, Denmark, Hungary, the Netherlands and Poland experienced a significant drop in the nutrient budget per hectare (Bouraoui, et al., 2011). From 1990 to 2000, the Netherlands and Denmark continued to present a considerable decrease. The nutrient budget of Cyprus also showed considerable decrease, and so did Slovakia and Greece, as noted by the OECD (2013). A few Member States showed a different trend with an increasing N surplus from 1985 to 2005. Spain, in particular in the Murcia region, Ireland, and Hungary increased the N surplus between 1990-92 and 1998-2000 (Bouraoui, et al., 2011; OECD, 2013). From 2000 to 2005 (Bouraoui, et al., 2011) and in 2007-2009, the nutrient surpluses continued to decrease for the majority of the Member States, the highest decreases were observed in Netherlands, Belgium, Luxembourg, Portugal, and Hungary. However, the Netherlands, Belgium, and Luxembourg still presented high surpluses in 2007-2009 (204, 121 and 75 kg N/ha, respectively). Overall, while some areas do not appear as N-saturated anymore, several regions with high nitrogen surplus in 1985 still remained saturated in 2005 and 2007-2009 (OECD, 2013).

The observed decrease is probably due to the change of practices – such as improved nitrogen use and the increased use of N₂ fixing crops, the economy and rising fertiliser prices, and policy reforms (Bouraoui, et al., 2011). More specifically, the introduction of milk quotas (1984) and set aside (1988) in the Common Agricultural Policy (CAP) have played an important role in the intensification level of agriculture and the more efficient use of fertiliser. The implementation of the Nitrates Directive in 1991 (Council Directive 91/676/EEC) has accelerated the decrease in fertiliser use, in particular with the introduction of requirements for a balanced fertilisation and a limit for manure application in Nitrate Vulnerable Zones (NVZ)\(^5\). Velthof et al. (2014) highlights the trends in total N leaching and run-off to groundwater and surface waters and in total NH₃, N₂O, and NOₓ emissions with and without implementation of the Nitrates Directive for the period 2000-2008. In both scenarios, N emissions

\(^5\) Areas that drain into nitrate polluted waters, or waters which could become polluted by nitrates
decrease, but the emissions of the scenario with implementation of the Nitrates Directive are smaller than without the Nitrates Directive, which provides evidence for their environmental effectiveness. However, the effects differ in particular among the regions, where regions with high livestock density and fertiliser consumptions show the largest impacts and decrease. The CAP reform in 2003 introduced important modifications that may have stimulated the decrease in nitrogen surplus after 2005. One example is the implementation of “decoupled” aid, which refers to a subsidy system where payments to farmers are independent of the obtained yields. Another example is the introduction of cross-compliance obligations, linked with good agricultural and environmental practices (GAECs). Farmers must comply with GAECs in order to obtain full direct subsidies. Since 2005, a decline in nutrient surpluses could possibly have occurred considering the regular updates of the national and regional policies affecting the nutrients surplus such as the nitrates action plans or the Rural Development Programs and the improvement of the agricultural practices.

**Phosphorus surplus**

Regarding phosphorus, even if no information is available at regional level, a decrease of the surplus can be observed at the national scale (Bouraoui, et al., 2011; OECD, 2013). A significant decrease in phosphorus surplus per hectare was observed between 1985 and 1990 in several Member States such as Bulgaria, Croatia, Czech Republic, Denmark, Germany, Hungary, Poland, Slovak Republic, Slovenia and Romania. This decrease continued after 1990, but to a lesser extent except for Belgium/Luxembourg that showed a very significant decrease, by nearly 75% in 5 years, and for Poland, whose surplus slightly increases after 2000 (Bouraoui, et al., 2011). These figures are confirmed by the OECD data, however, the decrease was more gradual: 66% of decrease for Luxembourg and 33% for Belgium from 1998-2000 to 2007-2009 (OECD, 2013). Only Slovenia and Spain had increasing P surplus between 1990 and 2000. From 2000, the P budget for Belgium and Luxembourg kept decreasing, reaching zero P surplus for Luxembourg in 2007-2009 and 6 kg P/ha for Belgium. Poland is the only country that shows an increase, with a surplus of 5 kg P/ha in 2007-2009 compared to 3 kg P/ha in 1998-2000. Some Member States even experience a deficit in potassium in 2007-2009 (Estonia, Greece, Hungary, Italy and Slovak Republic), which can undermine soil fertility. For other countries, the trend of the previous decade still remains, with the highest P surplus decrease for France and Germany56 (OECD, 2013).

The phosphorus surplus decrease may have the same drivers as for nitrogen. Indeed, the introduction of policies and respective measures57 that aimed at improving resource efficiency in agriculture influence the decrease in nitrogen as well as the decrease in phosphorus. Contrary to nitrogen, the P surplus decrease is not directly due to the implementation of the Nitrates Directive since only a few Member States address phosphorus and its surplus in their legislation, such as Belgium (Flanders), Estonia, France (Brittany), Germany, Ireland, Sweden and the Netherlands to date (Cherrier, et al., 2014; Amery & Schoumans, 2014). However, it is also to be noted that a number of mandatory measures and good agricultural practices under the Nitrates Directive, even if designed to prevent and reduce nitrate pollution, have also an impact in terms of prevention and reduction of phosphorus run-off. Thus, the decrease in P surplus may be due to the improvement of practices leading to the use of lower amounts.

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56 Spain also presented one of the strongest decreases according to the OECD (2013). However, Bouraoui et al. (2011) reports a slight increase in the Spanish P surplus for this period. Thus, considering the contradictory results, Spain is not mentioned along with France and Germany.

of phosphorus fertilisers and improved storage and application of livestock manure (Cherrier, et al., 2014; Smit, et al., 2009).

Hence, considering the overall decrease of the nutrient surpluses in numerous areas, some saturated regions, probably the regions with the lowest level of surplus in 2005, may not be significantly saturated anymore. Further indications supporting this are provided in the analysis of parameters influencing nutrient surpluses at the national level, which is presented below.

**Potential development of parameters influencing nutrient surpluses between 2000-2005 and present**

The development of the nutrient surpluses (N and P) is closely linked to the type of farming system. Livestock farms generally induce high nitrogen and phosphorus surplus (Leip, et al., 2011; Jarvis, et al., 2011). Moreover, a close correlation can be observed between nutrient surplus and manure inputs for both N (Figure 25 vs. Figure 28) and P (Figure 26 vs. Figure 29) both are also correlated to the livestock density\(^58\) (Figure 30) (de Vries, et al., 2011a; Van Grinsven, et al., 2012).

Large phosphorus surpluses also occur in areas of intensive horticultural production while the lowest P surpluses are recorded on cereal farms and extensive cattle farms (Torrent, et al., 2007; Bomans, et al., 2005). However, a map showing the contribution of vegetables, horticultural products and fruit outputs to the total agricultural outputs for the EU (Figure 31) does not match that of phosphorus surplus (Figure 26). Moreover, data at regional level on the surface areas dedicated for horticulture production are only available from 2010 to 2012 (Eurostat, 2013a). Thus, the development of horticulture production cannot be reliably compared to the calculated surplus. Therefore, the potential trends are estimated only through the evolution of livestock density, and nitrogen and phosphorus manure inputs.

\(^{58}\) Livestock density is the number of Livestock Unit (LSU) per ha or Utilised Agricultural Area (UAA)
Figure 28 – Map of manure nitrogen application (kg N/ha of total area) per sub-basin for 2005 \(^{59}\)

Figure 29 – Map of manure phosphorus application (kg P/ha of total area) per sub-basin for 2005 \(^{60}\)

Figure 30 – Livestock units per hectare of utilised agricultural area for 2007 \(^{61}\)

Figure 31 – Vegetables, horticultural products and fruit output as share of agricultural goods output for 2007 (kg N per ha of agricultural land per year) \(^{62}\)

---

\(^{59}\) Manure nitrogen application considered as very high: > 121 kg N/ha; high: 81 to 120 kg N/ha; moderate: 41 to 80 kg N/ha; low: < 40 kg N/ha

\(^{60}\) Manure phosphorus application considered as very high: > 41 kg P/ha; high: 31 to 40 kg P/ha; moderate: 21 to 30 kg P/ha; low: < 20 kg P/ha

\(^{61}\) Livestock density considered as high: > 1.2 LSU/ha UAA; moderate: 0.7 to 1.2 LSU/ha UAA; low: < 0.7 LSU/ha UAA

\(^{62}\) Outputs considered as very high: > 20 kg N/ha UAA/year; moderate: 10 to 20 kg N/ha UAA/year; low: < 10 kg N/ha UAA/year
The development of livestock density and manure application can provide information on the trends of nutrient budgets. The average European livestock density is about 1 Livestock Unit (LSU) / ha of Utilised Agricultural Area (UAA). The European livestock density decreased by 6.2 % between 2005 and 2010 (Eurostat, 2013a). However, most of the densest regions in 2005 experienced an increase in livestock density during the period 2005-2010, reflecting an intensification of production systems (Table 15).

### Table 15 – Regions with high livestock density (> 1.7 LSU/ha UAA) in 2005 and/or 2010

<table>
<thead>
<tr>
<th>Member State</th>
<th>NUTS 2 region</th>
<th>Livestock density (LSU/ha UAA) in 2005</th>
<th>Livestock density (LSU/ha UAA) in 2010</th>
<th>Variation compared to 2005</th>
</tr>
</thead>
<tbody>
<tr>
<td>Belgium</td>
<td>Prov. Antwerpen (Flanders)</td>
<td>5.94</td>
<td>5.89</td>
<td>-0.8 %</td>
</tr>
<tr>
<td>Belgium</td>
<td>Prov. Limburg (Flanders)</td>
<td>3.03</td>
<td>2.92</td>
<td>-3.6 %</td>
</tr>
<tr>
<td>Belgium</td>
<td>Prov. Oost-Vlaanderen (Flanders)</td>
<td>3.87</td>
<td>3.83</td>
<td>-1.0 %</td>
</tr>
<tr>
<td>Belgium</td>
<td>Prov. West-Vlaanderen (Flanders)</td>
<td>5.80</td>
<td>5.96</td>
<td>2.8 %</td>
</tr>
<tr>
<td>Belgium</td>
<td>Prov. Luxembourg</td>
<td>1.93</td>
<td>1.90</td>
<td>-1.9 %</td>
</tr>
<tr>
<td>Denmark</td>
<td>Syddanmark (South of Denmark)</td>
<td>1.83</td>
<td>2.01</td>
<td>9.8 %</td>
</tr>
<tr>
<td>Denmark</td>
<td>Midtjylland (Mid Jutland)</td>
<td>1.9</td>
<td>2.10</td>
<td>10.5 %</td>
</tr>
<tr>
<td>Denmark</td>
<td>Nordjylland (North Jutland)</td>
<td>2.02</td>
<td>2.24</td>
<td>10.9 %</td>
</tr>
<tr>
<td>Germany</td>
<td>Niedersachsen</td>
<td>1.72</td>
<td>1.74</td>
<td>1.1 %</td>
</tr>
<tr>
<td>Germany</td>
<td>Nordrhein-Westfalen</td>
<td>1.89</td>
<td>1.91</td>
<td>1.2 %</td>
</tr>
<tr>
<td>Greece</td>
<td>Ipeiros</td>
<td>2.12</td>
<td>1.24</td>
<td>-41.5 %</td>
</tr>
<tr>
<td>Spain</td>
<td>Galicia</td>
<td>1.75</td>
<td>1.97</td>
<td>12.3 %</td>
</tr>
<tr>
<td>Spain</td>
<td>Cataluña</td>
<td>2.26</td>
<td>2.39</td>
<td>5.8 %</td>
</tr>
<tr>
<td>France</td>
<td>Bretagne</td>
<td>2.77</td>
<td>2.84</td>
<td>2.5 %</td>
</tr>
<tr>
<td>Italy</td>
<td>Lombardy</td>
<td>2.66</td>
<td>2.77</td>
<td>4.1 %</td>
</tr>
<tr>
<td>Malta</td>
<td>Malta</td>
<td>4.50</td>
<td>3.64</td>
<td>-19.1 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Friesland (NL)</td>
<td>2.14</td>
<td>2.35</td>
<td>9.8 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Drenthe</td>
<td>1.83</td>
<td>2.02</td>
<td>10.4 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Overijssel</td>
<td>4.06</td>
<td>4.56</td>
<td>12.3 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Gelderland</td>
<td>5.15</td>
<td>5.47</td>
<td>6.2 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Utrecht</td>
<td>3.45</td>
<td>3.77</td>
<td>9.3 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Noord-Brabant</td>
<td>6.76</td>
<td>7.54</td>
<td>11.5 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>Limburg (NL)</td>
<td>6.03</td>
<td>6.92</td>
<td>14.8 %</td>
</tr>
<tr>
<td>Portugal</td>
<td>Região Autónoma dos Açores (PT)</td>
<td>1.52</td>
<td>1.71</td>
<td>12.6 %</td>
</tr>
<tr>
<td>Portugal</td>
<td>Região Autónoma da Madeira (PT)</td>
<td>2.96</td>
<td>2.44</td>
<td>-17.6 %</td>
</tr>
</tbody>
</table>

**Legend:**

The colour gradient from green to red represents the highest negative variation up to the highest positive variation in livestock density. Colours are attributed by percentile. The median of the variation percentage, i.e. the 50 percentile, is represented by the colour yellow.

Source: (Eurostat, 2013a)

Regarding manure application, due to the lack of data at regional level, manure application trends can be studied at national level only (Table 16). In 2010, the application of manure in EU-28 countries was on average 8.4 % lower than in 2000 and 3.4 % lower than 2005 (FAOSTAT, 2014). For the period

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63 UAA is the total area taken up by arable land, permanent grassland, permanent crops and kitchen gardens

64 The regions with high livestock density are the regions with the 10 % highest density (>1.7 LSU/ha UAA).

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2005-2010, 23 countries showed a decline in manure application ranging from 0.1 % for Latvia to -26.1 % for Malta. Only five Member States presented an increased amount of manure applied from 2005 to 2010.

Table 16 – Variation of manure applied (N content) in the EU-28 compared to 2005

<table>
<thead>
<tr>
<th>Member State</th>
<th>Presence of highly saturated areas according to Table 15</th>
<th>Presence of moderately saturated areas according to Table 15</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td></td>
<td></td>
<td>0.6 %</td>
<td>-1.1 %</td>
<td>1.0 %</td>
<td>-0.2 %</td>
<td>0.9 %</td>
</tr>
<tr>
<td>Belgium</td>
<td>x</td>
<td></td>
<td>-1.9 %</td>
<td>-2.6 %</td>
<td>-2.8 %</td>
<td>-2.2 %</td>
<td>-1.6 %</td>
</tr>
<tr>
<td>Bulgaria</td>
<td></td>
<td></td>
<td>-3.5 %</td>
<td>-0.4 %</td>
<td>-7.6 %</td>
<td>-14.5 %</td>
<td>-17.1 %</td>
</tr>
<tr>
<td>Croatia</td>
<td></td>
<td></td>
<td>10.4 %</td>
<td>4.7 %</td>
<td>-5.6 %</td>
<td>-1.5 %</td>
<td>-4.0 %</td>
</tr>
<tr>
<td>Cyprus</td>
<td></td>
<td></td>
<td>-5.8 %</td>
<td>-3.5 %</td>
<td>-3.2 %</td>
<td>-2.8 %</td>
<td>-3.5 %</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>x</td>
<td></td>
<td>4.2 %</td>
<td>1.4 %</td>
<td>-7.3 %</td>
<td>-9.3 %</td>
<td>-10.0 %</td>
</tr>
<tr>
<td>Denmark</td>
<td>x</td>
<td></td>
<td>-1.6 %</td>
<td>0.3 %</td>
<td>-4.9 %</td>
<td>-5.6 %</td>
<td>-2.6 %</td>
</tr>
<tr>
<td>Estonia</td>
<td></td>
<td></td>
<td>-1.0 %</td>
<td>-3.2 %</td>
<td>-2.2 %</td>
<td>-2.9 %</td>
<td>-3.7 %</td>
</tr>
<tr>
<td>Finland</td>
<td>x</td>
<td></td>
<td>-0.4 %</td>
<td>-2.6 %</td>
<td>-2.1 %</td>
<td>-5.5 %</td>
<td>-6.0 %</td>
</tr>
<tr>
<td>France</td>
<td>x</td>
<td></td>
<td>-0.7 %</td>
<td>-0.6 %</td>
<td>-4.1 %</td>
<td>-5.4 %</td>
<td>-6.0 %</td>
</tr>
<tr>
<td>Germany</td>
<td>x</td>
<td></td>
<td>-1.8 %</td>
<td>-0.5 %</td>
<td>0.5 %</td>
<td>1.1 %</td>
<td>-0.4 %</td>
</tr>
<tr>
<td>Greece</td>
<td></td>
<td></td>
<td>-0.4 %</td>
<td>0.1 %</td>
<td>0.4 %</td>
<td>0.6 %</td>
<td>1.1 %</td>
</tr>
<tr>
<td>Hungary</td>
<td>x</td>
<td></td>
<td>-1.3 %</td>
<td>-2.5 %</td>
<td>-3.8 %</td>
<td>-10.0 %</td>
<td>-11.5 %</td>
</tr>
<tr>
<td>Ireland</td>
<td>x</td>
<td></td>
<td>-0.8 %</td>
<td>-3.5 %</td>
<td>-2.0 %</td>
<td>-2.4 %</td>
<td>-4.7 %</td>
</tr>
<tr>
<td>Italy</td>
<td>x</td>
<td></td>
<td>-1.0 %</td>
<td>-0.7 %</td>
<td>3.1 %</td>
<td>3.0 %</td>
<td>4.4 %</td>
</tr>
<tr>
<td>Latvia</td>
<td></td>
<td></td>
<td>1.6 %</td>
<td>1.3 %</td>
<td>5.2 %</td>
<td>0.0 %</td>
<td>-0.1 %</td>
</tr>
<tr>
<td>Lithuania</td>
<td></td>
<td></td>
<td>2.6 %</td>
<td>5.1 %</td>
<td>-2.9 %</td>
<td>-5.6 %</td>
<td>-5.6 %</td>
</tr>
<tr>
<td>Luxembourg</td>
<td></td>
<td></td>
<td>-1.9 %</td>
<td>1.0 %</td>
<td>2.5 %</td>
<td>3.1 %</td>
<td>4.6 %</td>
</tr>
<tr>
<td>Malta</td>
<td></td>
<td></td>
<td>-2.3 %</td>
<td>-11.9 %</td>
<td>-16.9 %</td>
<td>-23.9 %</td>
<td>-26.1 %</td>
</tr>
<tr>
<td>Netherlands</td>
<td>x</td>
<td></td>
<td>-1.2 %</td>
<td>0.9 %</td>
<td>4.8 %</td>
<td>5.4 %</td>
<td>6.6 %</td>
</tr>
<tr>
<td>Poland</td>
<td>x</td>
<td></td>
<td>1.3 %</td>
<td>1.1 %</td>
<td>-5.8 %</td>
<td>-9.6 %</td>
<td>-9.4 %</td>
</tr>
<tr>
<td>Portugal</td>
<td></td>
<td></td>
<td>0.5 %</td>
<td>0.0 %</td>
<td>1.5 %</td>
<td>0.6 %</td>
<td>-0.3 %</td>
</tr>
<tr>
<td>Romania</td>
<td></td>
<td></td>
<td>1.5 %</td>
<td>3.0 %</td>
<td>-0.4 %</td>
<td>-3.6 %</td>
<td>-7.9 %</td>
</tr>
<tr>
<td>Slovakia</td>
<td></td>
<td></td>
<td>-2.5 %</td>
<td>-9.9 %</td>
<td>-19.1 %</td>
<td>-17.3 %</td>
<td>-21.1 %</td>
</tr>
<tr>
<td>Slovenia</td>
<td></td>
<td></td>
<td>0.0 %</td>
<td>0.9 %</td>
<td>2.8 %</td>
<td>-5.6 %</td>
<td>-5.1 %</td>
</tr>
<tr>
<td>Spain</td>
<td>x</td>
<td>x</td>
<td>1.6 %</td>
<td>3.0 %</td>
<td>0.4 %</td>
<td>0.9 %</td>
<td>-1.0 %</td>
</tr>
<tr>
<td>Sweden</td>
<td>x</td>
<td></td>
<td>-3.1 %</td>
<td>-4.3 %</td>
<td>-5.3 %</td>
<td>-7.6 %</td>
<td>-7.5 %</td>
</tr>
<tr>
<td>United Kingdom</td>
<td>x</td>
<td></td>
<td>-1.4 %</td>
<td>-4.4 %</td>
<td>-5.2 %</td>
<td>-6.7 %</td>
<td>-7.1 %</td>
</tr>
</tbody>
</table>

Legend: The colour gradient from green to red represents the highest negative variation up to the highest positive variation in livestock density. Colours are attributed by percentile. The median of the variation percentage, i.e. the 50 percentile, is represented by the colour yellow.

Source: (FAOSTAT, 2014)

Among the Member States that had high nutrient surplus for 2000-2005 (see Table 14), the amount of manure applied has generally been stable during 2005-2010 in Belgium (-1.6 %), Denmark (-2.6 %), Germany (-0.4 %), and Spain (-1.0 %). In contrast, the amount of manure applied in the Netherlands and Italy has increased by 4.4 % and 6.6 %, respectively. Thus, it is reasonable to hypothesise that the nutrient surplus observed in 2005 within these Member States globally remained in 2010, keeping in
mind that variation at national level may not reflect the variation at the level of individual regions. The nutrient surplus data from 2007-2009 reflect this with a nitrogen surplus above 50 kg N/ha UAA for the Netherlands, Belgium, Denmark, Germany, France, and Ireland. Italy and Spain also present surplus at national level but lower than the other Member States (respectively 28 and 14 kgN/ha UAA). However, it is also to be noted that Italy shows a negative phosphorus budget at the national level (-3 kg P/ha UAA). (OECD, 2013).

Among the regions that are saturated in one or two nutrients, France, Czech Republic, Finland, Hungary, Ireland, Poland, Sweden and United Kingdom presented a decrease in the amount of manure applied between 2005 and 2010 of 4.7 % for Ireland and up to 11.5 % for Hungary. For these Member States, the high variation may suggest that the level of saturation in these areas had changed from 2005 to 2010, in particular for areas with lowest saturation level such as Czech Republic, Finland, Hungary, Poland, Sweden, and the United Kingdom (see Table 14). Hence, the uncertainty on the saturation level for these regions has to be taken into account when selecting the focus regions for this project in order to ensure the robustness of the selection. According to the average 2007-2009 data from the OECD, Hungary did not show nutrient excess at national level anymore with a nitrogen budget of 1 kg N/ha UAA and – 10 kg P/ha UAA. The other Member States still present a remarkable nitrogen surplus. Among them, Czech Republic and Sweden show a balanced phosphorus budget (i.e. 0 kg P/ha UAA) (OECD, 2013).

4.2.3 Analysis of the pollution risk

The susceptibility of the environment to be polluted in case of nutrient surpluses is the second criteria considered for the selection of the nutrient saturated regions. In the case of nutrient excess, the nutrients can either be leached or stored in soil, and for nitrogen also emitted to air. In addition to agricultural practices, the fate of the nutrients in case of surplus mostly depends on soil characteristics (soil texture, structure, soil organic matter (SOM)\(^{65}\) content, etc.), climate conditions (temperature, precipitation) and water availability.

The available maps address the following aspects: nitrogen air emissions, nitrogen leaching/run-off and phosphorus load in water (Figure 32 to Figure 38). The most recent maps for nitrogen come from Velthof et al. (2014) who used data from 2008. Other maps on nitrogen fate are available in Annex 9 and Annex 10. No data is available for phosphorus leaching and run-off, but data on phosphorus load in water is available in the study of Bouraoui et al. (2011). The available maps show nutrients’ fate, whatever the source of nutrients is. Thus, it should be kept in mind that nutrients may come from agriculture but also from other sources such as wastewater treatment plants. However, it provides a first idea of issues caused by nutrients in the EU that can potentially be further aggravated by unsustainable nutrient management in agriculture.

\(^{65}\) SOM is defined by in Commission’s proposal for a Soil Framework Directive (COM(2006) 232 final as “the organic fraction of the soil, excluding undecayed plant and animal residues, their partial decomposition products, and the soil biomass”
Nitrogen emissions from agriculture

Figure 32 – N\textsubscript{2}O emissions from agriculture in the EU-27 in 2008 (kg N per ha of agricultural land per year)\textsuperscript{66}

Source: (Velthof, et al., 2014)

Figure 33 – NH\textsubscript{3} emissions from agriculture in the EU-27 in 2008 (kg N per ha of agricultural land per year)\textsuperscript{67}

Source: (Velthof, et al., 2014)

Figure 34 – NO\textsubscript{x} emissions from agriculture in the EU-27 in 2008 (kg N per ha of agricultural land per year)\textsuperscript{68}

Source: (Velthof, et al., 2014)

\textsuperscript{66} N\textsubscript{2}O emissions considered as high: > 5 kg N\textsubscript{2}O-N/ha/year; moderate: 2 to 5 kg N\textsubscript{2}O-N/ha/year; low: < 2 kg N\textsubscript{2}O-N/ha/year

\textsuperscript{67} NH\textsubscript{3} emissions considered as high: > 40 kg NH\textsubscript{3}-N/ha/year; moderate: 15 to 40 kg NH\textsubscript{3}-N/ha/year; low: < 15 kg NH\textsubscript{3}-N/ha/year

\textsuperscript{68} NO\textsubscript{x} emissions considered as high: > 2 kg NO\textsubscript{x}-N/ha/year; moderate: 1 to 2 kg NO\textsubscript{x}-N/ha/year; low: < 1 kg NO\textsubscript{x}-N/ha/year
According to Velthof et al. (2014), the regions with highest emissions are:

- **N₂O emissions**: in wet regions with high nitrogen inputs to agricultural soils such as Ireland (south and east), Belgium, and the Netherlands. To a lesser extent, France (Brittany), United Kingdom (south-west), Germany (north-west and south), and Italy (Po Valley) are also concerned.

- **NH₃ emissions**: in areas with intensive livestock breeding such as France (Brittany), Belgium (Flanders), Italy (Po valley), the Netherlands (south-east) and Germany (north-west). To a lesser extent, Ireland, United Kingdom (south-west), Denmark, Poland, Czech Republic, Austria, Slovakia, Portugal and Spain (Murcia and Galicia) are also concerned.

- **NOₓ emissions**: in areas with high N inputs such as Belgium (Flanders) and the Netherlands (south-east). To a lesser extent, France (Brittany), Ireland, United Kingdom (east), Italy (Po valley), Germany (north-west and south), Denmark, and Poland (centre) are also concerned.

The contribution of agriculture to total NOₓ emissions is less than 5 % while the contribution of agriculture to NH₃ emissions is more than 90 % (de Vries, et al., 2011b), while N₂O emissions from agricultural soils are the largest source of N₂O in the EU (Eurostat, 2012b). Hence, the selection of saturated regions will mostly focus on N₂O and NH₃ emissions.

The regions that present high emissions for both N₂O and NH₃ are Belgium (Flanders) and the Netherlands (south-east). The regions that present high emissions for N₂O or NH₃ and moderate emissions for the other emission type are: France (Brittany), Ireland (south and east), Italy (Po Valley), and Germany (north-west). These regions are all identified as highly or moderately saturated regions, confirming the importance to select them.

Among the other saturated regions, the United Kingdom (south-west) presents moderate emissions for both N₂O and NH₃. Denmark, Poland, Czech Republic and Spain (Murcia), though not having high N₂O emissions, presents high NH₃ emissions.

**Leaching and run-off of nitrogen surplus**

The fate of nitrogen in water cannot solely be explained by nitrogen inputs, but also by regional context such as soil type and climate conditions that affect the risk of leaching, run-off and emissions (see also sections 2.1 and 2.3.3). Moreover, the reader should keep in mind the several possible sources of nitrogen concentration and the uncertainty regarding the excess of nutrient for the current period.
Among the regions with high and moderate nitrogen leaching and run-off (>20 kg N/ha/year) there are Belgium (Flanders), the Netherlands (South), Ireland, United kingdom (south and east), France (Brittany and centre), Denmark (west), Italy (Po valley, Lombardy), Germany (north-west), Poland (west) and Spain (Galicia) (Figure 35 and Figure 36). It corresponds to areas with high livestock density and high manure inputs (Figure 28 to Figure 30). Except for Spain, all the regions mentioned show a medium level of nutrient saturation.

A difference can be observed between the regions with high nitrogen concentration in groundwater and regions with high nitrogen concentrations in surface water (Figure 37). Only few regions are affected by high N concentration in groundwater while more regions show high N concentration in surface water. The regions that show high nitrogen concentration in both groundwater and surface water are Belgium (Flanders), the Netherlands (south), and Czech Republic (Prague). Among the regions that present high concentration in groundwater or surface water and moderate to high concentration in the other type of water there are Germany (north-west) and Czech Republic (centre). Regions that present moderate concentration of N in both groundwater and surface water are Spain (Murcia), Poland (centre West), Sardinia, and Czech Republic (west). Among the other saturated regions, France (Brittany), the United Kingdom (south), and Denmark show moderate concentration of nitrogen in surface water.

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69 Nitrogen leaching and run-off considered as high: > 40 mg N/L; moderate: 20 to 40 mg N/L; low: < 20 mg N/L.

70 Nitrogen leaching and run-off considered as high: > 50 mg N/L; moderate: 20 to 50 mg N/L; low: < 20 mg N/L.
Figure 37 – Predicted nitrate concentration in groundwater (left) and surface water (right) for the NUTS 2 regions with the MITERRA model for the year 2005

Figure 38 – Average concentration level of phosphorus in rivers for the year 2008 (mg PO$_4^{3-}$/L)

Phosphorus load in water

The highest concentrations of phosphate in river basins are found in the United Kingdom (centre and south-east), Belgium, the Netherlands (south), Italy (south), Spain (Catalonia), Portugal (centre),
Romania (south), and Bulgaria. High concentrations can also be found in the east of the United Kingdom, the centre of Spain, eastern Poland, western Czech Republic, and Hungary. Above 0.1 mg P/L, phosphate concentration is sufficiently high to induce freshwater eutrophication (Lynche-Solheim, et al., 2010). As for nitrogen, the maps of phosphorus fate do not distinguish the source of phosphorus.

It should be noted that the concentration of phosphate in rivers is not necessarily correlated to the areas that present high phosphorus surplus. Among these regions, only Catalonia, Flanders, Poland, and south Netherlands are saturated in phosphorus. The high concentration level of phosphorus is closely related to high population density and intensive agriculture (Lynche-Solheim, et al., 2010). No clear link can be noticed between livestock density (Figure 30) and phosphorus concentration in rivers. However, a correlation can be observed with horticulture production (Figure 31), which confirms that regions within areas dedicated to this production system have to be selected.

### 4.2.4 Complementary criteria used to select saturated regions

The diversification of the selection of regions is essential to address the various nutrient surplus situations that can occur in the EU-28. This allows the providing of more specific solutions to as many concerned farming systems as possible. As mentioned previously, the drivers of nutrient surplus are the farming systems, in particular agricultural practices, and soil and climate conditions. Therefore, the regions are selected according to the variety of situations regarding these aspects:

- **Farming systems**
  - **Type of agricultural production**: livestock production and horticulture production
  - **Size of holdings**: size of holdings depends on the type of production system, but also on the economic, social, and cultural context. Thus, the management techniques chosen by farmers are different for a large holding than for a small holding, in addition to the influence of the economic and human resources available.
  - **Intensity level of the farming system**: the farmer’s choice of farming system is influenced by a combination of soil, livestock, water, landscape, and economic conditions. Farming systems are characterised by different levels of intensity, ranging from intensive systems, focused on maximising production, to more extensive agriculture, focused on limiting inputs, field operations and/or ensuring the ecological sustainability of the production system. Therefore, intensive and extensive systems imply the use of different practices that highly affect nutrient budget.

- **Soil and climate conditions**
  - **Soil condition**: soil characteristics such as soil texture, soil structure and soil organic matter content highly affect water and nutrient retention capacity of the soil. Depending on the texture, water and nutrients are more or less retained in the soil. **Soil texture** may also induce anaerobic conditions leading the nitrogen emissions. **Soil structure** drives the soil water retention (SWR) capacity. Thus, soil with low SWR capacity will more quickly be saturated with water than a soil with a high SWR capacity, inducing leaching and/or run-off and possibly nutrient pollution of groundwater and surface water. **Soil structure**

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71 Nitrate concentration in groundwater considered as high: > 50 mg NO₃⁻/L; moderate: 25 to 50 mg NO₃⁻/L; low: < 25 mg NO₃⁻/L. Nitrogen concentration in groundwater considered as high: > 10 mg N/L; moderate: 5 to 0 mg NO₃⁻/L; low: < 5 mg NO₃⁻/L.

72 Phosphorus load considered as very high: > 0.2 mg PO₄³⁻-P/L; high: 0.1 to 0.2 mg PO₄³⁻-P/L; moderate: 0.05 to 0.1 mg PO₄³⁻-P/L; low: < 0.05 mg PO₄³⁻-P/L.
**organic matter** is a key indicator of soil quality and productivity, and is influenced by inputs via crop residues and manure application and outputs via decomposition, which is affected by tillage practice. **Soil erosion** affects nutrient budget since it decreases nutrient content of soil and soil nutrient (and water) retention capacity, potentially reducing assimilation of nutrients by plant.

- **Water availability**: water availability depends on the resource available, related to precipitation and water abstraction level by users and the soil water retention capacity. Soil water content is one of the factors that determine nutrient concentrations in the soil solution and their availability for plants since nutrients are available to plants only in a dissolved form in the soil solution.

- **Climate condition**: two main parameters affect nutrient cycles: temperature and precipitation. Temperature affects the degradation of organic matter, while precipitation can have an effect on erosion (in case of intense precipitation events) or water availability (in case of precipitation deficit).

Complete information and data regarding these parameters are available in Annex 11.
4.2.5 Summary of selected nutrient saturated regions

According to the available elements described above, the proposed selected areas are presented in the Table 17 and Table 18. For each broad area, NUTS 2 regions are proposed as a reference region representing the area and on which the study would focus. If no information is available for this region, data research can be extended to other NUTS 2 regions included in the area, showing similar context as the reference region. The regions that were not selected and the reasons for not having selecting them are presented in Annex 7.

Table 17 – Selected saturated regions

<table>
<thead>
<tr>
<th></th>
<th></th>
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<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Share</td>
<td>UAA (thousand s of ha)</td>
<td>N</td>
<td>P</td>
</tr>
<tr>
<td>West of France</td>
<td>Brittany (B52)</td>
<td>Yes (1 project)</td>
<td>Yes</td>
<td>62.8 % (2012)</td>
<td>1728 (2012)</td>
<td>++</td>
</tr>
<tr>
<td></td>
<td>Lombardy (ITC4)</td>
<td>Yes (4 projects)</td>
<td>Yes</td>
<td>n.a.</td>
<td>1228.3 (2012)</td>
<td>+++</td>
</tr>
<tr>
<td></td>
<td>Midtjylland (Mid-Jutland) (DK04)</td>
<td>Yes (6 projects)</td>
<td>Yes</td>
<td>61.1 % (2009)</td>
<td>793 (2012)</td>
<td>+++</td>
</tr>
<tr>
<td>Po Valley, Italy</td>
<td>Murcia (ES62)</td>
<td>Yes (5 projects)</td>
<td>Partly</td>
<td>50.4 % (2012)</td>
<td>569.9 (2012)</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td>North-Brabant (NL41)</td>
<td>Yes (1 project)</td>
<td>Yes</td>
<td>50.5 % (2009)</td>
<td>246.8 (2012)</td>
<td>+++</td>
</tr>
<tr>
<td>South-east Ireland</td>
<td>Southern and eastern (IE02)</td>
<td>Yes (4 projects)</td>
<td>Yes</td>
<td>63.9 % (2008)</td>
<td>2501.8 (2012)</td>
<td>+++</td>
</tr>
<tr>
<td></td>
<td>Weser-EMS (DE 94)</td>
<td>Yes (1 projects)</td>
<td>Yes</td>
<td>n.a.</td>
<td>n.a.</td>
<td>+++</td>
</tr>
<tr>
<td></td>
<td>Wielkopolskie (Greater Poland) (PL41)</td>
<td>Yes (3 projects)</td>
<td>Yes</td>
<td>58.8 % (2009)</td>
<td>1779 (2012)</td>
<td>++</td>
</tr>
</tbody>
</table>

73 Rural Development Program
Legend:

“+” to “++++”: increasing level of saturation, emissions, load

0: no remarkable saturation, emissions or load

n.a.: not available

↑ to ↑: at national level, the N surplus had increased from 1990 to 2005

↓ to ↓: at national level, the N surplus had decreased from 1990 to 2005


* The rating is attributed according to the maps’ scale so that the maps colour code and related values comparatively correspond to the five possible scores (from 0 to ++++). A score is attributed for each territory unit and represents the average saturation, emissions or load level of the territory that is calculated by the authors. However, note that it does not imply a similar saturation, emissions or load level at smaller scale which can vary within the territory. For instance, a NUTS 2 region showing a medium phosphorus load (0.5 mg P/L > x > 1 mg P/L) does not imply that all water bodies within the territory present the same P load.
<table>
<thead>
<tr>
<th>Saturated area</th>
<th>Proposed reference NUTS 2 region</th>
<th>Comments for the choice of the proposed reference NUTS 2 region within the proposed Member State</th>
</tr>
</thead>
<tbody>
<tr>
<td>West of France</td>
<td>Brittany (B52)</td>
<td>Brittany is the most saturated region for both N and P in France and one of the most saturated in the EU. It is also affected by the highest NH₃ emissions and significant N₂O and NOₓ emissions. The risk of leaching and run-off is moderate for nitrogen, leading to moderate concentration of nitrogen in surface water but low phosphorus load. The farming system is intensive with the highest French livestock density and amount of manure applied. The soil is mostly made of silt, with low SOM. The region is also highly affected by erosion. Brittany was chosen because of its high surplus for nitrogen and phosphorus with moderate pollution risk with in particular, high NH₃ emissions and a high livestock density.</td>
</tr>
<tr>
<td>Po Valley, Italy</td>
<td>Lombardy (ITC4)</td>
<td>Po Valley is the most saturated area for N and P in Italy. It corresponds to several NUTS2 regions with a large surplus area located in Lombardy. Lombardy shows moderate level of emissions and in particular high NH₃ emissions. It also presents moderate nitrogen leaching although low nitrogen concentration is observed in surface or groundwater. The region shows moderate phosphorus load in river. Lombardy is also a region of intensive agriculture with the lowest average size of holdings compared to other saturated regions. It has a high livestock density and applies a large quantity of phosphorus fertilisers. The soil in Lombardy is mostly sandy with a low SOM content and high bulk density. Lombardy was selected for its high surplus for N and P. Lombardy is also interesting for its high livestock density and the low average size of its holdings.</td>
</tr>
<tr>
<td>West Denmark</td>
<td>Midtjylland (Mid-Jutland) (DK04)</td>
<td>West Denmark is one of the most saturated areas in Denmark and shows a moderate surplus in P. The region is very intensive with one of the highest average size of holdings. It has a moderate livestock density but it has increased by 10.5 % between 2005 and 2010. This region includes very sandy soils in the western part and has high SOM content. The climate in Denmark is the coldest of the selected regions. In addition to its high surplus, especially for N, Denmark Mid-Jutland was selected especially for its environmental conditions i.e. its sandy soil with high SOM content and cold weather. Although Spain is not among the most saturated countries, Murcia is one of the regions that show the highest nutrient surplus in Spain. It also shows moderate NH₃ emissions, moderate nitrogen concentration in both groundwater and surface water but a very low phosphorus concentration in water. Murcia is a region with high livestock density, with one of the highest levels of phosphorus manure application, one of the highest shares of horticulture in the EU-28 and one of the highest shares of organic farming. Moreover, Murcia is a region under severe water stress with high mean temperature and very little precipitation. The soil structure is very dense, with one of the finest texture, highest bulky structure and a very low SOM content. Although Murcia is less saturated in nutrients than Catalonia, Murcia was chosen considering the high share of horticulture production, its environmental conditions, i.e. its Mediterranean weather with fine and bulky soil poor in SOM, and its high phosphorus manure application while the region shows one of the lowest P load in water in the EU-28.</td>
</tr>
<tr>
<td>South of Spain</td>
<td>Murcia (ES62)</td>
<td></td>
</tr>
<tr>
<td>Saturated area</td>
<td>Proposed reference NUTS 2 region</td>
<td>Comments for the choice of the proposed reference NUTS 2 region within the proposed Member State</td>
</tr>
<tr>
<td>----------------</td>
<td>-----------------------------------</td>
<td>--------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Southern Netherlands</td>
<td>North-Brabant (NL41)</td>
<td>The North-Brabant region is the most saturated region for both nitrogen and phosphorus in the EU-28. The nitrogen surplus is 419 kg N/ha (Leip, et al., 2013). It is also the area in the Netherlands with the highest pollution risks, in particular regarding phosphorus loads. The farming system is intensive with medium average size of holdings. The region has the highest European livestock density: 7.5 LSU/ha UAA in 2010, a figure that has increased of 11.5 % since 2005. The region applies important quantities of manure on agricultural soils. The SOM content is high (as in the whole Member State) and the region is considered to be severely water-stressed. North-Brabant was selected for its very high surplus in both N and P and the associated emissions to air and concentration in water. The North-Brabant is also interesting for its very high livestock density, the high SOM content and its water stress.</td>
</tr>
<tr>
<td>South-east Ireland</td>
<td>Southern and eastern (IE02)</td>
<td>The region is the most saturated for both N and P in Ireland. It is subject to high air emissions, in particular for N₂O. It is also very affected by nutrient leaching and run-off but show low concentrations of phosphorus and nitrogen in water. Agriculture is moderately intensive. It has a medium livestock density and one of the lowest shares of horticulture production among the saturated regions. The soil has a high density and significant SOM content but much lower SOM than the western part of Ireland.</td>
</tr>
<tr>
<td>North-west Germany</td>
<td>Weser-Ems (DE 94)</td>
<td>Weser-Ems is one of the most saturated regions of north-west Germany for N and P with moderate pollution risks from nitrogen, especially regarding emissions to air. However, a low phosphorus load is observed in rivers. The farming system is intensive and the size of holdings is close to European average. The region applies a large quantity of phosphorus manure. The region presents medium water stress and is affected by wind erosion. Weser-Ems was selected for its high surplus in both N and P, its characteristics close to the average EU farming system, and the low phosphorus load in water despite significant manure phosphorus application and phosphorus surplus.</td>
</tr>
<tr>
<td>Poland</td>
<td>Wielkopolskie (Greater Poland) (PL41)</td>
<td>Although Poland is not among the most saturated countries, two regions, Wielkopolskie and Kujawsko-Pomorskie, present low surplus. The nutrient surplus of these two regions is subject to uncertainties: the surplus is not clearly visible on the maps, and Leip et al. shows a nitrogen budget of 58 and 52 kg N/ha for Wielkopolskie and Kujawsko-Pomorsie respectively (Leip, et al., 2013). However, another source shows that the nitrogen surplus is 90 and 84 kg N/ha respectively (Pastuszak &amp; Igras, 2012) which is similar to the average surplus of Denmark or the UK (Van Grinsven, et al., 2012). The surplus of phosphorus is similar to the surplus observed in Denmark. The level of emissions of NH₃ and NOx, the N concentration in water and the P load are moderate. The farming system is moderately intensive with one of the lowest average sizes of holdings. It has moderate livestock density and horticulture production. The region has sandy soils with medium organic matter content and medium water stress. Poland was selected due to its importance in terms of overall nutrient loads to the Baltic Sea despite the non-intensive (on average) agricultural system. Wielkopolskie was chosen because it shows the highest surplus in N and P in Poland.</td>
</tr>
</tbody>
</table>
4.3 Nutrient scarce regions in EU-28

4.3.1 Analysis of the available maps on nutrient budget

According to the Figure 23, Figure 24 and Figure 25 for nitrogen budgets and Figure 26 for phosphorus budgets, the regions with the lowest surplus are identified in the Table 19.

Table 19 – Areas with low or null surplus in nutrients in the different maps

<table>
<thead>
<tr>
<th>Saturated areas</th>
<th>Level of N surplus</th>
<th>Level of P surplus</th>
</tr>
</thead>
<tbody>
<tr>
<td>The United Kingdom</td>
<td></td>
<td></td>
</tr>
<tr>
<td>North of Scotland</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>South-east</td>
<td>/</td>
<td>/</td>
</tr>
<tr>
<td>Sweden</td>
<td></td>
<td></td>
</tr>
<tr>
<td>North</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Centre</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Finland</td>
<td></td>
<td></td>
</tr>
<tr>
<td>North</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Centre</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Spain</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Andalucia</td>
<td>· · · /</td>
<td>/</td>
</tr>
<tr>
<td>Jaen74</td>
<td>· · · ·</td>
<td>/</td>
</tr>
<tr>
<td>Galicia</td>
<td>· · · /</td>
<td>· · · /</td>
</tr>
<tr>
<td>France</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Provence Alpes Côte d’Azur (PACA)</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Languedoc Roussillon</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Ile de France</td>
<td>/</td>
<td>/</td>
</tr>
<tr>
<td>Corsica</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Italy</td>
<td></td>
<td></td>
</tr>
<tr>
<td>South</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Italy</td>
<td></td>
<td></td>
</tr>
<tr>
<td>North-east</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Greece</td>
<td></td>
<td></td>
</tr>
<tr>
<td>South</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Portugal</td>
<td></td>
<td></td>
</tr>
<tr>
<td>South</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>North-west</td>
<td>· · · ·</td>
<td>· · · ·</td>
</tr>
<tr>
<td>Romania</td>
<td>Whole country</td>
<td>n.a.</td>
</tr>
<tr>
<td>Bulgaria</td>
<td>Whole country</td>
<td>n.a.</td>
</tr>
<tr>
<td>Estonia</td>
<td>Whole country</td>
<td>n.a.</td>
</tr>
<tr>
<td>Latvia</td>
<td>Whole country</td>
<td>n.a.</td>
</tr>
<tr>
<td>Lithuania</td>
<td>South</td>
<td>n.a.</td>
</tr>
<tr>
<td>Portugal</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Centre</td>
<td>/</td>
<td>/</td>
</tr>
<tr>
<td>East</td>
<td>/</td>
<td>/</td>
</tr>
</tbody>
</table>

Legend: “-” to “- - - -”: decreasing saturation, from low surplus to negative surplus (the darker the box is, the lower the budget is)
“/”: area where surplus cannot be considered as low, null or negative
“n.a.”: information not available (countries not included in the study)

For nitrogen, the regions that have the lowest surplus (possibly reaching negative budgets) are northern Scotland in the United Kingdom; north and central Sweden; north and central Finland; Provence-Alpes-Côte d’Azur, Languedoc Roussillon, and Corsica in France; all of Romania; and south Greece. For

74 Part of Andalucia
phosphorus, the lowest surpluses are observed in northern Scotland and south-east of the United Kingdom; north of Sweden; north of Finland; Ile-de-France and Corsica in France; and central and east Germany. It is interesting to notice that some regions that do not show deficits in nitrogen nevertheless have very low phosphorus budgets.

### 4.3.2 Analysing nutrient scarcity

Regions that have very low or negative budgets are not necessarily nutrient scarce regions. Scarcity also depends on the initial soil nutrient content or concentration. Indeed a soil with a high content of N or P will not be subject to scarcity with a balanced budget, if the soil content stays high. On the contrary, a balanced budget or negative budget may be an issue if the nutrient soil content is already low. Therefore, in order to prevent nutrient scarcity, a positive budget may be recommended, although not beyond the nutrient requirements of the crops. For instance, a source recommends a surplus of nitrogen of 30 kg N/ha/year and a phosphorus and potassium surplus close to zero in the United Kingdom for the whole territory (Soil association (UK), 2013). This amount is a compromise between the optimisation of yield for the crop, the maintenance of fertility, in particular the amount of nutrients available for the next crop, and the estimated pollution risk. This target is lower than for nitrogen due to high reserves of P and K in the soil of this country. The high reserve may be due to past excessive fertilisation (Toth, et al., 2014) or to the nature of the minerals in the parent material from which the soils have formed. For phosphorus, P levels follow the main climatic patterns in Europe, with the highest levels in areas of Atlantic north-western Europe and the lowest levels in Mediterranean cropland soils (Toth, et al., 2014). For potassium, some clay soils, such as those of several regions in the United Kingdom, contain a large quantity of potassium. The latter is released through the weathering of soil to become available for plants. If the release is higher than crop uptake, it can lead to increasing soil potassium content and saturation (DEFRA, 2010).

The regions identified as low budget regions are compared to data on nutrient content to identify scarce regions. Regarding nitrogen, regions that have very low nitrogen content are the south-east of the United Kingdom, Spain (except north and north-west), south of Portugal, Italy (Po Valley and South), east PACA, Languedoc Roussillon, north and Ile de France in France, east of Germany, Poland (except the north-east), centre of Czech Republic, the whole of Hungary, and north of Greece (Figure 39). For phosphorus, the regions that have very low phosphorus concentration are Spain (in particular Galicia), Italy (in particular south), Romania (especially centre and East), south of France, centre of Sweden, Lithuania, Latvia, south of Hungary, Bulgaria, south of Austria, Slovenia and Greece (except West) (Figure 40). The figures proposed by Toth et al (2014) are average value from NUTS 2 level to Member States.

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75 The figures proposed by Toth et al (2014) are average values from NUTS 2 level to Member State level. For some countries such as Lithuania or Latvia, only average values at national level are proposed. This does not allow to precisely identify the area with the lowest concentration within the country.
The analysis of this map confirms that there is no direct link between low or negative nutrient surplus and nutrient deficiency. Therefore, defining nutrient scarce regions requires considering first the nutrient deficiency state of the regions (i.e. low nutrient content of the area) and then the low or negative surplus of nutrient to identify the evolution of this deficiency. Indeed, for land that shows poor nutrient content or concentration, low or negative surplus will results in maintaining or even increasing the nutrient deficiency, inducing soil fertility issues.

Hence, regions that can be considered as nutrient scarce with potential fertility issues (i.e. regions that show both low nutrient content and low or negative surplus) are for both nitrogen and phosphorus: France (Languedoc-Roussillon and PACA), Spain (Andalusia) and south Italy. Central Sweden can also be added for phosphorus, as well as the centre of Italy (Umbria), based on the mean P fertiliser need describes by Toth et al. (2014). Considering the lack of data on phosphorus surplus, the case of Romania and Bulgaria which present low phosphorus content is uncertain (Toth, et al., 2014).

Regarding potassium, a map of K soil content is available (see Annex 12) (Toth, et al., 2013), but the analysis of the potassium concentration as such is not enough to determine scarce regions at EU level. However, a decrease in K fertiliser application has been observed in many European countries and negative potassium budgets have been reported at farm and field level, especially in low inputs systems (Oborn, et al., 2009). Hence, a particular attention in potassium aspects should be paid when addressing nutrient scarcity.
5. Case studies

In the previous chapter, eight NUTS2 EU regions were selected. In the present chapter, a case-study approach is used to examine, for each region, the impacts of nutrient surplus on the environmental compartments, its underlying causes in terms of farming systems and practices, as well as the costs of these impacts. For each case study, the last section aims at identifying the possible good practices that could be implemented at farm level considering local conditions to reduce the nutrient losses linked with the agricultural activities conducted in saturated areas. This latter section is built on the solutions reviewed and evaluated in the chapter 3. The measures were selected based on their impacts on the agro-ecosystem in terms of reduced losses through improved nutrient utilisation. Emphasis was placed on measures that have not yet been exploited to their full potential within the regions. Further selection criteria were whether the measure might be feasibly implemented and whether the measure offers benefits which balance (or outweigh) the costs. Each case study was commented by regional and/or national experts (public institutions, NGOs, farmers associations, researchers).
5.1 Brittany (FR)\textsuperscript{76}

Box 4 – Brittany case study - In brief

Brittany has the highest nitrogen (N) and phosphorus (P) surplus in France and is one of the regions with the highest surplus in the EU-28. In the region, the most notable consequence of the nitrogen excess is the eutrophication of marine waters. During the 2004-2013 period, 53 beach sites and 32 estuaries were on average affected with at least one peak of algae bloom during the year. Such blooms negatively impact biodiversity, health and economic activities such as tourism, shellfish farming and real estate. In Brittany, several cases of wild animal mortality were reported. It is highly probable that they were exposed to hydrogen sulphide (H\textsubscript{2}S) emissions caused by the degradation of algae. In addition, excessive nitrogen load is observed in freshwater. In 2014, 6 % of the stations monitoring the water quality in Breton rivers, recorded concentrations above 50 mg NO\textsubscript{3} -/L. However, important variations in nitrate concentration are observed across watersheds. Eutrophication of freshwater (mainly due to phosphorus) also occurs locally in Brittany. This has consequences on drinking water quality and causes damages to the efficiency of water pipe systems in water abstraction stations. In addition, in eutrophic freshwater bodies, cyanobacteria can develop in warm weather months and release the toxin microcystin that may cause adverse health effects. In 2012, water activities such as swimming and fishing were prohibited for more than three weeks in 9 sites exposed to cyanobacteria blooms in Brittany.

In the region, the observed impacts are mostly due to the uneven distribution of intensive breeding activities, resulting in local excess of manure production compared to 1) the legal amount of manure that can be applied according to the Nitrates Directive and 2) the local crop needs. Yet organic and chemical fertiliser application is high, in particular for maize that is highly tolerant to excessive fertilisation. Considering that maize production (for grain and forage) represents 31 % of the regional arable land, the over-fertilisation of maize field has a significant impact on the regional nutrient losses and related impacts. The high amount of manure produced is often concentrated in small areas and must be stored, with additional risk of nutrient losses in case of inappropriate storage. The losses of nutrients are enhanced by natural factors such as abundant rainfalls that increase leaching risks, silt soils prone to compaction and slacking crust that increase N\textsubscript{2}O emissions and increase run-off.

Costs are incurred from the losses of nutrients in the region, in particular because of the green algae issue. The cost of the removal of green algae on beach areas was estimated at € 1 400 000 for 2012 and the economic loss for touristic accommodation activities in the Côtes-d'Armor (a part of Brittany) in 2011 was estimated at € 800 000. In France, the revenue losses due to eutrophication are estimated between 70 and 100 million euros a year for costal municipalities. The cost of the additional water treatment for making the water drinkable due to the excess of nitrates is estimated to € 120 to € 360 million per year, with a cost of nitrogen removal that is estimated to € 70 /kg N. In France, the theoretical clean-up cost of surface and coastal waters regarding the nitrates losses from agriculture is estimated at over € 49 billion per year while the theoretical decontamination treatment of groundwater (nitrates) would cost more than € 490 billion. Primarily in response to the green algae damages, the French government has carried out a 2010-2015 national plan to tackle the issue of the proliferation of green algae. The total budget for the plan is € 134 million over 5 years, with Brittany representing about 85 % of the area covered by the plan.

In Brittany, the first set of measures identified in this report aims to reduce the local source of pollution by controlling the geographic distribution of livestock, adapting the feeding strategies, processing manure to decrease its nutrient content, improving the fertilisation management plans for N and P and converting arable land to unfertilised grassland in areas at risk. The second set of measures that has been selected focuses on the reduction of nutrient losses during manure storage and housing, either by covering slurry tanks to prevent ammonia emissions and/or install “V” scrapers in pig buildings. A third set of measures has been identified to prevent nutrient losses when manure is applied in field or in pasture by optimizing grazing intensity trough rotational grazing, considering climate conditions before spreading and implementing grass strips along hedgerows. Finally, the last set of measures aims to improve soil quality and decrease the amount of fertiliser that farmers need to buy by processing manure to ease its use and transfer, using catch crops and preferring conservation tillage techniques.

\textsuperscript{76} Authors: Ms. Marion Sarteel, BIO by Deloitte (lead author); Mr. Clement Tostivint, BIO by Deloitte; Ms. Claire Basset, BIO by Deloitte; Mr. Giorgio Provolo, Milan University; Mrs. Helen Ding, BIO by Deloitte.
Brittany is a region located in the north-west of France (see Figure 41) with an oceanic climate. The region is criss-crossed by a dense network (1 km/km$^2$) of natural and man-made waterways. There are 560 small watersheds that discharge to the sea and five large watersheds accounting for over 55% of the regional area. The Vilaine River is the largest in Brittany and its catchment area represents on its own 33% of the territory (Novince & Aurousseau, 2014).

Brittany is a leading agricultural region in France and Europe. In 2012, it accounted for 12% of the national agricultural production income, with only 6% of the national surface area (Chambre d’Agriculture de Bretagne, 2014; INSEE, 2013a). In 2013, about 1.73 million hectares were dedicated to agriculture, representing 63% of the total regional area (Eurostat, 2014a). Arable land represented 87% of the UAA (Eurostat, 2014a). Breeding is the main agricultural activity with 70% of the holdings in 2010 (Préfecture de Bretagne, 2013). The main livestock in the regions are pigs, poultry and dairy cows (DRAAF Bretagne, 2013a; Chambre d’Agriculture de Bretagne, 2014) (see Figure 42). In 2013, Brittany counted 7.6 million pigs, 2.0 million beef and dairy cattle and 89.7 million poultry (DRAAF Bretagne, 2014). Pig and poultry production represented 58% and 33% of the national production respectively (in tonne) (Chambre d’Agriculture de Bretagne, 2014) and 20% of the dairy cows were located in the region (Eurostat, 2014a; Chambre d’Agriculture de Bretagne, 2014).

The main crops produced were tender wheat (1 676 kt), maize (583 kt), barley (320 kt) and triticale (247 kt) in 2013 (Chambre d’Agriculture de Bretagne, 2014), mainly grown for feeding livestock either on-farm or off-farm. Brittany is also the leading region for vegetable production, representing 23% of the national surface area in 2013 (Chambre d’Agriculture de Bretagne, 2014). In particular, cauliflower and artichoke production represents 85% and 83% of the national production respectively (in tonnes) in 2013 (DRAAF Bretagne, 2013a).

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Figure 41 – Geographical location of Brittany

Brittany is a leading agricultural region in France and Europe. In 2012, it accounted for 12% of the national agricultural production income, with only 6% of the national surface area (Chambre d’Agriculture de Bretagne, 2014; INSEE, 2013a). In 2013, about 1.73 million hectares were dedicated to agriculture, representing 63% of the total regional area (Eurostat, 2014a). Arable land represented 87% of the UAA (Eurostat, 2014a). Breeding is the main agricultural activity with 70% of the holdings in 2010 (Préfecture de Bretagne, 2013). The main livestock in the regions are pigs, poultry and dairy cows (DRAAF Bretagne, 2013a; Chambre d’Agriculture de Bretagne, 2014) (see Figure 42). In 2013, Brittany counted 7.6 million pigs, 2.0 million beef and dairy cattle and 89.7 million poultry (DRAAF Bretagne, 2014). Pig and poultry production represented 58% and 33% of the national production respectively (in tonne) (Chambre d’Agriculture de Bretagne, 2014) and 20% of the dairy cows were located in the region (Eurostat, 2014a; Chambre d’Agriculture de Bretagne, 2014).

The main crops produced were tender wheat (1 676 kt), maize (583 kt), barley (320 kt) and triticale (247 kt) in 2013 (Chambre d’Agriculture de Bretagne, 2014), mainly grown for feeding livestock either on-farm or off-farm. Brittany is also the leading region for vegetable production, representing 23% of the national surface area in 2013 (Chambre d’Agriculture de Bretagne, 2014). In particular, cauliflower and artichoke production represents 85% and 83% of the national production respectively (in tonnes) in 2013 (DRAAF Bretagne, 2013a).

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DRAAF Bretagne: Regional Department of Food, Agriculture and Forest of Brittany
5.1.1 Notable impacts of nutrient surplus

This section focuses on the impacts of nutrient excess that are specifically related to agricultural practices in Brittany. In this region, the main impacts are caused by nitrogen and phosphorus surpluses that mostly affect water and air quality. Potassium excess is not a major issue in this region.

Although the nutrient surplus had drastically decreased over the last decade (Agreste Bretagne, 2013), Brittany is still the most saturated region for both N and P in France and one of the most saturated in the EU-28 with high livestock density and amount of manure applied. In 2013, the surplus of nutrients in Brittany was 48 141 tons of N (average of 29 kg N/ha UUA) and 18 190 tons of P (11 kg P/ha UAA) (DRAAF Bretagne, 2013a). The distribution of the nitrogen surplus is showed in Figure 43 for 2011. Figure 44 shows the distribution of phosphorus surplus in the region the same year.
Impacts of nitrogen losses

In Brittany, nitrogen surplus in agricultural soils leads to an excess of nitrogen in groundwater and surface waters (freshwater and marine water) and high nitrogen emissions to air.

Eutrophication of marine waters

In Brittany, the most notable consequence of the excess nitrogen is the eutrophication of marine waters. Since the 70s, high nitrate concentration in marine water has resulted in green algae blooms on the northern coast of Brittany, and the situation has worsened in the last decade.

Currently, the two most severe green algae bloom spots in the region are the Saint-Brieuc bay and the Lannion Bay, near the town of Saint-Michel-en-Grève (see Figure 45). During the period 2004-2013, 53 beach sites and 32 estuaries were affected on average with at least one peak of algae bloom during the year (CEVA, 2013)\textsuperscript{78}. Moreover, in 2012, over 74 000 m\textsuperscript{3} of green algae, mostly *Ulva armoricana*, were collected on the beaches (70 000 tonnes).

\textsuperscript{78} CEVA: Centre of algae study and valorisation
However, the algae on beaches are only the most visible part of the biomass produced by eutrophication. Indeed, the production of biomass is estimated to 200 000 tonnes per year with a visible part unequally distributed along the coasts. While on Côtes-d'Armor beaches, almost all of the biomass produced reaches the shores, 95% of algae remain at sea in south Finistère (Peyraud, et al., 2012). The share of algae visible in beaches does not depend on the amount of nitrogen in water locally but on the natural containment of the water in certain bays.

These algae blooms have severe consequences on the environment, biodiversity and health. Macroalgae and phytoplankton proliferation may decrease the oxygen rate and reduce water clarity. Consequently, several species may not be able to develop due to the lack of oxygen or to reproduce due to the reduced water clarity. The lack of clarity and the disappearance of some marine species can affect birds that have trouble accessing food. Note, however, that certain birds such as the Brent goose ("Bernache Cravant") are able to feed on these algae.

The degradation of algae by bacteria combined with the anoxic conditions can enhance the release of ammonia and of hydrogen sulphide (H₂S). The latter is a toxic gas that causes instant death if inhaled at high concentrations (above 800 ppm) (Ministère de l'écologie, du developpement durable et de l'énergie, 2010; Oral, 2013). In Brittany, several cases of wild animal mortality were reported (see section 2.3.6). In particular in the summer of 2011, the bodies of several wild animals (thirty-six wild boars, three coypus and one badger) were discovered on the beach near Morieux and on the banks of the estuary of the Gouessant River (Côtes-d'Armor). In view of the available data, the French Agency for Food, Environmental and Occupational Health & Safety (ANSES) concluded that it is highly probable that the animals have been exposed to H₂S concentrations leading to the lesions and their death (ANSES, 2011).

In addition to these issues, algae blooms also impact economic sectors such as tourism, shellfish farming and real estate.
**Excessive nitrogen load in freshwater**

The average nitrate concentration in surface water was 33.4 mg NO$_3$-L$^{-1}$ in 2014 and 32.9 mg NO$_3$-L$^{-1}$ in groundwater in 2012 (Observatoire de l'eau en Bretagne, 2015d; Observatoire de l'eau en Bretagne, 2015b). Important variations in nitrate concentration are observed across watersheds. In 2014, 6% of the stations monitoring the water quality in Breton rivers, recorded concentrations above 50 mg NO$_3$-L$^{-1}$ (“bad” Nitrate status) and 70% of them recorded concentrations between 25 and 50 mg NO$_3$-L$^{-1}$ (“poor” Nitrate status) (Observatoire de l'eau en Bretagne, 2015b). The highest concentration of nitrogen in freshwater for both surface and groundwater is localised on the northern coast of Brittany in Finistère and Côtes-d’Armor (MEDDE, 2012; Observatoire de l'eau en Bretagne, 2015b; Observatoire de l'eau en Bretagne, 2015d). Moreover, as the nitrogen load is transferred to the sea, it contributes significantly to marine eutrophication. Simulations have shown that nitrate concentration in rivers should be lowered to 10-15 mg NO$_3$-L$^{-1}$ to significantly decrease the algae bloom in coasts (Peyraud, et al., 2012).

High nitrogen concentrations in groundwater and surface water require treatment of drinking water in order to avoid risks for human health. This is all the more important given that 80% of the drinkable water in Brittany is surface water (Novince, 2013; DRAAF Bretagne, 2013a). Indeed, while 10% of the population was exposed to drinking water with nitrate concentration above the legal threshold in 2000, this figure has decreased to 0.4% in 2011 with only six drinking water supply units out of the 731 units in the region having trouble meeting the nitrate concentration standards for drinking water (Bretagne Environnement, 2013c).

**Ammonia (NH$_3$) pollution in the air**

Agriculture is responsible for over 90% of NH$_3$ release in Brittany (see Figure 46). Within the agriculture sector, animal breeding represented 77% of NH$_3$ emissions in 2010. In 2004, 148,000 tons of NH$_3$ were released in the atmosphere, accounting for 18.8% of the national emissions (Bretagne Environnement, 2006). The share of the three main breeding activities in Brittany for ammonia emissions can be roughly estimated by using the number of animals (Agreste, 2011) and the ammonia emission quantities per animal per year found in the CORPEN study (CORPEN, 2006). Pig breeding is the primary source of NH$_3$ emissions in Brittany (41%), followed by cattle breeding (28%) and finally, poultry breeding (20%).

![Source: (Bretagne Environnement, 2006)](Figure 46 – Share of agriculture in NH$_3$ and N$_2$O emissions in Brittany in 2004)

NH$_3$ pollution in air has indirect impacts on environment and health. Indeed, NH$_3$ in air mainly results in acidic rainfalls which contribute to soil and water acidification. Others molecules such as SO$_2$ also play

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79 Data from the monitoring and surveillance network FRGSUSR

80 Observatoire de l'eau en Bretagne: Observatory of water in Brittany
a role in acidification but agriculture is only a minor contributor of these emissions. While little attention is paid to this issue, acidification may contribute to the dissolution and leaching of nitrogen.

Moreover, in Brittany, NH₃ is the main source of secondary particles. Ammonia reacts with nitrogen or sulphur oxides to form fine particles of ammonium sulphates or ammonium nitrates that may have severe consequences on health (respiratory and cardiovascular diseases) (AirBreizh, 2015). In 2004, 14 500 tons of PM10 were emitted to the air by agricultural activities in Brittany. Agriculture and forestry account for about 45 % of the total of PM10 emissions in Brittany (about 6 % of national emissions) (Bretagne Environnement, 2006). In 2010, each of the six measurement stations reported 1 to 30 cases where the threshold was breached (50 μg/m³ not to be exceeded on more than 35 days of a calendar year) for PM10 particles (MEDDTL, 2011).

**Nitrous oxide (N₂O) pollution in the air**

In Brittany, the main source of nitrous oxide in the air is agriculture (>95 %) (see Figure 46). In 2004, 18 000 tons of N₂O were released to the atmosphere because of agriculture. Brittany accounts for 7 % of the national N₂O emissions. N₂O highly contributes to global warming, but does not lead to any direct impact at local level (CORPEN, 2006). N₂O accounts for 37 % of the GHG emissions in Brittany (Bretagne Environnement, 2006).

**Impacts of phosphorus losses**

In 2014, the average concentration of total phosphorus in Britain watercourses was very high with 0.18 kg P/L (Observatoire de l'eau en Bretagne, 2015c) compared to the European average (0.026 kg P/L (EEA, 2012c). However, a high concentration of total phosphorus does not necessarily imply high impacts on the environment. Indeed, only the dissolved phosphorus can be assimilated by organisms and possibly lead to impacts. Consequently, the concentration of orthophosphate has consequences on the environment in the short-term while phosphorus bound with sediments that may release phosphorous ions progressively, may have longer term environmental consequences.

On average, the concentration of orthophosphate in freshwater is low with 0.29 mg PO₄³-/L in 2014 (eq. 0.065 mg P/L) (Observatoire de l'eau en Bretagne, 2015c), similar to the average European average orthophosphate concentration (0.07 mg P/L in rivers in 2010 (EEA, 2012c)). Considering phosphorous matter, 70 % of the surface water monitored was considered as good quality water since 2000 (Observatoire de l'eau en Bretagne, 2015c). The water quality is better in western Brittany than in eastern Brittany. In the Ille-et-Vilaine department (eastern Brittany), only a little more than 50 % of the monitoring catchment show good water quality (Observatoire de l'eau en Bretagne, 2013b). Also note that the concentration in orthophosphate is increasing since 2007, with a slight decrease in 2014 (Observatoire de l'eau en Bretagne, 2015c). It is estimated that 60 % of the phosphorus in freshwater comes from agriculture. Urban wastewater and industrial wastewater represent 30 % and 10 % of the load respectively (Secrétariat technique du bassin Loire-Bretagne, 2010).

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81 Particulate Matter 10 micrometers or less in diameter

82 It is important to note that agriculture is also the main source of methane (CH₄) emissions in Brittany but this specific impact of agriculture is not in the scope of this report.

83 Considering that N₂O, CH₄ and CO₂ are the three main GHG and converting the emission estimations into CO₂ equivalent (x298 for N₂O and x25 for CH₄).
Phosphorus is considered as the limiting nutrient to eutrophication in freshwater compared to N. Eutrophication of freshwater occurs locally in Brittany, in particular in the east of the region (Bretagne Environnement, 2013a). When eutrophication occurs in water abstraction stations, it has heavy consequences on drinking water quality and causes important damages to the efficiency of water pipe systems and increased costs for water treatment. For instance, in 2005, 24 % of the annual drinking water abstraction stations for the Loire Bretagne basin required extra maintenance because of algae blooms in the water pipes (CGDD, 2011).

In eutrophic freshwater bodies, cyanobacteria (often called blue-green algae) can develop in warm weather months and release microcystin toxin (Eau et rivières de Bretagne, 2004) that may cause adverse health effects such as skin irritations when in contact with water or even nausea or diarrhoea in case of ingestion of contaminated water. In 2012, swimming, fishing and other water activities were prohibited for more than three weeks in 9 sites exposed to cyanobacteria blooms in Brittany. As regards drinking water, 3.8 % of the samples of untreated water (over 324) showed a microcystin population above 1 micro g/L (ARS Bretagne, 2012).

Lastly, phosphorus transported by rivers is released in sea waters and may also contribute to the marine green algae phenomenon although in a lesser extent than N which is the limiting factor in sea waters.

5.1.2 Causes of nutrient losses

Farming system and agricultural practices

Farmers have made significant efforts to successfully reduce the nutrient losses in the region. However, the nitrogen and phosphorous surpluses are still high (Agreste Bretagne, 2013). The main causes for nutrient surplus in Brittany are the following:

Uneven distribution of breeding activities throughout the territory – Every year, 10 million tons of manure are produced in Brittany (FranceAgrimer, 2012). The intensification of agriculture has induced an increase of breeding activity and livestock density (an average of over 2.8 units/ha UAA in 2010 (Eurostat, 2013a), resulting in higher amounts of manure produced. For nitrogen, the total regional nitrogen needs were higher than the amount of nitrogen in the manure produced in the region in 2013 (DRAAF Bretagne, 2014). However, since the breeding activities are unevenly spread within the territory, in some river basins the quantity of manure produced is much higher than the amount of nitrogen that can be applied on field in terms of nitrogen budget or according to the threshold established by the regional Nitrates Action Program (170 kg N/ha) (DREAL Bretagne, 2014). For phosphorus, the total amount of P from the manure produced at the regional level exceeded the amount of phosphorus that was assimilated by crops in 2013 (DRAAF Bretagne, 2014).

In Brittany, the main source of nitrogen surplus from livestock is dairy and in a lesser extent poultry and pigs with a total cumulated surplus of respectively 18 000, 7 000 and 6 500 t N in the region in 2001 (Agreste Bretagne, 2013).

84 CGDD: Commissariat général au développement durable, Commissioner-General for Sustainable Development
85 Legal threshold above which treated water cannot be considered as drinking water.
86 DRAAF Bretagne: Regional Department of Food, Agriculture and Forest
87 DREAL Bretagne: Regional Directorate for Environmental, Planning and Housing of Brittany
Excessive and not evenly distributed amounts of nutrients applied on pastures – Grassland represented 38% of the regional UAA in 2013, including 77% of temporary pastures (Chambre d’Agriculture de Bretagne, 2014). They are fertilised by the grazing animals and the organic and mineral fertilisers applied by the farmers. In 2006, organic fertilisers accounted for 70% of the nutrients provided in pasture, among which two thirds were by grazing animals (Agreste Bretagne, 2008). In pasture; the average amount of nutrients provided exceeded the plants’ needs by 64 kg N/ha in 2006. This is due to the difficult management of nutrients from grazing animals. Since manure from grazing animals is concentrated on about 10% of the surface area, farmers must provide additional inputs, mostly through chemical fertilisers (Agreste Bretagne, 2008). This leads to local excess of nutrients and higher risks of nutrient losses. Although permanent pasture are less fertilised, the surplus is in general higher than for temporary pasture due to a lower output by plants that are more grazed (Agreste Bretagne, 2008).

Excessive P manure application – The main source of organic phosphorus is intensive poultry and pig breeding (SATEGE, 2013). Since manure is often spread locally88, in the farm or in the neighbours’ fields, the high manure production leads to phosphorus surplus in the river basins where the breeding activity is concentrated (ACTA, 2010). In areas concerned by the River Basin Management Plans (RBMPs, in French: SDAGE) the amount of phosphorus that can be applied is limited for large livestock farms considered as Classified Installation for Environmental Protection (ICPE) under the implementation of the Integrated Pollution Prevention and Control (IPPC) Directive89. Outside these areas, considering the high amount of available P, manure can possibly be applied in excess compared to crops’ needs (provided that the N:P ratio is appropriate to nutrient plant assimilation). In Brittany, 70% of the cantons showed an excess of phosphorus in soils in 2000-2004 (Bretagne Environnement, 2011).

Large area cultivated with maize crops that are highly tolerant to excessive fertilisation – In 2006, the manure produced was sufficient to cover the needs in nitrogen of maize at a level of 161 kg N/ha (Agreste Bretagne, 2008). However, the average amount of N provided to maize parcels in Brittany was 209 kg N/ha in 2006, including 178 kg N/ha of organic fertiliser and 31 kg N/ha of inorganic fertiliser (Agreste Bretagne, 2008). The excessive input of fertilisers is partly explained by the high tolerance of maize to excessive fertilisation (Barbut & Poux, 2000; Agreste Bretagne, 2008) and the lack of accuracy of fertilisation management plans. For maize production, it is considered that only 25% of the nitrogen contained in farmyard manure and 70% of the nitrogen contained in pig slurry is available the first year (Agreste Bretagne, 2008). Hence, some farmers prefer using mineral fertilisers than organic fertilisers or in addition to organic fertiliser due to their immediate assimilation by plants which implies less effort to implement an accurate fertilisation. Thus, farmers applied mineral fertilisers on 80% of their land in 2006, in addition to the organic fertilisers already applied (below the threshold of the Nitrates Directive) (Agreste Bretagne, 2008). As a consequence, and since maize is not very sensitive to nutrient excess, they sometimes over-fertilised to guarantee the yield (Barbut & Poux, 2000). Considering that maize production (for grain and forage) represents 31% of the regional arable land (Agreste Bretagne, 2015),

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88 A high correlation is observed between the quantity of organic nitrogen spread, primarily coming from dairy cattle and pig breeding and the nitrogen surplus observed (Agreste Bretagne, 2012). Note that the amount of organic nitrogen applied is calculated on 70% of UAA which corresponds to the average share of UAA where manure application is allowed. However, it does not reflect the real surface area where manure can be applied that varies from at river basin level to another.

89 ICPE are any industrial or agricultural installation that is likely to present a risk or cause pollution or nuisance, especially if it is likely to affect the safety or health of local residents. The concerned farms are indicated in (French Ministry of Ecology, Sustainable Development and Energy, 2015)
the over-fertilisation of maize field has a significant impact on the regional nutrient losses and related impacts.

For cereals that are more sensitive to the excess of nutrients, farmers prefer using to a larger extent chemical fertilisers, for a more accurate fertilisation. Organic fertilisers is only applied on 20 % of the surface areas producing cereals, representing on average 33 kg N/ha, mostly applied as pig slurry. In 2012, one third of the fertilisers applied in Brittany were mineral while two thirds were organic (MEDDE, 2013).

Note that since 2014, the over-fertilisation is limited in reinforced action zones (ZAR) where the nitrogen budget shall not exceed 50 kg N/ha (DREAL Bretagne, 2014).

Inappropriate storage equipment – Manure needs to be stored in proper storage vessels in those periods in which it cannot be applied on land. This is a requirement of the Nitrates Directive. Inappropriate storage can contribute to nitrate run-off and leaching, but also to ammonia emissions, nitrous oxide and methane emissions which are particularly significant in Brittany (Bretagne Environnement, 2006). Run-off and leaching can be caused by the storage of manure in the field, in particular in the case of rain if the heap is not covered. Uncovered manure heaps or slurry tanks also favour ammonia emissions. According to Eurostat, in 2010 in Brittany, 33 %, 32 % and 40 % of the solid manure, liquid manure and slurry vessels were covered (Eurostat, 2013c).

Environmental conditions

Natural factors influence the effects of farming practices and especially fertiliser inputs.

Abundant rainfalls in west Brittany – There are relatively abundant rainfalls in Brittany following an increasing east-west precipitation gradient. In west Brittany (Finistère), more than 1 300 mm of rain fall every year (Bretagne Environnement, 2013b) while the average precipitation in France is 867 mm (FAO-AQUASTAT, 2014). Because of the high amount of water provided by rain, irrigation is not a common practice in Brittany: only 4 % of the water abstracted in Brittany is used for irrigation (7.8 million m\(^3\) per year) (Observatoire de l'eau en Bretagne, 2015a). However, water provided by precipitation is less controllable than irrigation. Thus, some soils may be seasonally water saturated and may induce nutrient run-off, in particular in west Brittany (Barbut & Poux, 2000). Moreover, water overload creates anaerobic conditions which are favourable to the reactions of denitrification. As a result, N\(_2\)O emissions increase under such conditions (Comité de developpement du Finistère, 2012).

Compaction and run-off increased by local soil texture – Soil texture in Brittany is mostly silty and sandy-silty (AgroCampus Ouest, 2011). Silt soils in Brittany have a good water retention capacity. However, in the event of frequent rainfalls, silt texture is favourable to compaction and slacking (Bretagne Environnement, 2014c). Compaction results in the repeated passage of heavy machinery and slacking which consists in the formation of an impermeable crust due to the disintegration of large soil aggregates into smaller aggregates when water falls during rain episodes. Compaction decreases water content and creates anaerobic conditions that increase N\(_2\)O emissions (Barbut & Poux, 2000). Slacking crust prevents the nutrients from reaching the root which can run-off in case of heavy rain.

Low soil organic matter (SOM) content in the northern coast of Brittany and in the east that impacts water retention – Soil organic matter improves soil water retention, and consequently nutrient retention. The lowest SOM content is observed in the northern coast of Brittany and in the east (Bretagne Environnement, 2010a), where nitrogen surplus can be observed. Wetlands, rich in organic matter, exist in the west of Brittany and represent 10 % of the UAA (INRA, Agrocampus Ouest, 2011). In this area, less leaching and run-off are observed (de Vries, et al., 2011a).
High erosion rate that increases phosphorus losses – Brittany presents a high risk for soil erosion. 18% of land area, notably silt-rich and vegetables croplands presents a high to very high risk of water erosion due to significant precipitation (Bretagne Environnement, 2014a). Moreover, west Brittany is subject to high wind erosion (Joint Research Centre, 2013b). Soil erosion is one of the main causes for phosphorus inputs in surface water. This is due to the fact that phosphorus is mainly retained in soils. Volker Prasuhn (2005) has also shown that tillage weakens soil structure and thus enhances particle and phosphorus run-off towards water bodies.

Low pH in north Brittany – Between 2000 and 2004, the northern coasts of Brittany presented high pH while the rest of the territory showed moderate to acidic soils, in particular in the south of Brittany (Bretagne Environnement, 2010b). The average regional pH is 6.3 (Bretagne Environnement, 2011). In acidic soil, the assimilation of nutrients can also be reduced, which is counterbalanced by liming already widely done by farmers in Brittany (Barbut & Poux, 2000). When pH is below 6.5, there is no gaseous ammonia. If the soil pH is higher, such as in the northern coasts of Brittany, ammonia is converted into gaseous form during the volatilisation reaction (Rochette, 2008). However, the application of urea fertiliser in acid soils, the hydrolysis reaction of urea into ammonium ions locally increases the pH and thus enhances ammonia volatilisation (CORPEN, 2006).

Dense hydrographic network – The dense waterway network enhances water flows and run-off, thus increasing nutrient pollution in the waterways. Moreover, there are many small catchments in the region which means that the residence time of water in the river basin is short. As a result, water doesn’t have enough time in the river basin for self-purification and most of the nutrients are transported downstream and in groundwater. Moreover, groundwater reservoirs are fed with nutrients periodically and diluted during heavy precipitations which can explain the temporal variability of nutrient concentrations (Novince & Aurousseau, 2014; Scientific Council of the Environment in Brittany (CSEB), 2008).

5.1.3 Costs of the environmental and health effects

5.1.3.1 Socio-economic description of the region

Brittany represented 5.1% of the French population in 2013. With a GDP of € 86 billion in 2012, it is the seventh largest economic region in France, contributing to 4.1% of the national income (INSEE, 2012). Brittany is also one of the leading agricultural regions of the country.

Brittany is the region with the longest coastline in France (2730 km) representing over 30% of the national coastline (Portail de l’information environnementale en Bretagne, 2008). Thanks to this specificity, Brittany is characterised by rich marine biodiversity. It is also an important shellfish production area of France, accounting for about 29% of the national production in 2012 (in tonnes) (CNC, 2012). Furthermore, Brittany is among the top tourist regions in France (7th region in France) (INSEE, 2013b).

The notable amount of nutrient surplus due to farming practices over the past years has resulted in significant environmental damages and economic losses in Brittany.

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90 According to the MESALES model

91 CNC: National Committee for shellfish farming
5.1.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in Brittany, a wide range of literature was reviewed. Costs were identified and are presented below, based on the classification of economic damages caused by environmental impacts presented in Annex 14.

It is important to note that all cost data reported in this section are taken from the primary studies found in the literature and expressed in the value of the year when the study was conducted.

It should be underlined that some data provided in this section may cover a slightly different area from Brittany. In particular, some data refer to the “Channel and North Sea” (Manche Mer du Nord) marine subregion as identified in the figure below.

![Figure 47 – Channel and North Sea marine subregion (in orange)](image)

Source: (Préfecture maritime de la Manche et de la mer du Nord, 2012)

Clean up and restoration costs (CRC)

Algae removal

In Brittany, the removal of algae on the sea coasts is necessary to reduce visual and olfactory pollution. In response to this problem, local public authorities and private organisations have been working together to clean up the affected areas. Table 20 summarises the different clean up actions and their respective costs. For instance, according to the Centre for Study and Promotion of Algae (CEVA – Centre d’Etude et de Valorisation des Algues), the removal of green algae on the beach areas in Brittany cost € 1.4 million in 2012. In addition to the removal of algae on the beaches and in the shallow waters, further actions have been made to clean up algae in deeper waters (between 20 and 100 centimetres), which cost an extra € 351 000 in 2010 (Préfecture maritime de la Manche et de la mer du Nord, 2012). The removal of algae on the French coastline is estimated to € 30 to € 50 million per year (CGDD, 2011).

Once collected, the algae biomass needs to be processed. The lowest-cost approach for algae processing is to spread them on agricultural soils. In 2010, 30 701 tons of algae have been applied on agricultural soils at a cost of € 5 /ton and 11 112 tons have been composted at a cost of € 15 /ton. In 2010, the total cost for algae processing in the regions Finistère and Côtes-d’Armor was € 320 000. Furthermore, four processing units were also built in Brittany in 2010 to treat algae volumes. The total cost of this operation is estimated at € 6.9 million by the Préfecture de Bretagne (Préfecture maritime de la Manche et de la mer du Nord, 2012). On a smaller scale, according to the mayor of the town of...
Hillion and community advisor for green algae, the urban community of Saint Brieuc has spent € 260 000\textsuperscript{92} for removing, transporting and processing algae in 2011.

The accumulation of nutrients in water bodies induces algae blooms in water intake pipes. The pipes must be cleaned regularly in order to be used. Pipes clogged with algae require additional pumping strength to uptake water and therefore increase the energy costs for water catchment. In 2005, the Loire Bretagne Water Agency (Agence de l’eau Loire-Bretagne) conducted a study estimating the total cleaning costs at € 39.3 million for the basin (CGDD, 2011).

When the water intake stations are affected by eutrophication, the least expensive solution is to move the stations (avoiding the costs of heavy water treatment). According to the General Commissioner for sustainable development, the annual national cost of these measures is € 20-60 million\textsuperscript{93}.

High nitrate concentrations in water also lead to additional cost for their removal in order to make water drinkable (see next section).

### Nutrient removal from waters

According to the CEVA and the French Ministry in charge of Ecology\textsuperscript{94}, in 2009, the waters of the Channel and North Sea marine subregion received the equivalent of 855 000 PE\textsuperscript{95} of pollution (Préfecture maritime de la Manche et de la mer du Nord, 2012). The removal of nitrogen represents 20 % of the total cost of water sanitation, which is estimated at € 70/PE/year (Préfecture maritime de la Manche et de la mer du Nord, 2012). This gives rise to removal costs of € 12 million per year in the Channel and North Sea marine subregion.

At national level, the cost of the additional water treatment for making the water drinkable due to the excess of nitrates is estimated at € 120 to € 360 million per year, with a cost of nitrogen removal that is estimated to € 70/kg N (CGDD, 2011). This represents € 800 to € 2 400/ha UAA/yr in drinking water catchments areas. The total additional cost, including water treatment, water catchments clean-up, algae clean-up on the beach, increasing bottled water consumption, etc. is estimated at € 1 105 to € 1 1675 million per year. Households would contribute to a large share of this cost with an additional expenses of € 1 005 million to € 1 525 million per year (CGDD, 2011).

In France, provided that decontamination treatment of large water bodies would be available, the theoretical clean-up cost of surface and coastal waters regarding the nitrates losses from agriculture is estimated at over € 49 billion per year\textsuperscript{96}. The theoretical cost of the decontamination treatment of groundwater (nitrates) would be higher than € 490 billion (CGDD, 2011).

According to the Drinkable Water Commission of ASTEE (Scientific and technical association for water and environment), nitrate concentration needs to be reduced to a level of 25 mg/L to make the water drinkable. The cost of such additional treatment varies between € 0.4 and € 0.6/m\textsuperscript{3}.

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\textsuperscript{92} This amount includes 50 % of the removal and transport cost and 100 % of the processing cost.

\textsuperscript{93} The sanitary externalities are not taken into account here.

\textsuperscript{94} Ministry of Ecology, Sustainable Development, Transports and accommodations

\textsuperscript{95} PE - population equivalent in waste-water treatment is the number expressing the ratio of the sum of the pollution load produced during 24 hours by industrial facilities and services to the individual pollution load in household sewage produced by one person in the same time (central statistical office).

\textsuperscript{96} The cost are theoretical, provided that
Table 20 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
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<tr>
<td></td>
<td>Expenditure for clean-up and restoration purposes</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td></td>
<td>Algae clean-up on the beach</td>
<td>1 400 000</td>
<td>Brittany</td>
<td>2012</td>
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<td></td>
<td>Algae clean-up in the seawater</td>
<td>351 000</td>
<td>Channel and North Sea marine subregion</td>
<td>2010</td>
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<td></td>
<td>Algae processing</td>
<td>320 000</td>
<td>Côtes-d’Armor and Finistère</td>
<td>2010</td>
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<td></td>
<td>Construction of four processing units</td>
<td>6 900 000</td>
<td>Brittany</td>
<td>2010</td>
</tr>
<tr>
<td></td>
<td>Algae clean up in the water pipes</td>
<td>39 300 000</td>
<td>Loire Brittany basin</td>
<td>2005</td>
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<td></td>
<td>Relocalisation of water uptake stations</td>
<td>20 000 000 – 60 000 000</td>
<td>France</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Removal of nitrogen from water</td>
<td>12 000 000</td>
<td>Channel and North Sea marine subregion</td>
<td>2009</td>
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<tr>
<td></td>
<td>Nitrate abatement costs</td>
<td>€ 0.4 – 0.6 /m³</td>
<td>France</td>
<td>N/A</td>
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<td></td>
<td>Nitrate abatement costs</td>
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<td>France</td>
<td>Per year</td>
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<td></td>
<td>Removal of nitrates in water</td>
<td>49 600 000 000 – 76 200 000 000 € 800 – 2 400 /ha UAA/yr</td>
<td>France</td>
<td>Per year</td>
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<td></td>
<td>Nitrogen removal from water</td>
<td>€ 70 / kg N</td>
<td>France</td>
<td>Per year</td>
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<td></td>
<td>Decontamination of groundwater (nitrates)</td>
<td>490 000 000 000 – 742 000 000 000</td>
<td>France</td>
<td>N/A</td>
</tr>
</tbody>
</table>

Use value damages (UVD)

Eutrophication is not only an environmental problem with significant costs for the local and regional authorities, but also a threat to a number of economic sectors such as tourism or shellfish farming in Brittany.

For these use value damages, very few quantitative data was found in the literature. Most of the costs in this category had not been systematically estimated at the time of this study. In the present study, we are only able to provide some indicative damage costs for this category (see Table 21), mainly based on some qualitative information gathered in the Action plan for the Protection of the Marine Environment (2012).

In the context of recreation and tourism, eutrophication can have a range of adverse impacts, such as the loss of tourism revenue due to nuisance caused (e.g. visual). According to a survey-based study\(^\text{97}\), 56 % of the Channel and North Sea marine subregion population was concerned by the green algae issue in 2011. A study (Gicquel, 2003) in the bay of Saint Brieuc showed that 23.3 % of the visitors complained about green algae in 2003. In the town of Saint-Michel-en-Grève in the bay of Lannion, the number of hotels has considerably decreased in 30 years from six to only one that remains, which is in part caused by the green algae blooms on the coast. To estimate the shortfall for touristic accommodation activities (hotels, camp sites, etc.), the Tourism Committee of Côtes-d’Armor

\(^{97}\) Phone surveys led by the LH2 institute on 1 315 French people aged above 18.
commissioned a study in 2011, which has estimated the total economic loss due to green algae at € 800,000.

No study has been made on the impact of green algae blooms on real estate. Nevertheless, according to interviews carried out in the context of the Action plan for the Protection of the Marine Environment in the Channel and North Sea (2012), real estate agents in the Lannion bay indicated that the number of sales and the sale prices have decreased by up to 50% since the issue attracted the attention of global media.

Since eutrophication affects water quality, the shellfish production sector in Brittany is also impacted. Brittany is the second French region for oyster production (21% of the national sales) (Ministry of Agriculture). According to the National Committee for shellfish farming (Comité National de la Conchyliculture – CNC), the production losses reached 15% in some areas because of the algae. As a consequence, some shellfish farmers have been forced to change their production techniques, which have resulted in increased working time on the farm (multiplied by up to ten).

In France, revenue losses due to eutrophication are estimated between 70 and 100 million euros a year for coastal municipalities. (CGDD, 2011).

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Damage estimation</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduced revenues in tourism, fishery sector etc.</td>
<td>Decrease of the number of hotels in Saint-Michel-en-Grève</td>
<td>(from 6 to 1) meaning -5 over 30 years</td>
<td>Saint-Michel-en-Grève (in Lannion bay)</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Loss for tourism (touristic accommodations)</td>
<td>800 000 €</td>
<td>Côtes-d'Armor</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Decrease in the yield of oysters production</td>
<td>Between -2 and -15%</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Increase of working hours on shellfish farms</td>
<td>working hours multiplied by up to 10 in certain areas</td>
<td>Estuary zones of Aven and Belon</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Number of medical cases</td>
<td>7</td>
<td>Brittany</td>
<td>to date</td>
</tr>
<tr>
<td></td>
<td>Decrease of real estate sales</td>
<td>30 to 50%</td>
<td>Lannion bay</td>
<td>2012</td>
</tr>
<tr>
<td></td>
<td>Decrease of real estate sale prices</td>
<td>About 50%</td>
<td>Lannion bay</td>
<td>2012</td>
</tr>
<tr>
<td></td>
<td>Revenue losses due to eutrophication for coastal municipalities</td>
<td>70 000 000 - 100 000 000</td>
<td>Coastal areas, France</td>
<td>per year</td>
</tr>
<tr>
<td></td>
<td>Loss of sensitive species in favour of opportunistic species</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Increased healthcare expenditures</td>
<td>Increased cases of water pollution related disease</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
</tr>
</tbody>
</table>

**Passive use value damages (PUVD)**

**Water pollution – Negative effect on health**

Green algae when decomposing generate large quantities of NH₃ and H₂S, both toxic gases, which can impose direct threats to human health and impacts on ecosystems.

Since 2008, ten medical cases were reported including four workers in charge of the removal of the green algae and three recreational visitors (Préfecture maritime de la Manche et de la mer du Nord,
Although no estimation of PUVD on human health in Brittany region has been found, the Côtes-d’Armor delegation of the health regional Agency (Délégation territoriale de l’agence régionale de santé – Côtes-d’Armor) is currently carrying out studies to evaluate health hazards related to green algae.

**Water pollution – Negative effect on ecosystems**

Eutrophication can have diverse impacts on ecosystems and biodiversity. For instance, more than 30 wild boars were found dead on the northern coast of Brittany following the peak of algae blooms in 2011. On a longer term, eutrophication impacts benthic species. These impacts are difficult to evaluate, however studies have shown that some sensitive species have disappeared in the affected areas in favour of opportunistic species (such as *Ulva armoricana* and *Ulva rotundata*) (Préfecture maritime de la Manche et de la mer du Nord, 2012). The algae also represent a physical barrier to resources for some bird species. Nevertheless, the algae are an important food resource for Brent geese (“Bernaches Cravant”) (Préfecture maritime de la Manche et de la mer du Nord, 2012).

Although it is clear that algae blooms lead to economic cost falling in the PUVD category, only patchy quantitative data is available. No regional-level estimates of the magnitude of these damages could be found.

**Air pollution**

As previously mentioned, nutrient surplus can also lead to air pollution (see Table 22 for details) and consequently cause physical and economic damages to human health and ecosystems in Brittany.

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Emissions of pollutant (kt)</th>
<th>Share of Brittany/France (%)</th>
<th>Share of agriculture in pollutant emission (%)</th>
<th>Emissions of pollutant due to agriculture in Brittany (kt)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH₃</td>
<td>148.6</td>
<td>18.9</td>
<td>99</td>
<td>147.7</td>
</tr>
<tr>
<td>NOₓ</td>
<td>71.3</td>
<td>5.1</td>
<td>19</td>
<td>13.4</td>
</tr>
<tr>
<td>N₂O</td>
<td>18.4</td>
<td>6.9</td>
<td>93</td>
<td>17.1</td>
</tr>
</tbody>
</table>

*Calculated value

The economic losses can be both tangible and intangible, expressed in terms of direct lost market and non-market values. Take the cost of human health due to air pollution as an example, the market value losses are typically evaluated in term of direct expenditures incurred for medical treatment and hospitalisation, reduced wages and productivity due to the lower performance affected by pollution-related disease. On the other hand, the non-market value losses can be only estimated in a contingent market, in which a survey-based valuation technique can be used to assess the individual’s willingness to pay for avoiding the risk of being infected by the disease or death. Based on the use of different valuation techniques Corjan Brink (2011) has estimated average damage cost of public health and ecosystem in the EU 27, caused by NH₃, NOₓ and N₂O emissions, respectively. This study is used as a reference for estimating the total economic damages in Brittany.

NH₃ emission caused by agriculture in Brittany is estimated at 147.7 kt of NH₃ in 2004. The average unit damage cost for health impacts by airborne NH₃ in France is € 15 /kg N (Brink & van Grinsven, 2011), which leads to a total cost of approximately € 1.8 billion for health impacts caused by airborne NH₃ in Brittany alone. In addition, the same author has also estimated that the average damage cost of deposition of NH₃ on terrestrial ecosystem ranges between € 2 and € 10 /kg N in the EU-27. The lower

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98 Considering that 1 ton of NH₃ is 0.823 ton of N
bound is derived from the EU NEEDS project (Ott, et al., 2006), representing the cost of biodiversity and ecosystem restoration.

As regards NO\textsubscript{x} emission, agriculture in Brittany generated 13.4 kt of NO\textsubscript{x} in 2004. Using the unit damage cost for health impacts estimated for airborne NO\textsubscript{x} at € 25 /kg N (Brink & van Grinsven, 2011), the associated economic damage in Brittany can be estimated to approximately € 156 million\textsuperscript{99} in total.

Finally, N\textsubscript{2}O emissions' impacts on public health cost € 1-3 /kg N in the EU27 (Brink & van Grinsven, 2011). Therefore, the cost of N\textsubscript{2}O emissions on public health in Brittany is estimated between € 5-16 million\textsuperscript{100}.

**Policy action costs (PAC)**

Primarily in response to the green algae damages, policy actions have been implemented in order to protect the environment and to support scientific monitoring. In particular, the French government has initiated a 2010-2015 national plan to fight against green algae. The total budget for the plan is € 134 million over 5 years, and Brittany represents nearly 86 % of the area covered by the plan.

In terms of policy actions to protect the environment, an 8-year program of assistance to farm owners has been set up by the local government to help farmers with administrative procedures and subsequent changes to meet environmental standards regarding nitrogen emissions. It is estimated that an average cost per case is around € 11 000, of which 30 % is paid through state subsidies (Préfecture maritime de la Manche et de la mer du Nord, 2012). In the seven affected bays of the Channel and North Sea marine subregion, the total cost of this assistance process would be € 28 million, or € 3.5 million per year.

In addition, policy actions have also focused on informing the local population and raising public awareness about farming pollution and environmental impacts in Brittany. This type of communication action costs € 2 697 000 per year (state subsidies included) in the north of Brittany. Additional costs are also incurred for producing brochures and signs to inform visitors and the local population about the green algae and the associated sanitary risks. In 2010, this action alone cost € 47 000 for the Channel and North Sea marine subregion (Préfecture maritime de la Manche et de la mer du Nord, 2012). Furthermore, in 2010, to respond to the significant impacts of green algae on tourism, the Regional Tourism Committee of Brittany launched an advertising campaign to promote tourism and restore the positive image of the area costing € 125 000 (Préfecture maritime de la Manche et de la mer du Nord, 2012).

Finally, between 2007 and 2011, the affected bays (Saint Brieuc, Douarnenez, Saint Michel, Locquirec, l’Horn Guillec, etc.) put up agro-environmental measures to fight against the eutrophication phenomenon at the cost of € 1.3 million (Préfecture maritime de la Manche et de la mer du Nord, 2012).

The general concern about the eutrophication situation on the Brittany coast has led to the establishment of several scientific research projects. Several monitoring projects with the aim to control the eutrophication phenomenon on the coast are ongoing with an annual cost of € 380 000 in the Channel and North Sea marine subregion (Préfecture maritime de la Manche et de la mer du Nord, 2012). A scientific project in Brittany was developed to assess the nitrogen concentration levels in 2010 and cost € 343 000. In addition, the monitoring of phytoplankton in the Channel and North Sea marine subregion in 2009 cost € 674 000 and the sanitary monitoring of microtoxines liberated by microalgae cost

\textsuperscript{99} Considering that 1 ton of NO\textsubscript{x} is 0.467 ton of N

\textsuperscript{100} Considering that 1 ton of N\textsubscript{2}O is 0.318 ton of N
€ 41 000 (Préfecture maritime de la Manche et de la mer du Nord, 2012). Moreover, according to the study (Préfecture maritime de la Manche et de la mer du Nord, 2012), the scientific research around eutrophication cost € 63 000 per year based on the rough estimations made by two research directors at IFREMER\textsuperscript{101}.

### Table 23 – Policy actions costs found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (euros)/year</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Restoration of the image of the region</td>
<td>Advertising campaign to promote tourism</td>
<td>125 000</td>
<td>Brittany</td>
<td>2010</td>
</tr>
<tr>
<td>Positive actions for environment on the watersheds</td>
<td>Assistance to help farms meet the environmental standards</td>
<td>3 500 000</td>
<td>Affected bays of the Channel and North Sea marine subregion</td>
<td>Since 2004</td>
</tr>
<tr>
<td></td>
<td>Education and awareness campaign</td>
<td>2 697 000</td>
<td>North of Brittany</td>
<td>Per year (average)</td>
</tr>
<tr>
<td></td>
<td>Brochures and signs for the surrounding towns</td>
<td>47 000</td>
<td>Channel and North Sea marine subregion</td>
<td>2010</td>
</tr>
<tr>
<td>Scientific monitoring of the eutrophication situation</td>
<td>Protection contracts (studies, measures, surveillance, etc.)</td>
<td>380 000</td>
<td>Channel and North Sea marine subregion</td>
<td>Per year (average)</td>
</tr>
<tr>
<td></td>
<td>Nitrogen concentration measures</td>
<td>343 000</td>
<td>Brittany</td>
<td>2010</td>
</tr>
<tr>
<td></td>
<td>Monitoring eutrophication (OSPAR program)</td>
<td>No data</td>
<td>Channel and North Sea marine subregion</td>
<td>2009</td>
</tr>
<tr>
<td></td>
<td>Sanitary monitoring of microtoxines (microalgae Alexandrium)\textsuperscript{102}</td>
<td>41 000</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Phytoplankton monitoring</td>
<td>674 000</td>
<td>Channel and North Sea marine subregion</td>
<td>2009</td>
</tr>
<tr>
<td></td>
<td>Eutrophication research (Ifremer)</td>
<td>63 000</td>
<td></td>
<td>2009</td>
</tr>
</tbody>
</table>

### 5.1.4 Good practices to reduce nutrient losses at farm level

In Brittany, the first set of measures identified in this report aims to reduce the local source of pollution by controlling the geographic distribution of livestock, adapting the feeding strategies, processing manure to decrease the nutrients content of manure, improving the fertilisation management plan for N and P and converting arable land to unfertilised grassland in areas at risk. The second set of measures that has been selected focuses on the reduction of nutrient loss during manure storage and housing, either by covering slurry tanks to prevent ammonia emissions and/or installing “V” scrapers in pig buildings. A third set of measures has been identified to prevent nutrient losses when manure is applied on fields or on pastures by optimizing grazing intensity through rotational grazing, considering climate conditions before spreading and implementing grass strips along hedgerow. Finally, the last set of measures aims to improve soil quality and decrease the amount of fertiliser that the farmers needed to

\textsuperscript{101} French Research Institute for Exploitation of the Sea

\textsuperscript{102} The national cost of such monitoring, cofinanced by the ministry in charge of Food resource, was € 123 000 in 2009 according to Ifremer. In a study, Yann Laurans (2012) evaluated the cost for the Channel and North Sea marine subregion by dividing in three the national cost.
buy by processing manure to ease its use and transfer, using catch crops and preferring conservation tillage techniques.

### 5.1.4.1 What has already been done in the region

Brittany has been designated as a Nitrate Vulnerable Zone (NVZ) in the implementation of the EU Nitrates Directive. Therefore, the region must comply with mandatory requirements under the French Nitrates Action Programme (NAP). Measures include mandatory nitrogen fertilisation management plans for the use of manure, mandatory soil coverage, and a ban on wetland drainage (DREAL Bretagne, 2014). More restrictive measures apply to certain areas with pollution risks around drinking water catchments (in French “Zones d’Actions Renforcées” (ZAR)). For instance, the global nitrogen budget shall be lower than 50 kg N/ha UAA per year or on average over the last three years. In ZAR and in bays subject to green algae blooms (in accordance with the Green Algae Control Plan), the input of (organic and mineral) nitrogen fertiliser is restricted to 210 kg N/ha. In addition, soil coverage during winter rainy periods (with high risk of nitrate leaching) and grass strips along water courses over a width of 10 metres in ZAR (versus 5 metres usually) are mandatory, while the ploughing up of grasslands in floodplains and wetlands drainage is forbidden.

Since 2011, the amount of phosphorus that can be applied on fields is limited in areas concerned by the River Basin Management Plans (SDAGE) 2010-2015 for large farms considered as Classified Installation for Environmental Protection (ICPE)\(^{103}\). The SDAGE concerns 12 % of the UAA in Brittany (Landrain & Pape, 2012). In these areas, the fertilisation plan shall be balanced\(^{104}\) for farms producing more than 25 000 kg N/yr and the application of P shall not exceed 80 to 95 kg P/ha\(^{105}\) for farms producing less than 25 000 kg N/yr (Landrain & Pape, 2012; Chambre d’Agriculture Bretagne, 2013a). In addition, the farms shall implement landscape management measures to limit phosphorus losses. The phosphorus threshold also indirectly limits the nitrogen application. Since the N:P ratio varies according to the organic matter, this means that only about 140 kg N from pig slurry can be applied per hectare (Loussouarn, 2012; Chambre d’Agriculture Bretagne, 2013a). On the contrary, the quantity of phosphorus contained in manure from dairy cattle that can be applied is lower than the threshold otherwise the amount of nitrogen provided would exceed the amount authorised by the Nitrates Directive (Chambre d’Agriculture Bretagne, 2013a).

Brittany has largely promoted measures such as adaptation of feeding and fertilisation strategies, band spreading, immediate incorporation of manure, and improved manure management equipment and installations (DREAL Bretagne, 2014; Portail de l’état en Bretagne, 2010). The region also aims to achieve 100 MW of energy production through anaerobic digestion of manure and other materials by 2020. Many studies are on-going to improve the manure processing techniques, in particular in experimental farms. In 2007, the excess of nitrogen in Brittany was reduced by the transfer of manure (41.6 % of the nitrogen excess), manure treatment – in particular composting and biological treatment of slurry (25.7 % of the nitrogen excess), feeding practices – especially biphase feeding (23.5 % of the nitrogen excess) and by the reduction of the number of livestock (8.9 %) (Bretagne Environnement, 2009a). Between 2000 and 2010, the amount of organic nitrogen applied decreased by 38 %, the consumption of mineral fertilisers decreased by 26 % for N in 12 years and 41 % for P in 7 years (FDSEA Bretonnes, 2012). In addition to changes in practices, farmers have invested € 1 billion (on 20 years), mostly self-financed (FDSEA Bretonnes, 2012). The nutrient surplus and losses still remain and certain

\(^{103}\) The concerned farms are indicated in (French Ministry of Ecology, Sustainable Development and Energy, 2015)

\(^{104}\) with a tolerance rate of 10 %

\(^{105}\) depending on the type of livestock bred
measures have a high potential for improvement, such as livestock feeding, manure cover during storage, manure processing, nitrogen and phosphorus fertilisation plans, rotational grazing and the conversion of arable into unfertilised grassland.

### 5.1.4.1 Good practices to reduce the nutrient losses in livestock production

**Control the geographic distribution of livestock**

As mentioned above, the uneven distribution of livestock lead to the production of a high quantity of manure that can locally exceed the carrying capacity of the farm or the territory. Thus, in addition to good livestock management practices, the control of the geographic distribution of livestock through the limitation or the decrease of livestock density and production intensity in areas presenting an excess of nutrients would be needed in the long term in order to work towards a sustainable reduction of nutrient surplus.

The control of the livestock density in areas with nutrient surplus is possible through the reduction of the number of livestock and/or the incitation of the installation of farms in area with no surplus and higher manure demand. In areas formerly considered to have an excess of manure production compared to the amount of organic fertiliser that can be applied (ZES), the reduction of the number of animals has already contributed to the reduction of the excess of nitrogen (Goypieron, 2012). A study showed that, in the short-term it pig and laying hen breeders choose to treat manure or transfer it. In the long term they prefer reducing the number of livestock by 20 % for pigs and 40 % for laying hens. Table poultry breeders prefer composting and transferring manure in the short-term and reducing the number of livestock by 30 % in the long-term (Djaout, et al., 2009).

However, controlling the number of animals could have a serious impact on the profitability of the farms as it would lead to a reduction of the quantity of meat and live piglets produced and economical losses in the short term. The reduction of concentrated feed purchases and veterinary services does not compensate the reduction of revenue and the possible additional expenses. Larger farms may be impacted less as they benefit from the economies of scale and have better access to capital. The feasibility of the reduction of the number of animals in farms has been studied by the Chambre d’Agriculture de Bretagne in 2009 in two river basins as a solution to reduce the phosphorus excess. The reduction of livestock in pig breeding-fattening system with 140 sows would result in a loss of € 40 000 to € 45 000 per farm, with a possible workforce loss and a very limited capacity to face the debt (Landrain & Pape, 2012).

Moreover, the measure could have long-term effects on the regional economy. In addition, this could possibly cause a decrease of the number of installations. As the leading French agricultural region, agriculture and especially animal breeding is the economic and cultural base of Brittany. The limitation and, in the long term, the reduction in the number of heads may affect the regional agriculture and economic competitiveness (Préfecture de Bretagne, 2013). Nevertheless, considering that the manure production exceeds the carrying capacity of the territory in several areas of the region (Portail de l’état en Bretagne, 2010), this measure falls in the range of options in order to aim for a durable reduction of nutrient surplus.

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106 Considering the economic consequences of taking a decision in the short term or in the long term. For instance in the short term, the farmer may not be able to invest in another activity since the previous investments may not be amortised yet.
**Adapt feeding strategies to reduce nutrient load in manure**

To reduce nutrient load in manure, a good practice is to adapt the feeding strategies. Two approaches can be considered to reach good results:

- Adapt the feeding strategies to stick as closely as possible to the animal’s needs, which are different in the different stages of the animal’s life,
- Increase the assimilation of the nutrients by animals

The revised feeding strategy should reduce the load of nutrients in manure. The quantity of manure produced can also be reduced, depending on the measures implemented.

These measures are particularly adapted to Brittany as it is the first region for animal breeding in France, in particular for pig production. The reduction of nutrient content in manure in a region with such an important number of livestock may have considerable impact on nutrient management at the farm level. These measures are already largely promoted. In 2010, 90 % of the farmers used biphasic feeding for pigs (Agreste Bretagne, 2012). Over the past 20 years, the phosphorus content of industrial feed used in the poultry sector in France has also decreased from 5 % to 28 % depending on the species (Landrain & Pape, 2012).

These practices can still be improved or further implemented. Biphasic feeding could be replaced by multiphase feeding with a similar cost (Institut de l’élevage, IFIP, ITAVI, 2010). Among the feeding practices, the addition of phytase could be very interesting for pig and poultry production in Brittany, especially considering the high amount of P manure produced. According to the COOPERL, a Breton cooperative for pig farmers, “precision nutrition” allowed the amount of nitrogen excreted to be reduced by 40 % in 20 years. The use of phytase allowed phosphorus excretion to be reduced by 60 % (Delcour, et al., 2013). The Chambre d’Agriculture de Bretagne proposes information to farmers to help them using phytase (Heugebaert, et al., s.d.).

In France, a study calculated that € 2.5 per livestock unit for dairy cattle and € 7.4 per livestock unit for suckler cows can be saved by reducing the amount of concentrate and fodder to fit the animal’s nitrogen needs\(^{107}\) (Martin & Mathias, 2013). However, these costs do not include the possible addition of amino-acids for pigs or phytase. The additional costs related to the management change are also not included. Hence, the profitability of this measure should be further studied to determine whether this measure is really economically sustainable.

**Use “V” scraper in pig buildings**

The “V” scraper is a solid-liquid fraction separation device that can be directly used in buildings under a slatted floor – in France, 95 % of pigs are bred on such a floor (INAPORC, 2014) – while other separation devices are usually used once the manure is collected.

With the “V” scraper, the solid fraction concentrates 90 % of the phosphorus and 55 % of the nitrogen from pig manure while ammonia and nitrous oxide emissions are reduced by 54 % and 49 % respectively (Landrain, et al., 2009). For a breeding-fattening system with 150 sows, the investment cost of a “V” scraper is estimated at € 88 051 and the running costs € 2 800. The “V” scraper is the cheapest device per kg P\(_2\)O\(_5\) extracted compared to other concentration devices, due to its low running cost (Loussouarn, et al., 2012). This device is still experimental. It has been tested on an experimental farm in Guernévez since 2006 (Chambre d’Agriculture Bretagne, 2014a). Between 2010 and 2013, 25 to 30

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\(^{107}\) The costs are based on the 2010 average cost for fodder and concentrated feed in Brittany, extrapolated to France.
pig holdings were built with this system for any pig breeding system (Chambre d'Agriculture Bretagne, 2014b).

The high investment cost is a barrier to the implementation of this system. Currently, it is mostly implemented in new buildings. The process is simple and robust and the breeders are generally satisfied. Composting with 2% of straw, the material obtained is rich in phosphorus and organic nitrogen and complies with the legislation relative to the organic fertiliser (NF U 42-001). In addition, a regular increase of the average daily weight gain and a regular decrease of the feed gain ratio were observed between 2007 and 2013 in the experimental farm of Guernévez for fattening system (Chambre d'Agriculture Bretagne, 2014b).

**Cover slurry tanks to prevent ammonia emissions**

According to Eurostat, only 40% of the slurry tanks were covered in 2010 in Brittany (Eurostat, 2013c). Covering the slurry reservoir limits the contact between manure and the atmosphere, limiting nitrogen loss by volatilisation. Additionally, covers also provide longer storage capacity since the quantity of liquid manure is not uselessly increased by rainwater. This is particularly important in a region like Brittany with high precipitation. In areas with more than 1 300 mm of rain per year, such as in the Finistère (western Brittany) (Bretagne Environnement, 2013b), farmers must increase the storage volume by 60% (Chambre d'Agriculture de Bretagne; GIE Elevages Bretagne, 2012). Assuming an application cost of the slurry of € 2.0/m³, tank coverage is economically profitable in the event of more than 1 000 mm of rainfall per year (Chambre d'Agriculture de Bretagne; GIE Elevages Bretagne, 2012). Hence, tank coverage will be particularly interesting in western Brittany.

**Process manure to eliminate, extract or concentrate the nutrients**

Considering the local excess of manure compared to the crop needs, processing manure is a particularly relevant solution in Brittany to 1) decrease the nitrogen content of manure (in particular nitrogen) and 2) ease its use and sale by concentrating nitrogen and phosphorus, increasing the share of ammonia compared to other N forms and obtaining a stable material with a small volume that can be easily transported and stored. Processing or exporting manure is mandatory for farmers in areas where manure production exceeds 170 kg N/ha/yr since 1998. Processing manure would also ease manure transfer that is frequent in the region in case of excess of manure compared to the legal requirements, mostly between neighbouring farms and other farms within the same canton. Each year, an average of 200 000 tons of manure are transferred out of Brittany (Bégos, 2009).

For pig manure, the main treatment in the region is the activated-sludge process with a nitrification-denitrification (NDN) treatment, preceded or not by a separation of the liquid and solid fraction of pig manure (Levasseur & Lemaire, 2006; Levasseur & Beline, 2001; Gerard, 2014). For beef and dairy cattle, and solid manure in general, the main treatments in Brittany are composting and anaerobic digestion (Chauvin, 2010). Even though these techniques are more developed than others, they can still be highly promoted considering the relevance of their use in the region for the productive use of nutrients from slurry.

**Nitrogen abatement through nitrification-denitrification (NDN) treatment of pig manure**

In 2011, 443 slurry treatment plants where installed in Brittany, for about 630 breeders involved in individual or collective activities (Goypieron, 2012). More than 85% of the plants use an activated sludge process for pig manure (Loussouarn & Le Bris, 2012). This process often begins with a mechanical liquid-solid fraction separation. The liquid fraction is treated in a reactor by a nitrification-denitrification (NDN) process during which 70% of the nitrogen is transformed into invert gas (N₂). The solid fraction, rich in phosphorus, is exported and possibly treated to extract phosphorus.
A survey conducted on 28 regional plants in 2009 showed that on average 93% of the nitrogen and 86% of the phosphorus were captured or exported (Loussouarn & Le Bris, 2012). The initial investment was on average € 450 000 which equals to € 6.13/m³ slurry in amortised costs. Subsidies represented 26% of this amount. The running costs accounted for € 1.47 to € 7.20/m³ slurry, with an average at € 3.51/m³ slurry. This represented on average € 33 000 per year. Despite the high cost, breeders are satisfied of their investment that provides autonomy and flexibility in their manure and nutrient management. The negative elements that they mentioned concerned the undersizing of the reactor, the high time spent for monitoring the process and the infrastructure that could be optimised to ease the recovery of the solid fraction of pig manure (Loussouarn & Le Bris, 2012). In 2008, Veolia, the National School of Chemistry in Rennes and the Chambre d’Agriculture of the Finistère region have developed a project called “Zero nuisance piggeries”¹⁰⁸ to improve the treatment of pig manure at the farm scale. After a separation of the solid and liquid fraction of the slurry, the liquid fraction is treated by an NDN process while the solid fraction is composted. After treatment, the total nitrogen content decreased from 1.3 g N/L to 0.08 g N/L and phosphorus decreased from 0.22 g PO₄³⁻/L to 0.035 g PO₄³⁻/L (Veolia Environnement, 2008). The stakeholder consultation of this project highlights the high potential of this measure to decrease the nutrient losses.

**Nutrient concentration in digestate from anaerobic digestion treatment**

Many initiatives of installation of anaerobic treatment plants have been launched and many calls for projects around agricultural biogas reactors have been made. Anaerobic digestion (AD) does not reduce nutrient surplus as such but produces a stable material that is concentrated in nutrients (in addition to energy). This digestate can easily be transferred to other river basins that would need nutrients. Since the digestate is free of active pathogens, it is also suitable for vegetable production. Manure, and in particular pig slurry, may require the addition of crop residue to balance the C/N ratio of the treated materials that may encourage intensive energy crops production such as maize that consumes a significant amount of nutrients and water.

The figure below shows (green circles) the agricultural biogas reactors that have been installed in the region as of 2014 for electricity and heating.

¹⁰⁸ [www.zeronuisancepiggeries.fr](http://www.zeronuisancepiggeries.fr)
In January 2014, anaerobic digestion plants produced 46 million kWh (Chambre d'Agriculture de Bretagne, 2014). According to a representative of the regional agency in charge of energy and environment (ADEME Bretagne), 40 biogas reactors are installed in the region, among which 30 use manure as a reaction substrate (Web-agri, 2014). In 2013, a call for projects of agricultural AD in Brittany called “Plan Biogaz” was led by departmental councils (AILE, 2013) to encourage and assist project leaders in this transformation for manure at the farm and industrial level. Financial support is provided to the selected projects.

Currently, two types of initiatives exist for treating manure by AD. The main type of plant is the large co-digestion plants that can treat 50 000 to 100 000 t slurry (Chauvin, 2010), such as the Biogasyl initiative implemented in 2008 (Douard, 2010) or the Geotoxia initiative implemented in 2011 (Idex Services, 2011). The other type of plant is the individual co-digestion plant at farm level that can treat from 3 000 t slurry. The biogas installation in Plélo is one of these examples. The farm has an intensive pig breeding activity and has installed in 2007 the equipment to process manure by AD. The electric capacity of the reactor is 130 kW (+75 kW backup) and the heating capacity is 155 kW (+90 kW backup) with a total investment cost of € 900 000. The first year, the balance sheet was positive in 2010, mostly due to the sale of electricity (AILE, 2010). The example of the farm in Plélo highlights that despite the high investment, which are one of the major obstacles for this measure, the plant is still profitable. Initiatives to implement AD on farms are developing and examples of successful initiatives are increasing. This is
for instance the case of Bio4gas\textsuperscript{109} or Valogreen\textsuperscript{110} that helps farmers to implement AD reactors on their farms since 2012 and 2013 respectively. According to Bio4gas, the initial investment for a plant producing 28 kW (thermal energy) and treat manure from 250 sows is € 250 000 – 270 000 with a return on investment estimated at six to nine years (Vergonjeanne, 2012).

However, while the development of anaerobic digestion appears to be an interesting option for recycling manure and facilitating its transfer and use in the form of a digestate, it is also important to note that this technique requires the use of substrate such as crop residues from corn production. Hence, it may induce land use change to more intensive cropland and increase the surface of maize production that is a source of nutrient losses considering the current practices.

*Concentration of phosphorus (and nitrogen) by composting or centrifugation*

Other interesting practices could be used by farmers. In particular, techniques aiming at concentrating phosphorus are very relevant considering the excess of P manure in the region and in the soils. Concentrating phosphorus from any type of manure will ease its transfer to other areas in need.

After mechanical liquid-solid fraction separation, the solid fraction is already more concentrated in phosphorus than the initial manure. This fraction can be further concentrated to extract phosphorus. Composting and centrifugation are the most efficient methods to extract phosphorus from manure or slurry with the addition of straw promoted by the Chambre d’Agriculture. They can catch up to 100 % of the P contained in manure (Loussouarn, et al., 2012). For a breeding-fattening system with 150 sows, the total costs varies from € 6.38 to € 9.44 /kg $P_2O_5$ extracted with an investment cost varying from € 71 871 to € 100 000 (Loussouarn, et al., 2012).

For cattle, the cost of manure composting was estimated to € 1.7-1.8 /t manure. The total additional cost of the use of composted manure, including composting cost, material and workforce for application is estimated to € 18 to € 24 /ha. The nutrient content of the composted manure is well suited for pasture fertilisation (Chambre d’Agriculture de Bretagne; Bretagne Eau Pure, 2005). Composting can also be used for poultry manure.

*Phosphorus precipitation in struvite*

This practice allows the catching of about 90 % of P in pig slurry (Capdevielle, et al., 2013). The struvite concentrates phosphorus with a lower volume than with other processes such as composting (Chambre d’Agriculture de Bretagne, 2012a). While phosphorus precipitation as struvite is sometimes used to extract phosphorus from sewage sludge, it is seldom used for effluents such as slurry. Indeed, 80 % of the phosphorus contained in slurry is in the particulate part. Consequently, phosphorus has to be dissolved to increase the share of phosphorus that can precipitate. Currently, experiments are led to improve the quantity of crystals formed (Capdevielle, et al., 2013; Daumer, et al., 2008).

As a consequence, the main barrier to the use of this technique is the high cost of the fertiliser produced compared to other fertilisers. This is due to the high cost of the acidification stage. This technique could become economically viable in the future in case of increase of the costs of the phosphorous fertilisers (IRSTEA, 2014)\textsuperscript{111}, the increase in demand for fertiliser, the increase in price of phosphate rock and mining exploitation and the depletion of existing phosphate supplies (European Commission, 2013c).

\textsuperscript{109} www.bio4gas.fr

\textsuperscript{110} www.valogreen.fr

\textsuperscript{111} IRSTEA: French National Research Institute in Sciences, and Technologies for environmental and agriculture
**Optimise grazing intensity**

This practice is particularly relevant for the region considering the high share of pasture in Brittany and the surplus of nutrients, in particular nitrogen in pasture. The surplus is partly due to the difficult management of manure from grazing animals that is concentrated on 10% of the pasture surface (Agreste Bretagne, 2008). Rotational grazing can be used to prevent over-grazing and localised deposition of manure and to optimise the grazing intensity so that plants can regenerate. Regularly moving the feeders or the drinking water around the pasture can also help to decrease the concentrated deposition of manure. Soil compaction can be avoided by removing stock from pasture during and shortly after heavy rainfall, significantly reducing treading damage.

**5.1.4.2 Good practices to reduce the nutrient losses in crop production**

*Improve the fertilisation management plan for N and extend the elaboration of a fertilisation plan for P to the whole region in order to balance the nutrient budget*

For nitrogen, farmers should use more accurate and specific information to establish their fertilisation management plan, in particular for maize production and pastures that present a high nitrogen surplus. In 2011, half of the farmers in Brittany took into account the nitrogen residues in soil to establish their fertilisation plans (Agreste Bretagne, 2013).

In Brittany, the Chamber of Agriculture (Chambre d’Agriculture de Bretagne) has published a fertilisation guide to help farmers with a step-by-step procedure (Chambre d’Agriculture de Bretagne, 2013b). A tool called Equi-Ferti is also available since 2013 (DRAF Bretagne, 2015). Information on the quantity of nutrients that should be provided to crops and the theoretical nutrient content of manure are available in the NAP. Data on the nutrient residues in soil are provided each year by the Chamber of Agriculture of Bretagne (Chambre d’Agriculture de Bretagne, 2015). Frequent manure and soil analysis that allow for the collection of data on the nutrient content of manure and on nutrient residues in soil that are specific to the farm should be promoted in order to develop a more accurate nutrient budget and fertilisation management plan. In 2011, 40% of the holdings performed analyses for solid manure and 85% for slurry (25% and 61% respectively in 2004) (Agreste Bretagne, 2013).

A fertilisation management plan should also be prepared for phosphorus, in particular in phosphorus-saturated areas, which is the case in 70% of the cantons in 2000-2004 (Bretagne Environnement, 2011).

Advisors from farm management centres, agricultural cooperatives or the milk recording agencies helped two-thirds of the farmers to prepare their fertilisation management plan in 2011. Training advisors is also essential to help them prepare an accurate fertilisation management plan. Local associations would also be important actors by helping farmers with the methodology during their first years of fertilisation planning and thus avoiding discouragement.

*Use catch crops to reduce nutrient leaching, reduce erosion and increase the SOM content*

The 5th NAP specifies that each farmer has the obligation to cover bare soils during the period of high risk of leaching. In 2011, only 4% of agricultural soils were left uncultivated during the critical periods (DRAAF Bretagne, 2013b). Therefore, while it is important to encourage farmers to continue using this measure, the potential for development of this measure is low since it is already largely implemented in the region.

In 2010, catch crops were implemented on 58% of the farms, compared to 17% at national level the same year and 15% in Brittany in 2000 (Agreste Bretagne, 2012). While Britain farmers has significantly improved their practices regarding catch crops, this measure should continue to be promoted considering its benefit in terms of protection against erosion, and nutrient leaching. The cost of
implementation of catch crops varies from € 10 to 70 /ha for the seeds, in addition to the cost of seeding and destruction (Chambre d’Agriculture de Bretagne, 2012b).

Chemical destruction (use of herbicides) of catch crops will be forbidden in 2016 according to the 5th NAP. Note that nitrogen fixing crops, such as legumes, should not be used considering the additional nitrogen capture from atmosphere which would add up to the nitrogen surplus.

**Promote conservation tillage techniques to increase SOM and reduce erosion**

In Brittany, many conservation technique initiatives (called in French “Techniques Culturales Sans Labour” (TCSL) have been led. They include techniques such as no-tillage, minimum-reduced tillage and ploughing (Labreuche, et al., 2007). In Brittany, in 2006, 21 % of the maize and cereal production used conservation tillage technique (Chambres d’Agriculture de Bretagne; Arvalis, 2008). In 2010 in Brittany, non-tillage was used on 35 290 ha (< 2 % of the arable land), conservation tillage concerned 197 580 ha (13 % of the arable land) and conventional tillage was the main technique used on 748 440 ha (50 % of the arable land). Hence, this technique is advanced compared to others (Eurostat, 2013d).

The use of direct seeding increases in cases of rotation. For instance, in 2006, 88 % of the surface area where sunflower was replaced by wheat used direct seeding while this was the case for only 30 % of the surface in France (Agreste, 2008). According to the survey performed by the Agreste in 2006, 66 % of the farmers did not use specific machinery (Chambres d’Agriculture de Bretagne; Arvalis, 2008). The Chambre d’Agriculture of Brittany and technical institute Arvalis drafted reports to advice and guide farmers through the process of conversion of techniques (Chambres d’Agriculture de Bretagne; Arvalis, 2008).

Conservation tillage techniques are also beneficial in terms of time savings (Chambres d’Agriculture de Bretagne; Arvalis, 2008). Nevertheless, these techniques may require an important reorganisation of the field work and investment in machinery if needed. Thus, this practice is preferred on large farms (Agreste, 2008) considering the high investment for specific machinery. Moreover, this technique requires some dry days before seeding, which is not very compatible with the climate in Brittany. No-tillage can reduce the possibility to incorporate manure and increase the use of herbicides. It should be avoided in highly compacted areas.

**Consider climatic conditions before spreading manure**

Figure 49 presents the average precipitation in the region. It is not recommended and sometimes forbidden to apply manure or mineral fertilisers in autumn because of the risk of heavy rain episodes and because autumn does not correspond to a period of maximum water and nutrient uptake by crops. Although the precipitations are low in July and August, fertilisation during this period might result in the leaching of the nutrient applied.
In Brittany, according to the requirements of the Nitrates Directive, fertiliser application is subject to periods of prohibition. In the 5th NAP, specific periods were established depending on the type of manure and crops (DREAL Bretagne, 2014).

The farmers should avoid applying fertilisers before rainfall events to limit nutrient run-off. This aspect requires an accurate monitoring of the weather considering the frequent rainfalls in Brittany.

**Convert arable land to unfertilised grassland in areas at risk**

This measure concerns areas that are particularly sensitive to erosion or with high level of precipitation, more specifically in areas in the valley floor or near the catchment area of river basins with high nutrients surplus. This measure would be appropriate at the river basin of Lieue-de-Grève for instance (Dalmas, et al., 2010).

Converting arable land concerned by nutrient surplus, such as land with maize production, to unfertilised pasture would allow the nutrient pressure to be decreased due to a reduced fertilisation but also reducing the losses of nutrients to water bodies and to the sea. The pasture would capture the nutrients and reduce soil erosion.

This measure implies a loss of revenue. However, a study in Brittany from 1993 to 1997 on a panel of farmers showed that although the animal and cereals productivity decreased, the gross margin increased by 15-20 % due to the reduction of inputs and feed purchase when the share of grassland increases while the share of arable land with maize decreases (Journet, 2003). The nitrogen budget decreased compared to the baseline and the nitrogen losses by leaching were 2.5 times lower but it remained high and a more recent similar study would be necessary to validate the observed effects on the nutrient surplus.

While increasing the share of grassland, pasture management should still be carefully managed to avoid localised deposition of manure by grazing animals.

**Implement grass strips along hedgerow to limit nutrient run-off and erosion**

Grove used to be the main landscape in Brittany. However, it has progressively been reduced with the development of intensive agricultural activities (Bretagne Environnement, 2009b). Although the number of hedgerows still decreases (Bretagne Environnement, 2009b), a growing awareness has risen regarding the role of groves in the cultural identity of Brittany and their positive effect on erosion and reduction of nutrient run-off (Agro Transfert Bretagne, 2006; Ouvry, 2014). In 2010, Brittany accounted...
for 182 500 km of hedgerow (FDSEA Bretonnes, 2012). In areas concerned by the SDAGE, the maintenance and implementation of new hedgerows is mandatory (Chambre d'Agriculture Bretagne, 2013a). The extension of this measure would be relevant, in particular in ZAR. The stakeholder consultation of this project revealed that the landscape-related measures are considered are very promising to decrease nutrient losses. These type of measures are under study in the region (INRA Rennes, 2013).
Box 5 – Central Denmark case study - In brief

Central Denmark is affected by (1) marine and freshwater eutrophication due to surplus of nitrogen and phosphorus, especially in lakes potentially causing health effects, and (2) emissions of NH$_3$ to air causing acidification and health issues. Toxic algal bloom, particularly in coastal areas, may negatively affect biodiversity, human health and economic activities such as tourism and fisheries. High nutrient concentration in groundwater affects drinking water causing high economic costs to depollute waters to avoid effects on human health.

Central Denmark is characterised by intensive agriculture where a large amount of fodder is imported to feed the livestock (mainly pigs and cattle). This leads to a high volume of manure produced in comparison to the needs of the crops in the region. The main drivers of nutrient surplus in the Central Denmark region are the high livestock density, the high nutrient content of the excreted manure and the inefficient use of fertilisers leading to nutrient losses and emissions of ammonia and greenhouse gases. The losses of nutrients are enhanced by the soil texture of the region (mainly sand) and the significant erosion rate.

Costs are incurred from the losses of nutrients in the Central Denmark region and, more broadly, in Denmark. Thus, the marginal costs of reducing the coastal discharges of nitrogen and phosphorus from Danish wastewater treatment plants of € 2.83 to € 7.1 / kg N and € 4.84 to € 8.04 /kg P. Moreover, nitrogen volatilisation is an important source of ammonia emissions. It has been estimated that, between 1988 and 2009 a total of 387 538 t of ammonia emissions have been abated in Denmark, representing a cost of € 500-600 million. This has led to a reduction of 325 778 tonnes of N leaching to water.

For Central Denmark, a first set of measures focuses on reducing the quantity of nutrients created or applied through the adjustment of the feed quantities, the transfer of manure, the practice of organic farming, the reduction of the quantity of fertilisers applied and the use of best suited application techniques. Measures related to manure processing would help at easing its use and transport and valorise it if it is transferred to another farm or used to produce energy. The third set of measures focuses on reducing leaching to soil and water and emissions to air at different stages: storage, housing and land application on field or on pasture. The measures are the use of partly slatted floor, slurry tank coverage, the optimisation of the grazing intensity, the better implementation of buffer strips and the use of catch crops. Lastly, the measures aiming at improving soil quality and sol nutrient retention capacity such as appropriate tillage techniques and the use of perennial crops are also relevant to decrease the nutrient surplus in Central Denmark.

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The Central Denmark region (Midtjylland in Danish) is a region located in west Denmark on the Jutland peninsula (Figure 50).

Denmark is divided into four river basin districts and the Central Denmark region is located within the river basin district DK1. This river basin district is further divided into sub-districts. For this case study, the sub-districts of interest are: southern part of 1.2 (Limfjorden), 1.3 (Mariager Fjord), 1.4 (Nissum Fjord), 1.5 (Randers Fjord), 1.6 (Djursland), 1.7 (Arhus Bugt), northern part of 1.8 (Ringkebing Fjord) and 1.9 (Horsens Fjord) (Figure 51).

The total catchment area of the region is around 19 000 km$^2$, with Limfjorden, Ringkoebing and Rangers representing 75 % of the total catchment area (7 500 km$^2$, 3 477 km$^2$ and 3 250 km$^2$, respectively).
Central Denmark is one of the main agricultural regions of the country, comprising 30% of total Danish agricultural land in 2012. In the region, more than 60% of the regional territory was used for agriculture in 2010, which represents more than 790,000 hectares (Eurostat, 2015a). In 2013, Central Denmark generated 20% of the total national GVA and 31% of the national agricultural revenue.

The agriculture conducted in the region is intensive with a high share of organic farming compared to other studied regions. In 2010, out of the 2,100 organic farms registered in Denmark, 680 of them (representing 52,100 hectares, or 7% of the national agricultural land) were located in the Central Denmark region (Eurostat, 2015a).

In the region, arable land accounted for 92% of the UAA in 2012 (Eurostat, 2015a). The main crops are barley (240,000 hectares in 2013), wheat (147,000 hectares in 2013) and other cereals (c. 50,000 hectares) mostly consisting of grass and maize for silage (Eurostat, 2015b). In 2012, around 66% of the farms in Central Denmark were livestock farms and had an average size of 74 hectares. For almost all of the sub-districts, pigs are the main livestock kept followed by poultry and cattle. In the region, pig, broiler poultry and cattle production represented 4.3 million heads, 3.4 million heads and 502,000 heads respectively, including 173,000 heads of dairy cows (Eurostat, 2015b). The western municipalities of the Central Denmark region have a high concentration of livestock farms. The non-livestock farms are smaller, with an average size of 55 hectares and include a high proportion of small farms.

5.2.1 Notable impacts of nutrient surplus

This section focuses on the impacts of nutrient surplus that are specifically related to agricultural practices in the Central Denmark region. The main impacts are caused by nitrogen and phosphorus surpluses, which mostly affect water, soil and air. There is no evidence to suggest that potassium excess is a notable issue in this region. Most of the impacts on coastal waters are from nitrogen losses, while phosphorus losses affect mainly lakes.

The Central Denmark region is one of the most heavily N and P saturated regions in Denmark. In 2010, the surplus of nutrients for Denmark was 254,000 tonnes of N (97 kg N/ha UAA), 13,000 tonnes of P (5 kg/ha UAA) and 33,000 tonnes of K (13 kg/ha UAA) (Vinther & Olsen, 2011). Surpluses of P and K have been reduced by 75% since 2005 (Vinther & Olsen, 2011).

Data reported from the river basin districts in 2012 showed that the average N surplus at field level is 29 kg/ha for crop farms where no indication is provided on whether manure is used and 71 to 110 kg N/ha for livestock farm and crop farms using manure (Blicher-Mathiesen, 2013).

Description of the nutrient losses

The main environmental pressures linked to nutrients in Denmark are: pollution from agriculture and aquaculture leading to nutrient and organic enrichment of the water and the soil, pollution of water from urban wastewater and hydromorphological pressures leading to altered habitats (European Commission, 2012d). The overview of the river basin sub-district pressures shows that whilst nitrogen is mainly an issue for groundwater and coastal water, eutrophication of lakes and algae blooms are mainly caused by phosphorus (Danish Nature Agency (Naturstyrelsen), 2014).

Denmark has decided to apply the action programme implementing the Nitrates Directive to its whole territory. Moreover, in the river basin district DK1 (where the region is located) there are 76 protected areas for birds’ species and 23 protected areas for shellfish, and 169 habitats have been designated as sensitive under the Habitats Directive.
**Impacts of nitrogen losses**

**Nitrogen load in freshwater**

**Surface water** - A total of 495 km of rivers are directly affected by an excess of nitrogen, almost half of which are located in the Randers Fjord (205 km) (Danish Nature Agency (Naturstyrelsen), 2014). N levels in watercourses in Central Denmark vary across the region (Figure 52), It can be observed that watercourses in the western part of Central Denmark generally have a lower concentration of nitrogen than streams on the eastern part (i.e. 2-4 mg/L of N (9-18 mg NO$_3$-L) in eastern part and 2-6 mg/L of N (9-27 mg NO$_3$-L) in the western parts). The highest concentration exceed 8 mg/L of N (35 mg NO$_3$-L) and is found only at one monitoring station located in the north east part of Central Denmark (Figure 52).

According to the report on the implementation of the Nitrates Directive carried out by the European Commission, in Denmark, less than 1 % of surface water stations exceeded the concentration of 50 mg/L of nitrate and around 15 % exceeded the concentration of 25 mg/L of nitrate during the 2008-2011 reporting period. It also highlighted that 75 % of surface water measurements of nitrate concentrations were lower than the previous reporting period (2004-2007) (European Commission, 2013b).
A review of the sub-districts was conducted during the finalisation of the RBMP in 2014\(^{113}\) and it identified nutrient surplus in all of them. In 2012, an analysis of the water bodies found that 53% of the watercourses were at risk of not meeting environmental objectives initially set for 2015, with the most affected in the Limfjorden and Djursland sub-districts. Updated information will be included in the 2015-2021 RBMP that are expected to be published no later than 22 December 2015 (Natyrstyrelsen, 2014).

Lastly, whilst a significant proportion of watercourses are failing to meet their environmental objectives because of excess nutrients, the main challenges for rivers stem from physical conditions (European Commission, 2013b). In the western part of Central Denmark, the volume of rainwater is transported for a considerable distance through reducing (anoxic) aquifers before it reaches the stream catchments and ultimately the estuary (Nordemann & al, 2013). In the eastern part of the region, a large part of the water flow with its nitrogen content is transported through the upper aquifer or through drain pipes without passing anoxic zones.

**Groundwater** - On average, 81% of the groundwater is affected by an excess of nutrients, with Mariager Fjord, Nissum Fjord and Ringkebing Fjord being the most affected (Danish Nature Agency (Naturstyrelsen), 2014). Nitrogen leaching from arable land is more important in the western part of the Central Denmark region with on average 26 - 30 kg N/hectare leached annually into the groundwater than the eastern part (on average 15 - 25 kg N/hectare) (Figure 53). A central area of Central Denmark (Randers Fjord) still presents high level of nitrogen leaching in groundwater with 31-40 kg N/hectare.

\(^{113}\) Denmark submitted its river basin management plan to the European Commission in 2012 (instead of 2009 as required). Shortly after submission the plans were annulled by Denmark as its Courts found that the consultation period had been too short. Denmark has approved the modified plan in October 2014 and re-submitted them to the European Commission.
In Denmark, qualitative data for wells and data on abstraction of groundwater must be reported by the counties to the Geological Survey of Denmark and Greenland (GEUS). According to the report on the implementation of the Nitrates Directive for the period 2008-2011, almost 20% of the total groundwater reporting stations exceeded the concentration of 50 mg/L of nitrate and nearly 35% exceeded 25 mg/L of nitrate (European Commission, 2013b). This represents a small increase compared to the 2004-2007 reporting period for groundwater stations exceeding 50 mg/L nitrate but an almost 5% increase of monitoring stations that report groundwater exceeding 25 mg/L nitrate. During the 2008-2011 period, 38% of the groundwater station results showed increasing levels of nitrates.

In addition, the latest review of data by GEUS found that the nitrates concentration in the shallowest groundwater is decreasing in 62% of the monitoring points. However, only 22% of the monitoring points with deeper groundwater showed a similar significant decreasing trend. Overall, a decrease of nitrate concentration in groundwater was identified in 41% of the groundwater monitoring points and an increase in another 41% of the monitoring points. It is suggested that the decrease in nitrate in the shallow groundwater is due to a reduction in nitrogen leaching (Geological survey of Denmark and Greenland, 2012).

One of the characteristics of the Danish water supply is that the quasi totality of the water supply for domestic use and drinking water is abstracted from groundwater. Over recent decades many waterworks have been closed, forcing to drill deeper or forced to buy their water from neighbouring water suppliers due to pollution (Geological survey of Denmark and Greenland, 2012). Nitrate pollution of groundwater due to agriculture has led to the closure of many minor water works based on shallow aquifers (Dalgaard, et al., 2014).

**Eutrophication of marine water**

The eastern part of the Central Denmark region is bordered by the Baltic Sea for which the Helsinki Convention regulates its protection from all sources of pollution. The latest assessment of the Baltic Sea found that the quasi-totality of the Sea was eutrophied. In 2010, Denmark was responsible for 53 429 000 t of total normalised water and airborne input of nitrogen in the Baltic Sea, representing 6.7% of the total normalised nitrogen inputs. The waterborne nitrogen inputs accounted for 71% of the total inputs (HELCOM, 2013). The specific contribution of the Central Denmark region to these inputs is not available. Leaching from freshwater run-off to the Danish coastal waters varies considerably from year to year. For example, in 2012, run-off was 9% higher than the average for the period 1990-2011 (Hansen, 2013). It was estimated that in 2012, run-off to marine water contained 2 600 t of phosphorus and 59 600 t of nitrogen. The run-off from land has decreased by about 50% since 1990 due to reduced leaching from agriculture and improved wastewater treatment (Hansen, 2013). In 2012, the assessment of coastal waters found that the nitrogen concentrations were very low for the fjords, coasts and open inland waters. It concluded that both nitrogen and phosphorus concentrations in marine water show downward trends since 1989, with a slight decrease for nitrogen since 2002 (Hansen, 2013).

In the Baltic Sea there are usually two algae bloom events every year. The first one occurs in spring between March and May and is dominated by diatoms and dinoflagellate algae. It is generally not toxic to human. However, the blooms give the water a red / brown appearance, which can have a visual impact in recreational areas. The second blooming event is a cyanobacterial bloom occurring during the summer period. These are common occurrence in the Baltic Sea and can cover up to 100 000 km² (HELCOM, 2013).

Oxygen deficiency in marine water is a natural phenomenon, which is increased by eutrophication and climate change. Eutrophication and the associated algae bloom provide the basis for an oxygen deficiency beyond a natural level when oxygen consumption near the seabed is greater than the oxygen supply; this typically occurs during still and warm periods (Hansen, 2013).
Eutrophication and related acute oxygen depletion events have been a regular phenomenon in Danish estuaries. A well-documented example is from 1986 when fishermen caught dead Norwegian lobsters in the Kattegat (Baltic Sea). Between 1981 and 2002, a total of 21 macrozoobenthos killing events were recorded in the coastal Danish waters, of which 12 were located in the Central Denmark region. These events were mostly triggered by oxygen depletion and one toxic algal bloom (Nutrient Eutrophication in Danish Marine Waters, 2004).

Benthic animals are also affected by prolonged periods of severe oxygen deficiency. These animals represent a food resource for the fish population (Hansen, 2013). In the Central Denmark region fluctuation in the number of marine species have been influenced by oxygen deficiency and eutrophication. Table 24 below demonstrates these fluctuations between 1999 and 2001 in the region. In some other parts of the region, the number of species has increased; this is not necessarily a positive outcome as some opportunistic species may develop to the detriment of others.

Table 24 – Fluctuation in number of coastal species in the Central Denmark region between 1999 and 2001

<table>
<thead>
<tr>
<th>Coastal areas</th>
<th>1999</th>
<th>2000</th>
<th>2001</th>
</tr>
</thead>
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<tr>
<td>Horsens Fjord</td>
<td>24</td>
<td>24</td>
<td>40</td>
</tr>
<tr>
<td>Ringkobing Fjord</td>
<td>22</td>
<td>22</td>
<td>17</td>
</tr>
<tr>
<td>Nissum Fjord</td>
<td>29</td>
<td>33</td>
<td>28</td>
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<tr>
<td>Nissum Bredning</td>
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<td>57</td>
</tr>
<tr>
<td>Mariager Fjord</td>
<td>14</td>
<td>28</td>
<td>26</td>
</tr>
</tbody>
</table>

Source: (Nutrient Eutrophication in Danish Marine Waters, 2004)

Estuaries are generally the most nutrient-loaded marine areas receiving the vast majority of freshwater run-off in Denmark (Nordemann & al, 2013). Agriculture is responsible for 70 % of the total nitrogen loadings in coastal waters in 2007-2011 (Dalgaard, et al., 2014). A recent report studied the relationship between the agricultural nitrogen surplus and the nitrogen load from diffuse sources at 10 Danish estuaries of which 6 are in the Central Denmark region. The authors found a close relationship between the nitrogen load from diffuse sources (total nitrogen load excluding point sources) and the agricultural nitrogen surplus in the catchment areas. The slope of relationship between nitrogen loads as a function of nitrogen surplus varies between catchment areas, which is probably due to catchment-specific variations in nitrogen removal in groundwater as a consequence of varying geology, differences in nitrogen retention in surface waters as well as time lags in the groundwater flow (Windolf, et al., 2012).

The total area of the Danish estuaries affected by oxygen deficiency varies from year to year, and the locations may also change. However, there are a number of areas that are exposed to recurring (annual) oxygen deficiency. In the Central Denmark region, these areas are the estuaries Mariager Fjord and Limfjorden. Most of the oxygen deficiency events occur between August and September and have disappeared completely by the end of October.

In 2012, the combination of windy weather during the typical oxygen deficiency period (late summer and autumn) and normal bottom water temperatures, resulted in relatively normal oxygen conditions (Hansen, 2013). In addition, a low concentration of nutrients contributed to a relatively low algae production in the water column and therefore low oxygen consumption at the seabed during decomposition. Together, these factors resulted in a limited oxygen deficiency in 2012.
The most pronounced oxygen deficiency events were observed in the Limfjorden estuary located north-west of the Central Denmark region. Here the oxygen deficiency peaked in late August, where 12% of the estuary area was affected – of which 7% severely\textsuperscript{114}. In late August, production of hydrogen sulphide was identified at several sampling locations in the estuary. Anoxic conditions were also observed at the Mariager estuary located north-east of the Central Denmark region. In Aarhus bay, oxygen deficiency was registered in September (Hansen, 2013).

The long-term development in oxygen concentrations is influenced by changes in sea temperature, which has risen 1°C during the last 40 years and is expected to increase further as a result of climate change. An increase in sea temperature itself means reduced oxygen content in seawater and thus an increased risk of oxygen deficiency (Hansen, 2013).

\section*{Ammonia (NH$_3$) emissions to air}

In 2010, Denmark was exceeding its emission ceiling of 69 kt ammonia (NH$_3$) to air set by the EU under the National Emission Ceilings Directive. In 2011, it reported a decrease in its total ammonia emissions to reach 69 kt of NH$_3$ emitted, however these emissions have increased again to reach 70.8 kt in 2012 (EEA, 2012d). The majority of these emissions (97\%) originate from the agricultural sector. There is no specific data for ammonia emissions from the Central Denmark region.

Information presented in the 2009-2015 RBMP suggested that agriculture was responsible for a very large share of N deposition in all sub-districts, in particular for Djursland (94\% of N is from agriculture). The most important N deposition was reported for the largest sub-districts of Randers and Ringkebing (respectively 4 600 and 4 400 tonnes of N in total). Data were available for 6 out of the 8 sub-districts, for which 12 375 t of N out of 14 580 t were deposited from agriculture (Danish Nature Agency (Naturstyrelsen), 2014).

The main source of emissions of NH$_3$ from agriculture is livestock manure (82\%) and more specifically emissions from housing and during field application. Other contributions come from excretion by livestock grazing on pastures (3\%). Emissions from the use of mineral fertiliser and crops contribute to 7\% and 8\% respectively of the agriculture emissions (Nielsen & al, 2010). In the Central Denmark region, NH$_3$ emissions from volatilisation of nitrogen are particularly important, with 16 to more than 25 kg/N/hectare annually emitted from arable land (Figure 54).

\begin{footnotesize}
\textsuperscript{114} In Denmark, oxygen deficiency is defined as oxygen concentration in the water below 4 mg/L. The deficiency is rated as severe when the concentration is below 2 mg/L.
\end{footnotesize}
Nitrous oxide ($N_2O$) emissions to air

In 2010, a total of 19 kt of $N_2O$ was emitted from Denmark; of which 17.3 kt was from the agriculture sector (UNFCCC, 2013). In 2011, $N_2O$ emissions from agricultural soils accounted for 53\% of emissions from the agriculture sector with the highest emissions in the south-east part of the Central Denmark region (Figure 55).

\footnote{The NCU is based on the NUTS2 regional classification and integrate soil and slopes characteristics.}
The largest sources of emissions were manure and fertiliser application. Another large source was indirect N₂O emissions originating from leached nitrogen (Nielsen & al, 2010). When nitrate leaches through anoxic aquifers it is converted by biological or chemical denitrification to N₂O or gaseous nitrogen (N₂) (Nordemann & al, 2013).

**Impacts of phosphorus losses**

In the period 2006-2007 to 2010-2011, the phosphorus surplus measured as farm budgets declined by almost 50%. Despite the significant reduction in farm phosphorus surplus a reduction in phosphorus concentrations in watercourses was not observed. This is because phosphorus binds to soil particles and enters watercourses mainly as particle-bound phosphorus, as a result the lag time is important. Since livestock farms still show phosphorus surplus, the soil’s ability to bind phosphorus in these areas will gradually decrease with subsequent risk of increased phosphorus losses (Børgesen & al, 2013). Phosphorus losses are a major cause of eutrophication in lakes and surface freshwater.

**Phosphorus load in freshwater**

Evidence suggests that at a national level the quantity of phosphorus entering streams was in the range of 0.20 - 0.47 kg P/ha for the period 1990-2011. Out of this, the majority binds to sediments of the soils or enters into the deeper soil layers. The remainder dissolves in the watercourses and lakes (Blicher-Mathiesen, 2013).

The share of phosphorus surpluses from agriculture varies at the sub-district levels between 28% and 50% (Danish Nature Agency (Naturstyrelsen), 2014). Other pressure identified are linked to municipal and industrial activities (all basins), use of pesticides (e.g. Arhus Bugt and Ringkebing Fjord), fisheries (e.g. Nissum Fjord), sewage sludge (e.g. Mariager Fjord) and ochre pollution (e.g. Limfjorden, Mariager Fjord and Nissum Shord). A total of 27.2 tonnes of P were deposited from agriculture in three sub-districts. In two sub-districts whilst no data on the total P deposition was available, it was indicated that agriculture is responsible for 40% of the phosphorus deposited. It is important to highlight that the highest content of phosphorus is found in streams that receive urban wastewater (e.g. Randers Fjords received 100 tonnes of P; less than half was from agriculture and the rest from wastewater) (Danish Nature Agency (Naturstyrelsen), 2014) (Figure 56). However, the concentration of phosphorus in watercourses located in agricultural catchments or with significant point source discharges was, in 2012, on average 2-3 times higher than the levels measured in natural streams located in areas without agriculture (Nordemann & al, 2013).
Lakes in Central Denmark are particularly vulnerable to phosphorus as the nutrients accumulate at the bottom and get released gradually. On average, 53% of the lakes were at risk of not meeting environmental objectives set for 2021 (Danish Nature Agency (Naturstyrelsen), 2014). Lakes are mostly affected by the accumulation of nutrients in their beds (phosphorus) and the excessive inputs from land, in particular in the Randers Fjord, Djursland, Aarhus Bugt and Horsens Fjord sub-districts (Danish Nature Agency (Naturstyrelsen), 2014). For the period 2008-2011, 88% of the Danish lake were eutrophic or hypertrophic (European Commission, 2013b).

Analysis of the monitored data shows that an important part of the phosphorus in the upper groundwater is bound to organic elements (and not dissolved in the groundwater). As a result, transport from agriculture to surface waters through the groundwater may be more significant than the current evidence indicates (Geological survey of Denmark and Greenland, 2012).

Phosphorus load in coastal waters

Concentrations of phosphorus in Danish estuaries and coastal areas are significantly lower than in the early 1990s, mainly due to improved phosphorus removal in wastewater treatment plants (Nordemann & al, 2013). Most importantly, this has caused an increase in potential phosphorus limitation in estuaries, where phosphorus may currently limit growth of algae during 50-60% of the growing season compared to only c. 20% around 1990.

In 2010, the normalised total water and airborne input of phosphorus to the Baltic Sea was 32 248 tonnes. Denmark is responsible for 5.3% of the phosphorus deposited, exclusively through water (HELCOM, 2013). The specific contribution of the Central Denmark region to these inputs is not available.
5.2.2 Causes of nutrient losses

**Farming systems and agricultural practices**

Several factors related to the farming systems and other agricultural practices have a high effect on nutrient losses.

**Intensive livestock breeding leading to high amount of manure produced that may locally exceed the crop need in the region** – Central Denmark is characterised by intensive agriculture, where a large amount of fodder is imported to feed the livestock (mainly pigs and cattle). The average livestock density of the Central Denmark region is high with 0.9 LU/ha (which corresponds to 1.95 LSU per utilised agriculture area which is slightly higher than the national average of 1.8 LSU/UAA) of farmland. The high livestock density is one of the main sources of nutrients surplus. The Limfjorden, Nissum and Ringkebing sub-districts have the highest livestock density with 1.3 LU per hectare (2.8 LSU/ha UAA) (Danish Nature Agency (Naturstyrelsen), 2014).

Recent changes in farm structures seem to indicate further concentration of the agricultural production on fewer farms. In 2010, the number of farms of 5-10 hectares had decreased by 2.2 % when compared to the figures from 2009; whereas, over the same period, the share of farms of over 200 hectares rose by 5.5 %, resulting in fewer but larger farms. The structural changes also affect the livestock numbers and types with a distinct tendency toward specialisation. The latest data found (from 2008) indicated that only 3 % of the Danish farms reared both cattle and pigs (Danish Agriculture and Food Council, 2012). Over the past five years, the number of cattle has slightly increased whereas the number of finishers has declined due to the significant increase in the export of live piglets during the same period. It is worth mentioning that this export is still increasing, rising by 75 % between 2007 and 2011 and representing 8 million piglets exported in 2011 (Børgesen & al, 2013).

Denmark is among the world’s largest pig meat exporters; 77 % of the pigs are kept on farms with herds of more than 2 000 pigs. In 2010, the total pig population was 13 million, which represented an increase of 6.5 % from the 2009 data (Danish Agriculture and Food Council, 2012). However, it is important to note that as pigs are reared and slaughtered at around 4 months old, actual Danish production of pigs for slaughtering was 21.4 million in 2011 (Børgesen & al, 2013).

**High mineral fertiliser use** – According to the Nitrates Action Programme, Danish farmers are required to fertilise up to 15 % below the "economic optimum".  

<table>
<thead>
<tr>
<th>Table 25 – Mineral fertiliser use in Denmark in thousands tonnes of N</th>
</tr>
</thead>
<tbody>
<tr>
<td>2005</td>
</tr>
<tr>
<td>---</td>
</tr>
<tr>
<td>Fertilisers bought</td>
</tr>
<tr>
<td>Fertiliser used</td>
</tr>
</tbody>
</table>

Source: (Blicher-Mathiesen, 2013)

At national level, it is interesting to note that the consumption of mineral fertilisers has decreased from 394 000 tonnes N in 1990 to 185 000 tonnes N in 2012 (Blicher-Mathiesen, 2013). Over the same

The Danish Ministry for the Environment applies what is termed an economic optimum principle, under which there are compulsory norms for the amount of nitrogen applied to crops. The norms are set by deducting 10 to 15 % from the economic optimum. The economic optimum is a theoretical rate meaning that the economic gain from the crop increase achieved by applying fertiliser cannot increase further, i.e. if more fertiliser were applied, from that point onwards, the cost of doing so would exceed the economic value of the increased crop.
period, the use of nitrogen from manure has slightly decreased from 244 000 tonnes to 228 000 tonnes (Table 25).

**Tillage erosion that may lead to high phosphorus loss** – Intensive tillage and mouldboard ploughing are both part of the modern Danish agricultural system (Schjønning, et al., 2009). However, these have been shown to trigger erosion from tillage, which has caused severe soil redistribution on arable land (Schjønning, et al., 2009). Such redistribution is considered to present a substantial long-term threat to soil productivity. Modelling has found that erosion rates due to tillage frequently exceed 20 ha\(^{-1}\)yr\(^{-1}\) on eroding sites within fields, this means that tillage is 10 to 100 times as erosive as water erosion, and much more widespread. As a result, it is expected that farmers bear a substantial cost due to loss of productivity on eroded sites or due to the implementation of fertility enhancing measures.

**Environmental conditions**

Natural factors modulate the effects of farming practices and especially fertiliser inputs.

**Leaching and run-off increased by local soil texture** – Denmark, and the Central Denmark region in particular, is covered by very sandy soils in the western part (Figure 57) with a high SOM content. Most of the eight river basin sub-districts are characterised by sandy and / or loamy soils.

![Soil types map](Figure 57 – Soil texture map for Denmark)

Whilst nitrate leaching is generally more important in coarse textured soils, the N load to the estuaries may be lower for the coarse textured catchment areas. This is because the fine textured soils in Denmark are generally pipe drained; therefore leached nitrate is led to the surface waters relatively fast and with less opportunity for denitrification. In contrast, the transport route is longer in sandy soils, as leached nitrate passes through the groundwater, where anaerobic conditions stimulate denitrification of nitrate to N\(_2\)O and N\(_2\). Figure 58 illustrates these differences with some actual numbers for N input, leaching and N load to surface waters for sand (left hand side) and clay soils (right hand side).
Soil organic matter (SOM) content – Denmark is the most intensely cultivated country in Europe (Norfelft, 2011) which relies on tillage. Each time the soil is tilled, it is also aerated. However, the decomposition of organic matter is an aerobic process that is stimulated by oxygen. So each time the soil is tilled, the residues are incorporated in the soil with air and come into contact with micro-organisms which accelerate the nutrient and carbon cycle. As a result, the decomposition is faster and the humus created is less stable leading to higher ammonia and CO$_2$ release and a reduction in SOM content. (FAO, 2007a).

During a review of soil quality in Denmark, it was found that the SOM of Danish soils had generally declined in the past decade (Schjønning, et al., 2009), and a low SOM content of the soil means that fewer nutrients are available for the crops. It was also found that the SOM of a soil plays a key role in the soils ability to fragment when mechanically disturbed.

Erosion rate – Due to the low elevation of the land and low rainfall, the erosion risk in Denmark is generally perceived to be low (Schjønning, et al., 2009). It occurs on most soil types, in autumn and winter, especially after prolonged periods of rainfall, after snowmelt or rainfall on frozen soil. However, erosion leads to a loss of soil resource and can cause eutrophication of surface waters. Erosion is a significant source of phosphorus losses (Bergstrom, et al., 2007). In Denmark, the most severe effects from erosion are loss of fine-textured material, organic matter, nutrients and available water capacity. Deposits transported by rain on sites with sandy or weakly structured soils are often depleted in clay and in organic matter. The impacts of erosion on the loss of productivity of the soil are difficult to estimate as other factors such as climate, land geological properties and its management are important.

pH of the soil – Danish soils’ pH ranges between 4.0 and 8.3 and are mostly acidic (alkaline from pH7). Acidic soils are less prone to ammonia volatilisation so acidic soils will suffer less from further losses of ammonia than alkaline. The soil management has an impact on the variation of pH of the soil; in Denmark farm soils are regularly limed to maintain a favourable pH of the plough layer.

Climate – Central Denmark has relatively low mountains area with elevation ranging between 7m below and 171m above mean sea level (Deng, et al., 2014). It has a temperate climate, with a mean
temperature of 0°C in winter and 16°C in summer. Mean annual precipitation ranges from 500mm in the Great Belt region (east) to 800 mm in the western part of Central Denmark (Deng, et al., 2014). This results in further erosion from rainwater and water logging for areas with highest precipitations which in turn lead to nutrient run-off and leaching.

5.2.3 Costs of environmental and health effects

5.2.3.1 Socio-economic description of the region

In 2014, the Central Denmark region had a total of 1 278 840 inhabitants (Statistics Denmark, 2014a) dispersed over a territory of 13 005 km$^2$ (Statistics Denmark, 2014b).

The main economic activity of the region is manufacture and trade. Agriculture and fisheries provided 2 % of the region’s gross value added representing c. € 921 million in 2013 (Eurostat, 2015c). Central Denmark is Denmark’s second most important source of revenue from agriculture, generating almost 31 % of the total national gross value added (GVA) from agriculture in 2013 (Eurostat, 2015c). In 2010, the Danish cooperatives reported a turnover of DKK 49.5 billion (c. € 6.4 billion) for the dairy sector and DKK 53.5 billion (c. € 6.9 billion) for the meat sector. The value of Danish agricultural exports has been increasing to reach € 9.7 billion in 2011. The agriculture and related goods sector as a whole represent 20 % of the total Danish commodity exports. Moreover, the EU export subsidies have been phased out and account for only 0.2 % of the value today (this compares to 12 % in 1990) (Danish Agriculture and Food Council, 2012).

5.2.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in Central Denmark, a wide range of literature was reviewed. Costs were identified and are presented below, based on the classification of economic damages caused by environmental impacts presented in Annex 14.

It is important to note that all cost data reported in this section are taken from the primary studies found in the literature and expressed in the value of the year when the study was conducted.

Clean up and restoration costs (CRC)

Algae removal

Whilst no specific information was found for Central Denmark, algae blooms are regularly reported in Danish coastal waters, especially during the spring and summer. A project from the South Baltic programme tested several algae removal techniques and their applicability to the Baltic region, including the Central Denmark region. It found that the costs were similar for the different methods of removal of algae on the beach. The main differences were related to the efficiencies and applicability of the methods. For removal of algae in the seawater, a cost of € 87 per hour for the rental of the equipment was provided (Wetland, Algae Biogas, 2009a).

The project also included estimated costs of transport of the algae. Costs depended on the distance transported but also on the origin of the algae. For example, to transport four months’ worth of algae collected in small Baltic Sea municipalities, corresponding to 5 640 tonnes, from the beach to the collection area (20 km away) costs an estimated € 25 000 (around € 4.4 per tonne of algae). Transport by barge of algae collected from the sea is more costly; € 520 for the removal of 22 tonnes of algae (around € 23.6 per tonne of algae). The costs associated with unloading and packaging the algae, are an additional element to take into account with € 42 500 for 5 640 tonnes of wet algae (€ 7.5 per tonne of algae) (Wetland, Algae Biogas, 2009b).
In many Danish lakes, phosphorus is the most important limiting nutrient for phytoplankton growth; in other lakes phosphorus and nitrogen are equally important, or change in relative importance with the seasons. No estimate was available for the Central Denmark region specifically, but almost 40% of the Danish lakes are suitable candidates for lake restoration initiatives aimed at improving water quality. For example, the project of restoration of the Lake Stubbe was funded from the State Nature Management body and covered the costs of the technical consultancy assistance for the mass removal of planktivorous fish. This lake is located near the eastern coast of Central Denmark and its example represents the most common restoration methods applied in Denmark (Country of North Jutland, 2006).

Information about another example of lake restoration involving different methods was also identified. Whilst this example is not in the Central Denmark region, it is relevant considering that the impacts were similar to those faced in the Central Denmark region. Lake Fure was threatened by high levels of phytoplankton biomass, high biomass of planktivorous fish, lack of oxygen and high levels of phosphorus loading. The final report includes information on the different costs involved. Different steps were taken in order to restore the lake, however the two main actions were the removal of fish and the establishment of an oxygenation unit. Through a public forum, dissemination steps were taken and public meeting were organised. Overall, more than 2,500 people have attended presentations on the restoration activities and/or public meetings and lectures (Frederiksborg County, 2006).

Reduction of nitrogen losses

Nitrogen volatilisation is an important source of ammonia emissions. It has been estimated that, between 1988 and 2009 a total of 387,538 tonnes of ammonia have been abated in Denmark, representing a cost of € 500-600 million. This has led to a reduction of 325,778 tonnes of N leaching to water. Indicative costs are € 1.3–1.5/kg NH$_3$-N per kg of reduced NH$_3$-N and € 1.5–1.8 per kg of reduced leached N (Gyldenkaerne, 2012).

Nitrogen removal

According to article 2.a of the Act on Waste water price$^{117}$, waste water treatment plants are charged based on their actual nutrient (nitrogen and phosphorus) emissions. This provides an incentive for the treatment plants to reduce emissions (DANVA, 2015). According to this legislation the water treatment contribution, paid by private companies, is composed of a fixed and a variable part. The fixed part is set at € 53.7 (500 kr) and the variable part is calculated according to the actual water consumption. In 2014, the average price of water, including waste water and sewage treatment costs as perceived by consumers, was € 8.40 per m$^3$, out of which 25% (€ 2.1 per m$^3$) represent taxes on drinking water and wastewater (DANVA, 2015). In addition, a green tax for extraction of drinking water is applied. It amounted to € 0.9/m$^3$ in 2013. Finally a drinking contribution tax of € 0.1/m$^3$ is applied until 2017. Gren et al. (2000) estimated the marginal costs of reducing the coastal discharges of nitrogen and phosphorus from Danish waste water treatment plants of € 2.83 to € 7.1 / kg N and € 4.84 to € 8.04 /kg P. At the catchment level in the Randers Fjord, the study suggests that the cost of nutrients removal was € 300 million for the period 1995-2005 (Atkins & Burdon, 2006).

$^{117}$ Bekendtgørelse nr. 633 af 07/06/2010 af lov om betalingsregler for spildevandsforsyningsselskaber
https://www.retsinformation.dk/forms/r0710.aspx?id=131457
Table 26 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Expenditure for clean-up and restoration purposes</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Scale</th>
<th>Year</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Algae clean-up on the beach by Grip claw loader, Power rike and Beach Cleaner</td>
<td>€ 600 renting/hour for all 3 machines incl. drivers + fuel costs (60 l for 5 hours work)</td>
<td>South Baltic Region</td>
<td>2009</td>
<td>(Wetland, Algae Biogas, 2009a).</td>
<td></td>
</tr>
<tr>
<td>Algae clean-up in the seawater (DM Truxor)</td>
<td>€ 87 per hour including driver</td>
<td>South Baltic Region</td>
<td>2009</td>
<td>(Wetland, Algae Biogas, 2009a).</td>
<td></td>
</tr>
<tr>
<td>Transport of algae (labour cost € 130/hour)</td>
<td>€ 4.4-23.6 / tonne algae</td>
<td>South Baltic Region</td>
<td>2009</td>
<td>(Wetland, Algae Biogas, 2009b)</td>
<td></td>
</tr>
<tr>
<td>Unloading and packing algae (labour cost € 100 / hour)</td>
<td>€ 7.5 / tonne algae</td>
<td>South Baltic Region</td>
<td>2009</td>
<td>(Wetland, Algae Biogas, 2009b)</td>
<td></td>
</tr>
<tr>
<td>Point source N reduction</td>
<td>€ 0.97 billion (€ 52 /kg N)</td>
<td>Denmark</td>
<td>2004</td>
<td>(Windolf, et al., 2012)</td>
<td></td>
</tr>
<tr>
<td>Nitrogen surplus reduction</td>
<td>€ 2.571 billion (€ 13 /kg N)</td>
<td>Denmark</td>
<td>2004</td>
<td>(Windolf, et al., 2012)</td>
<td></td>
</tr>
<tr>
<td>Restoration of Lake Fure (including biomanipulation and oxygenation)</td>
<td>€ 2,157,233</td>
<td>Lake Fure</td>
<td>2006</td>
<td>(Frederiksborg County, 2006).</td>
<td></td>
</tr>
<tr>
<td>Reduction of ammonia emissions from nitrogen</td>
<td>€ 1.3 –1.5 /kg NH$_3$- N</td>
<td>Denmark</td>
<td>2009</td>
<td>(Gyldenkaerne, 2012)</td>
<td></td>
</tr>
<tr>
<td>Reduction of leached N</td>
<td>€ 1.5 –1.8 /kg N</td>
<td>Denmark</td>
<td>2009</td>
<td>(Gyldenkaerne, 2012)</td>
<td></td>
</tr>
<tr>
<td>Green tax for water and abstraction charge</td>
<td>€ 1/m3</td>
<td>Denmark</td>
<td>2013</td>
<td>(DANVA, 2015)</td>
<td></td>
</tr>
<tr>
<td>Cost of water including waste water and sewage treatment costs</td>
<td>€ 8.40 /m3</td>
<td>Denmark</td>
<td>2013</td>
<td>(DANVA, 2015)</td>
<td></td>
</tr>
<tr>
<td>Marginal costs of reducing the coastal discharges of nitrogen from Danish waste water treatment plants</td>
<td>2.83 to € 7.1 / kg N</td>
<td>Denmark</td>
<td>2000</td>
<td>(Atkins &amp; Burdon, 2006)</td>
<td></td>
</tr>
<tr>
<td>Marginal costs of reducing the coastal discharges of phosphorus from Danish waste water treatment plants</td>
<td>€ 4.84 to € 8.04 /kg P</td>
<td>Denmark</td>
<td>2000</td>
<td>(Atkins &amp; Burdon, 2006)</td>
<td></td>
</tr>
<tr>
<td>Costs of reducing the coastal discharges of nutrients from waste water treatment plants at catchment level</td>
<td>€ 300 000 000</td>
<td>Randers Fjord</td>
<td>2000</td>
<td>(Atkins &amp; Burdon, 2006)</td>
<td></td>
</tr>
</tbody>
</table>
Use value damages (UVD)

Eutrophication is not only an environmental problem with significant costs for the local and regional authorities, but also a threat to a number of economic sectors such as tourism or fishing in the Central Denmark region.

For other activities, very few quantitative data sources were found in the literature regarding use value damages. Most of the costs in this category have not been systematically estimated at the time of this study. In the present study, we are only able to provide some indicative damage estimates for this category (Table 27).

The impact of eutrophication on fisheries is important. The main Danish fishing areas are located in the North Sea, Skagerrak/Kattegat and the Baltic Sea. Cod, herring, mackerel and flatfish species are the main fishery species for human consumption in Danish waters; accounting for more than 60 % of the value of fish landings. Fishing for Norway lobster in the Kattegat and for blue mussels in Limfjorden is of significant local importance (FAO, 2012).

A monitoring program for toxins in the seawater has been established by the mussel industry. The total costs of the programme for 1997 were DKK 2.5 million (approximately € 335 000). This was split between DKK 0.7 million (approximately € 91 000) for the analysis of algae and DKK 1.8 million (approximately € 234 000) for the analysis of algal toxins in shellfish. The sampling activity is conducted directly by the fishermen however there is no further cost data available for this (Anderson, 2001).

In the context of recreation and tourism, eutrophication can have a range of adverse impacts, such as potential health risks associated with the exposure to toxic algal blooms and loss of tourism revenue due to nuisance caused (e.g. visual or smell). When the algae reach the coastline, the algae decomposition give rise to a bad odour that is damaging for tourism, especially camping sites and other recreational areas which depend on the coast to attract tourism (University of Kalmar, 2008). In 2005, the Oland area suffered from a particularly harmful algal bloom. It was estimated that this created a loss of € 10 million. These losses include revenues from tourism but also reputational damages. A survey conducted of Danish tourists revealed that 55 % of them had algal blooms in mind when planning their vacation, and 82 % of them were likely to be influenced by media reporting on algal blooms. Furthermore, 81 % of those questioned believed that algal blooms were harmful to humans (University of Kalmar, 2008).

Tourism is a source of revenue for the Central Denmark region, which recorded 16 % of total Danish overnight stays in 2011, representing 4.4 million nights. This makes Central Denmark the third most popular region in Denmark. Denmark is renowned for its nature and landscape. It was estimated that for 23 % of EU (27) citizens, the choice of Denmark as a destination was motivated by its nature (Bastis Tourism, 2011). In 2011, the tourism sector has made a contribution of 33 billion DKK (c. € 4.3 billion) to the national GDP, representing 1.8 % of total GDP. Moreover it is expected that the contribution of the tourism sector to the national GDP will increase by 3.4 % per annum between 2012 and 2022, equivalent to DKK 47.8 billion (c. € 6.3 billion) in 2022 (Bastis Tourism, 2011). As a result, the potential negative impact of eutrophication on tourism cannot be ignored and could become more significant over the next ten years.
Table 27 – Use value damages of eutrophication found in literature

<table>
<thead>
<tr>
<th>Type of economic value damages</th>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Damage estimation</th>
<th>Scale</th>
<th>Year</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>UVD – Use value damages</td>
<td>Reduced revenues in tourism, fishery sector etc.</td>
<td>Potential decrease in the number of nights (touristic accommodations in coastal area / marina)</td>
<td>189 283 nights</td>
<td>Central Denmark Region</td>
<td>2013</td>
<td>(Bastis Tourism, 2011).</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loss of sensitive species in favour of opportunistic species</td>
<td>N/A</td>
<td>Central Denmark Region</td>
<td>N/A</td>
<td>(Nutrient Eutrophication in Danish Marine Waters, 2004)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loss of revenues from tourism due to algae bloom</td>
<td>€ 10 million</td>
<td>Oland coastal area</td>
<td>2005</td>
<td>(University of Kalmar, 2008)</td>
</tr>
<tr>
<td>Increased healthcare expenditures</td>
<td>Increased cases of water pollution related disease</td>
<td>N/A</td>
<td>Central Denmark Region</td>
<td>N/A</td>
<td>(Nutrient Eutrophication in Danish Marine Waters, 2004)</td>
<td></td>
</tr>
<tr>
<td>Industry expenditures</td>
<td>Costs of monitoring programme for toxin for shellfish</td>
<td>2.5 million DKK costs (c.€ 335 000)</td>
<td>Denmark</td>
<td>1997</td>
<td>(Anderson, 2001).</td>
<td></td>
</tr>
</tbody>
</table>

**Passive use value damages (PUVD)**

**Water pollution – Negative effect on health**

When decomposing, green algae generate large quantities of NH$_3$ and H$_2$S toxic gases, which can impose direct threats to human health and impacts on ecosystems. Nitrate in drinking water has been suspected of negative effects on human health with an associated costs for the loss of healthy life years of € 6.6 per capita in Denmark (Dalgaard, et al., 2014).

The willingness to pay (WTP) for a Baltic Sea undisturbed by excessive inputs of nutrients was studied in 2008. During the study, a survey was conducted throughout the Baltic States asking for the household WTP for a 50 % reduction in nutrient loading level. The survey found a WTP range of € 70- € 160 per household for the Eastern European Baltic States with lower GDP and € 500-€ 800 for the Baltic States with higher GDP (including Denmark). Danish households reported a high WTP of € 920 million (€ 823 per household) which as a cost function can be converted to € 42 /kg N (Brink & van Grinsven, 2011).

**Water pollution – Negative effect on ecosystems**

Whilst it is not possible to directly translate the diminution of species into financial losses, it indicates a loss of biodiversity that impacts economic activities such as fishing, recreational fishing and tourism. Some habitats are more vulnerable to nutrient enrichment; that is the case of wetlands, for example.

A study has estimated the cost of mitigating reactive nitrogen inputs to the Baltic Sea for a range of measures at € 7-18 /kg N reduced (Brink & van Grinsven, 2011).

**Air pollution**

Excessive nutrient loading can also lead to air pollution and consequently cause physical and economic damages to human health and ecosystems in the region.
A study reports the average unit damage cost for health impacts in Denmark by airborne NH\textsubscript{3} at € 10 /kg N and at € 14 /kg N for NO\textsubscript{x} (Brink & van Grinsven, 2011). In addition, the study estimates the unit health damage costs from N-leaching from agricultural land at € 0.6 /kg N (Brink & van Grinsven, 2011).

**Policy action costs (PAC)**

Some literature was available on total costs of different activities related to nutrient losses and related impacts (Table 28), however very little information was found relating specifically to the Central Denmark region (Table 28).

A review of the Danish river basin management plans was conducted, it was estimated that the application of measures included in the CAP on 15 000 hectares would reduce 5 250 tonnes of nitrogen and 126 tonnes of phosphorus for an annual cost of 135 million DKK (approximately € 18 million)\textsuperscript{118}. Similarly, the restoration of 10 000 hectares of wetland would reduce 1 131 tonnes of nitrogen for an annual cost of 62 million DKK (approximately € 8.3 million) (Jacobsen, 2012).

For measures related to nutrients, the following costs were identified: for no tillage in autumn measure, a cost of DKK 18/kgN/ha (€ 2.4/kgN/ha) was calculated and it is expected that the efficiency is greater on sandy soil. Some measures require that the land is not used; the loss of income due to the unavailability of the land was compared to its lease value and is presented for the Central Denmark region (Table 28).

<table>
<thead>
<tr>
<th>Cost</th>
<th>West Jutland</th>
<th>East Jutland</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rental price</td>
<td>DKK 3 877 (c. € 500)</td>
<td>DKK 3 761 (c. € 490)</td>
</tr>
<tr>
<td>Rental price minus payment under the CAP*</td>
<td>DKK 1 577 (c. € 205)</td>
<td>DKK 1 461 (c. € 190)</td>
</tr>
<tr>
<td>Expected loss of earning</td>
<td>DKK 1 892 (c. € 245)</td>
<td>DKK 1 753 (c. € 230)</td>
</tr>
</tbody>
</table>

\* CAP payment is DDK 2,300 /ha (c. € 300/ha)  

Source: (Jacobsen, 2012)

In 2004, the Danish Government has adopted its Action Plan on the Aquatic Environment III for the period 2005-2015. Information on the costs of the previous Action Plan on the Aquatic Environment was available. For example, the first action plan covering the 1985-1989 period has been estimated to have cost € 1.2 billion. This includes investments in agriculture, but mainly investments in municipal wastewater treatment plants (Conley, 2002).

BalticSTERN (Systems Tools and Ecological-economic evaluation – a Research Network) has conducted research on costs and benefits of mitigating eutrophication and meeting environmental targets of the HELCOM Baltic Sea Action Plan. The targets are maximum allowable loads of nitrogen and phosphorus for each of the seven sea-basins of the Baltic, of which two are in Denmark: the Danish Straits (DS) and Kattegat (KT). The costs of reducing nitrogen and phosphorus vary depending on the modelling tools used. For Denmark, they range between € 122.5 and € 629.9 million per year (Baltic Stern, 2013).

\textsuperscript{118} These measures include buffer zones, reduction of tillage in autumn and prohibiting plowing grassfields
Table 29 – Policy action costs found in literature

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Cost</th>
<th>Scale</th>
<th>Year</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Positive actions for the environment – Action Plan III</td>
<td>Tax for mineral phosphorus in feed</td>
<td>4 DKK/kg (c. € 0.5 /kg)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>50 000 hectares of buffer zones established</td>
<td>375 million DKK (c. € 50.3 million)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>Measures relating to catch crops requirements</td>
<td>30-60 million DKK per year (c. € 4-8 million)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>Measures for use of manure</td>
<td>50-90 million DKK per year (c.€ 6.7-12 million)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>Establishment of wetlands</td>
<td>140 million DKK (c.€ 18 million)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>Research programme for agriculture</td>
<td>155 million DKK (c.€ 20 million)</td>
<td>Denmark</td>
<td>2005-2015</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td></td>
<td>Research programme for organic farming</td>
<td>40 million DKK per year (c.€ 5 million)</td>
<td>Denmark</td>
<td>2005-2009</td>
<td>(Jacobsen, 2012)</td>
</tr>
<tr>
<td>Maintenance and monitoring</td>
<td>Planning and management of the aquatic environment by state</td>
<td>€ 1.28 billion / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Planning and management of the aquatic environment by local authorities</td>
<td>€ 68 billion / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Maintenance and restoration of rivers, streams and lakes</td>
<td>€ 81 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>National monitoring of aquatic environment</td>
<td>€ 40 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Supply of clean and healthy drinking water (incl. nutrient removal)</td>
<td>€ 405 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Discharge and cleaning of wastewater</td>
<td>€ 676 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Danish National Aquatic Monitoring and Assessment Programme</td>
<td>€ 26 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Compliance and monitoring</td>
<td>€ 11 million / year</td>
<td>Denmark</td>
<td>-</td>
<td>(Conley, 2002)</td>
</tr>
<tr>
<td></td>
<td>Monitoring programme for toxin for shellfish</td>
<td>2.5 million DKK costs</td>
<td>Denmark</td>
<td>1997</td>
<td>(Anderson, 2001)</td>
</tr>
<tr>
<td>HELCOM</td>
<td>Reducing N fertiliser at source</td>
<td>€ 1-125 /kg</td>
<td>Helcom region</td>
<td>2012</td>
<td>(Baltic Stern, 2013)</td>
</tr>
<tr>
<td></td>
<td>Reducing N fertiliser at sea</td>
<td>€ 2-158 /kg</td>
<td>Helcom region</td>
<td>2012</td>
<td>(Baltic Stern, 2013)</td>
</tr>
<tr>
<td></td>
<td>Reducing P fertiliser at source</td>
<td>€ 0-350 /kg</td>
<td>Helcom region</td>
<td>2012</td>
<td>(Baltic Stern, 2013)</td>
</tr>
<tr>
<td></td>
<td>Reducing P fertiliser at sea</td>
<td>€ 0-463 /kg</td>
<td>Helcom region</td>
<td>2012</td>
<td>(Baltic Stern, 2013)</td>
</tr>
<tr>
<td></td>
<td>Meeting HELCOM action plan targets</td>
<td>€ 122.5 and € 629.9 million / year</td>
<td>Denmark</td>
<td>2012</td>
<td>(Baltic Stern, 2013)</td>
</tr>
<tr>
<td></td>
<td>Raising public awareness (meetings, pamphlet, notice boards)</td>
<td>€ 152 400</td>
<td>Lake Fure</td>
<td>2006</td>
<td>(Frederiksborg County, 2006).</td>
</tr>
</tbody>
</table>
5.2.4 Good practices to reduce nutrient losses at farm level

For Central Denmark, a first set of measures focuses on reducing the quantity of nutrients created or applied through the adjustment of the feed quantities, the transfer of manure, the practice of organic farming, the reduction of the quantity of fertilisers applied and the use of best suited application techniques. Measures related to manure processing would help at easing its use and transport and valorise it if it is transferred to another farm or used to produce energy. The third set of measures focuses on reducing leaching to soil and water and emissions to air at different stages: storage, housing and land application on field or on pasture. The measures are the use of partly slatted floor, slurry tank coverage, the optimisation of the grazing intensity, the better implementation of buffer strips and the use of catch crops. Lastly, the measures aiming at improving soil quality and sol nutrient retention capacity such as appropriate tillage techniques and the use of perennial crops are also relevant to decrease the nutrient surplus in Central Denmark.

5.2.4.1 What has already been done in the region

Denmark has adopted a territory-wide approach for the implementation of the Nitrates Directive. The Danish Action Programme for the Aquatic Environment III has been adopted for the period 2005-2015 with a strong emphasis on phosphorus surplus in agriculture. The aim was to halve phosphorus surplus by 2015 and to reduce nitrogen leaching by a minimum of 13 % (compared to 2003). A mid-term review in 2008 showed that phosphorus surpluses were reduced, but nitrogen leaching had not decreased, even though farmers are required to use a nitrogen fertiliser rate, which is approximately up to 15 % below the economic optimum level. Some of the measures most recently implemented to reduce the eutrophication of water bodies include:

- Establishment of buffer zones along certain watercourses and lakes
- Increased area of catch crops
- Re-establishment of wetlands, which stimulate nitrate removal by microbial conversion
- Ban on certain forms of soil cultivation during autumn and winter

Under the Action Programme, Danish livestock holding must ensure a balance between agricultural land and the number of livestock units corresponding to a maximum of 170 kg N/ha of nitrogen from manure for cattle holdings and 140 kg N/ha of nitrogen for all other livestock holdings. The Danish derogation to the Nitrates Directive allows a maximum of 230 kg N/ha for cattle holdings where the crop rotation includes more than 70 % of crops with high nitrogen uptake and long growing season\(^\text{119}\). The current Danish legislation also demands that new livestock production facilities implement the best available techniques to reduce ammonia emissions\(^\text{120}\). Existing livestock farms can only be expanded if the neighbouring ecosystem’s maximum capacity for nitrogen deposition will not be exceeded. Liquid manure used as fertiliser on bare soil or grass must be directly injected or acidified to reduce ammonia volatilisation. In addition, the Danish AgriFish Agency developed an environmental technology programme supporting new green technologies on farms.

In addition, the Danish Agricultural Advisory Service (DAAS) provides support to farmers. It is owned and managed by farmers, via their membership of farming organisations. It includes the Knowledge

\(^{119}\) According to Decision 2012/659/EU Denmark has a derogation for specific holdings where manure may be spread in a quantity equal to maximum 230kg/N per hectare per planning period

\(^{120}\) Description of the Danish BAT for livestock production are available here: [http://eng.mst.dk/topics/agriculture/environmental-technologies-for-livestock-holdings/bat-%28best-available-techniques-%29/](http://eng.mst.dk/topics/agriculture/environmental-technologies-for-livestock-holdings/bat-%28best-available-techniques-%29/)
Centre for Agriculture which coordinates the activities of 31 local advisory centres and acts as a development and support unit for DAAS.

Denmark is part of the Baltic Sea catchment area where every year WWF award a ‘farmer of the year’ prize and a regional farmer award. One of the criteria is that nominees must have undertaken concrete measures to reduce nutrient emissions from their farm. For the competition, nutrient emissions include both nutrient losses to water and gaseous losses as ammonia emissions from manure.

5.2.4.2 Good practices to reduce the nutrient losses in livestock production

The good practices were selected based on their impacts on the agro-ecosystem in terms of reduced losses through improved nutrient utilisation. Thus, the selected measures presented below provide some economic advantages for the farmer and at the same time reduce the nutrient surplus of the farming system, benefitting both the environment and society. Emphasis was placed on measures that have not yet been exploited to their full potential within the Central Denmark region. Further selection criteria for possible solutions included whether the measure might be feasibly implemented and whether it is cost-effective.

Use feeding practices to reduce the amount of nutrient produced

In Central Denmark, intensive livestock rearing systems are sources of manure whose nutrient content is excessive relative to the needs of the crops in the region.

The quality of livestock feed can be improved by adding enzymes, which increase the digestibility and nutritional value of the feed, especially concerning phosphorus availability. This measure has already been implemented in Central Denmark by farmers to some extent and has resulted in a reduction of nutrient excretion per kg of produced meat, milk or eggs. In order to assist farmers nutrient requirement standards have been published for pigs by the Pig Research Centre. They are based on the physiological energy value of nutrients and on the standardised digestibility of these nutrients (Pig Research Centre, 2013).

In addition, since 2009 a tax applies on mineral phosphorus in feed.

Favour organic farming

In 2010, out of the 2 100 organic farms registered in Denmark, 680 of them (representing 52 100 hectares, or 7 % of the agricultural land) were located in the Central Denmark region (Eurostat, 2015a). Further conversion of conventional farms to organic farms could reduce nitrogen leaching.

For the current Danish conditions, it is estimated that a conversion from conventional to organic farming practices could reduce the nitrogen leaching by 10-17 kg N/ha. For comparison, a study carried out in Schleswig-Holstein found a reduction in leaching from the root zone of 2-9 kg N / ha on organic farms compared to leaching from conventional farming systems (Børgesen & al, 2013).

It is important to note that Danish consumers are characterised by a high willingness to pay for organic products (Certcost, 2011). Further development of organic production will be driven by stronger demand (local or to export) for organic products.

Frameworks are in place in Denmark to support the development of organic farming. The Danish Agricultural Advisory Service is administrated by the Danish farmers’ union and the Danish Agriculture and Food Council. The Advisory Service consists of 30 regional agricultural centres providing advises for farmers. In 2011, these centres included 90 advisers (corresponding to 40 full-time jobs) specialised in organic farming. Their task includes providing information to conventional farmers that are inspired to
adopt organic methods of production (Norfelt, 2011). In addition, Økologisk Landsforening also has advisers providing support on organic production methods.

**Optimise grazing intensity**

As a result of structural changes in the Danish dairy sector with larger farms, the management practice for cattle has changed from grazing on pasture in the summer season to the current practice where cattle are kept in the stable throughout the year (Børgesen & al, 2013). This shift toward more indoor housing and less grazing could reduce ammonia emissions from manure and urine deposited outdoors but emissions of \( \text{CH}_4 \) would be increased as more manure is being stored (ADAS, 2011). Indoor housing allows for a better control of feed; however this is not compatible with practices such as organic farming. Indoor housing also poses animal welfare issues that need to be considered.

**Transfer manure**

Denmark applies its Nitrate Action Programme to the whole territory, which limits the amount of nitrogen from manure that can be applied. The Danish Nitrates Action Programme allows the transfer of manure outside the farm where it is produced.

The success of this measure depends on the availability of land for the surplus manure to be applied, the crops on this land and biosecurity issues. Research has identified that 50 % of Danish farmers have been involved in manure transfer (Asai, et al., 2014). A study on livestock farmer perceptions of manure exchange in Denmark was conducted in 2014 and consisted of a survey of 644 manure exporters and importers. The survey identified that exporters are mostly larger livestock famers (14 % of sample) and the importers mainly arable farms (29 % of the sample). A total of 6 % of the farmers interviewed were both importers and exporters. The study also revealed that exporters appreciate four qualities: timely communication regarding establishment of a contract, potential for long-term relationship, physical and social accessibilities to the partners and flexibility of acceptance of manure. In addition, organic dairy farmers seemed to place less emphasis on the distance and accessibility of their partner. Exporters on the Danish islands, where the crop production dominates were significantly more concerned about the characteristics of the partner in terms of professional skills and business expertise (Asai, et al., 2014). Furthermore, data included in Eurostat for 2010 indicated that the farms actually exporting manure in Central Denmark, the majority (1,020) are exporting 1 to 24 % of the manure produced\[121\].

In Denmark, exporting farmers are required to submit information about the manure received, including the quantities and the type of manure exported. This information is cross-checked by agencies to ensure all manure is registered and applied correctly. While manure has a value as source of nutrients, in Denmark exporting farmers often have to pay for the transport or the application and are very rarely paid for the nutrient value of its manure (Asai, et al., 2014).

Finally 19 % of farms registered in the Danish Fertilizer Account solely applied mineral fertiliser in 2009. Manure transfer could be appealing to these farmers (Asai, et al., 2014).

**Use partly slatted floor**

A better collection system avoids losses of nutrients to volatilisation. This is relevant for Central Denmark as the overwhelming majority of ammonia's emissions at national level are from agriculture (97 %). Furthermore, these techniques are more suited for larger units, which is appropriate for Central Denmark.

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\[121\] According to Eurostat data for 2010, out of the 42,100 farms involved in transferring manure in Denmark, 12,840 are located in Central Denmark region. However, further analysis of these data showed that the majority (10,120) have indicated either exporting 0 % of manure or that manure export is not applicable.
where a large number of intensive farms are located. There are no available figures on the number of livestock housing units which have already adopted a partly slatted floor design in Central Denmark.

**Cover slurry tanks**

Over the past 20 years, the storage facilities for manure have improved markedly, leading to an increased volume of manure stored and an increased proportion of that manure being applied in spring and summer. This has led to reduction of nutrient losses to water and emissions to air. In 2012, 100 % of pig farms and 71 % of cattle farms had slurry storage capacity for 9 months.

Taking into account these improvements, slurry tank coverage is an efficient practice to reduce the ammonia emissions from storage. Since 2007, new slurry tanks located less than 300 meters to households must have a cover. A floating cover (natural or artificial) is the most popular solution in Denmark (Olesen & al, 2013). Currently, about 6-7 % of the slurry tanks are covered with a solid cover which are mostly the larger ones containing up to 10-12 % of Danish livestock slurry (Olesen & al, 2013).

**Process slurry**

In Denmark, 90 % of the manure produced is slurry. Processing slurry could help at decreasing the volatilisation during storage (e.g. acidification) and help at using slurry at farm level or transport and sell it (e.g. solid-liquid separation).

**Acidify slurry**

During the period 2010-2013, the Danish AgriFish Agency has supported investments in slurry acidification equipment at 318 Danish livestock farms. In 2012, about 11 % of the Danish slurry was acidified, of which only 3 % was acidified in the stable (Olesen & al, 2013). The remaining slurry was acidified either in the slurry tank immediately before application or in connection with field application. Feedback from Danish expert review indicated that acidification of the slurry could lead to soil acidification, increasing the need for liming up to 40-50 % of the additional liming needed. Theoretically, 1.4 kg lime should be land spread to neutralise each kg of sulphuric acid used for slurry acidification (Joint Research Centre, 2013a).

Other techniques to reduce ammonia emissions such as slurry cooling is considered by experts to have a limited relevance and apply only in barns on farms where the heat can be utilised. No information was identified on the extent to which this measure is implemented in Central Denmark.

**Separate liquid and solid fractions of slurry**

An analysis made by AgroTech, found that ammonia emissions from pig stables can be reduced by 75 % if a new stable design is implemented, which includes separation of slurry into a liquid and solid fraction (Agro Tech, 2013). The liquid slurry fraction is subsequently acidified, leading to reduced ammonia emissions from the stable and during field application. The solid slurry fraction is then used for biogas production. This technique requires specific measures to be installed which can be costly if retro-fitted. For example the cost of installing a manure channel with slopped floor to an existing housing

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122 Acidification adds around 3 litre of acid per tonne of slurry (= 5 kg acid). Approximately 30 tonnes of slurry can be applied per hectare which corresponds to 150 kg acid per hectare. Soil requires 1.4 kg lime per kg acid, so for 150 kg of acid, 210 kg lime per hectare is required. Usually, no more than 2 tonne lime is used every fourth year which corresponds to 500 kg lime per year. The additional 210 kg of lime correspond to 42 % extra lime in comparison to non acidified slurry.
unit is estimated at € 6.45 – 7.74 per place per year for finishing pigs (Gothenburg Protocol Guidance, 2011).

Due to the relatively high investment and operation costs for slurry separation systems, this measure has a significant economy of scale. Farmers with manure quantities less than 5,000 tonnes per year will probably not benefit economically from slurry separation, but at 10,000 tonnes manure annually the profitability increases. Mobile solutions, where a group of farmers share a slurry separation system may be an option. For instance at Bornholm, six farmers have established a manure separation co-operative, where they share a mobile separator and regularly deliver the solid slurry fraction to the island’s biogas plant.

Use digested manure from biogas production

In the long term, there is a greater nitrogen leaching from organically bound nitrogen than from inorganic nitrogen applied to soil (Børgesen et al., 2013). Anaerobic digestion of manure in a biogas reactor mineralises organically bound nitrogen into inorganic nitrogen. Therefore, slightly lower nitrogen leaching can be expected in the long term when fertilising crops with digested manure instead of untreated manure (Børgesen et al., 2013). The long-term reduction of using anaerobic digestion of pig manure was estimated in Denmark to reach to 2.3 kg N per livestock unit on the assumption that leaching would affect 30% of the inorganic nitrogen and 45% for organic nitrogen in the manure (Børgesen et al., 2013). The digested manure can substitute for mineral fertiliser application with a lower risk of leaching in the long term than for untreated manure (Børgesen et al., 2013; Homan, 2014). However, as for other processes, careful use is required while applying the treated product into field to avoid over-fertilisation. Since the digested manure is stable with a lower volume than before digestion, it also indirectly avoids ammonia emissions during storage. The saved amount of nitrogen from the digested manure will lead to reductions in nitrous oxide emissions, ammonia emissions and nitrate leaching from field applied fertilisers (Børgesen et al., 2013). However, reductions in these environmental impacts will depend on whether farmers take the greater nitrogen availability in the digested manure into account when calculating fertilisation rates. Currently, Denmark does not differentiate the nitrogen availability in the digested manure from undigested manure as the same amount of total nitrogen is applied whether digested or untreated manures are used (Børgesen et al., 2013).

This measure is not applicable directly for individual farms mostly due to the costs involved, but a co-operative of farms may decide to set up and operate a biogas plant in order to use the slurry produced and generate some revenue from methane production. It was estimated that biogas could provide 10% (60 PJ or 17 TWh) of total Danish energy needs by 2030, of which 20-25 PJ could be generated from manure (Baltic Manure, 2013).

In Central Denmark, there were seven joint biogas plants in operation in 2010 and another 2 to 4 at a planning stage (Region Midtjylland, 2010). Based on the livestock density, it has been estimated that at least 12 to 14 new plants would be necessary to process all the manure generated in the region (Figure 59) (Baltic Manure, 2013).

However, it is important to highlight that the biogas production has possible negative effects such as land use change. It should also be noted that slurry requires large digester volumes and the biogas production per unit of digester volume is low (Baltic Manure, 2013). Indeed, these systems need other substrates, apart from the pig slurry, to obtain a significant methane production because of the low

123 A biogas reactor is an anaerobic treatment technique that produces digested slurry and biogas. [Not only slurry, it depends on the input materials!]

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content on organic charge of the pig slurry. In addition, the substrate combination should be thoroughly studied in order to avoid inhibitions in the gas production which could result from the mix of substrates.

In total, 34 million tonnes of manure is produced in Denmark annually, of which 80 % is slurry. It is considered that 1/3 of this manure is being produced by the Central Denmark region. At the national level, currently only 5 to 7 % of the manure is used for biogas production (Baltic Manure, 2013).

The Danish government has set a target that 50 % of the manure generated should be treated in a biogas plant by 2020 (Region Midtjylland, 2010). This goal has been judged to be very ambitious and the ‘upper limits’ for the techno-economic potential have been estimated to be nearer 2/3 of all manure produced in the Danish stables as the remaining manure is either produced on small farms or cannot be economically recovered (Luostarinen, 2013b).

Investment costs in Danish biogas plants vary in the range of € 33-66 per tonne of manure, however to reach a target of 50 % of all manure being digested to biogas, an investment of € 0.5-1.0 billion would be necessary (Baltic Manure, 2013). The running and maintenance costs for manure-based biogas plants in Denmark are around € 7-10 per tonne of manure, and the revenue for the methane produced is between € 0.5-0.8 /m$^3$ methane or € 15-21 per GJ of energy (Baltic Manure, 2013).

Beyond the high investments needed to build and operate a biogas plant, one of the main obstacles to the uptake of biogas is public opposition to the opening of new plants. Biogas plants can generate odour and increase road traffic, which are considered as a nuisance for the neighbourhood. Furthermore, planning times are quite long (5-10 years in Central Denmark) which can make attracting investment difficult (Region Midtjylland, 2010). Finally, it is important to note that the slurry produced in the region is relatively difficult for biogas to digest due to its lignocelluloses structure (Frandsen, 2011). Furthermore, the slurry has a high level of water content (90-97 %) so high quantities of slurry need to be imported to the biogas plant for only a small fraction being used. These issues have been identified as making difficult the profitable exploitation of a slurry based biogas production (Frandsen, 2011).
5.2.4.3 Good practices to reduce the nutrient losses in crop production

Reduce the inorganic and organic fertiliser use

The reduction of the surplus of nutrients in Central Denmark’s soils can be achieved through the reduction of mineral fertiliser use. Denmark uses significant quantities of inorganic fertilisers although they have been declining since 2000 (Table 30).

<table>
<thead>
<tr>
<th>Year</th>
<th>2000</th>
<th>2005</th>
<th>2009</th>
</tr>
</thead>
<tbody>
<tr>
<td>Used in agriculture</td>
<td>234</td>
<td>192</td>
<td>190</td>
</tr>
</tbody>
</table>

Source: (National Environmental Research Institute, 2011)

The choice of fertilising techniques depends on agricultural practices and farmers’ preferences. In some circumstances it may be more convenient and/or economical to purchase inorganic fertilisers. The Danish Nitrates Action Programme requires that farmers fertilise 10 % to 15 % under the economic optimum. This is considered by the farmers’ unions to be detrimental to the competitiveness of the Danish farmers. They argue that due to the general increase in crop yields and the lack of update of the economic optimum the current level at which they are allowed to fertilise is actually closer to 15 % under the economic optimum. As a result, although very effective, further reductions of the fertilisation rate that farmers must meet may not be deemed acceptable by farmers’ unions (Jensen-Skov, 2013).

Use adequate tillage techniques to reduce erosion and increase soil quality in order to limit nutrient leaching

The reduction of nutrient losses during soil application in Central Denmark can be achieved through the improvement of soil quality. Soil tillage is a disturbance of the soil as it prevents the development of the litter layer and the improvement fertility improvement and it reduces the SOC/sequestration. Some good practices have been identified specifically for Denmark in order to limit tillage erosion (Schjønning, et al., 2009):

- The conversion from conventional to reduced tillage systems reduces the tillage erosion.
- Reducing the tillage speed and depth substantially reduces the tillage erosivity. If applying this measure, low speed must be kept on slope operations.
- After contour tillage, slantwise tillage (turning the soil upslope) is the least erosive.
- The frequency of tillage operations should be reduced and the loosening of soil before ploughing avoided.

Use best suited application techniques

Denmark has already implemented requirements in relation to application techniques and the incorporation of manure. As a result the scope for reducing losses during the application of manure is more limited than in other countries. For example, the use of splash plates to broadcast liquid manure is banned and the injection of slurry to grass is mandatory (Gyldenkaerne, 2011). Furthermore the application of slurry to crops must be by trailing hose or injection (AEA, 2010c). In 2012, 40 % of the slurry was applied with trailing hose techniques and the remainder injected (Blicher-Mathiesen, 2013).

However, for winter crops (sown in autumn) slurry fertilisation takes place during spring using trail hoses as the injection equipment would damage the roots. Regarding manure incorporation, all manure (solid or liquid) applied to bare soil, must be incorporated within 6 hours.

Table 31 details the application technique used and the length of time before incorporation in Denmark. No specific information on the technique used in Central Denmark is available.
Table 31 – Estimate of the use of different application techniques in Denmark

<table>
<thead>
<tr>
<th>Application method</th>
<th>Application time</th>
<th>Distribution of manure (in %)</th>
<th>Length of time before incorporation (in hours) and not incorporated</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Cattle</td>
<td>Pigs</td>
</tr>
<tr>
<td>Liquid manure</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Injected</td>
<td>Winter-spring</td>
<td>49</td>
<td>24</td>
</tr>
<tr>
<td>Injected</td>
<td>Summer-autumn</td>
<td>14</td>
<td>4</td>
</tr>
<tr>
<td>Trailing hoses</td>
<td>Winter-spring</td>
<td>26</td>
<td>64</td>
</tr>
<tr>
<td>Trailing hoses</td>
<td>Spring-summer</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Trailing hoses</td>
<td>Late summer - autumn</td>
<td>9</td>
<td>6</td>
</tr>
<tr>
<td>Solid manure</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Broad spreading</td>
<td>Winter-spring</td>
<td>81</td>
<td>81</td>
</tr>
<tr>
<td>Broad spreading</td>
<td>Late summer - autumn</td>
<td>19</td>
<td>19</td>
</tr>
</tbody>
</table>

Source: (National Environmental Research Institute, 2011)

The Danish legislation requires a minimum utilisation rate of the total nitrogen content in manure to be respected in the fertiliser plans and the fertilisation accounts (Baltic Deal, 2012a). The minimum rates vary according to the type of manure and the livestock it is excreted from (Table 32).

Table 32 – Demanded utilisation rate of the nitrogen content of manure

<table>
<thead>
<tr>
<th>Type of manure</th>
<th>Demanded utilisation rate (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pig slurry</td>
<td>75</td>
</tr>
<tr>
<td>Cattle slurry</td>
<td>70</td>
</tr>
<tr>
<td>Mink slurry</td>
<td>70</td>
</tr>
<tr>
<td>Poultry slurry</td>
<td>70</td>
</tr>
<tr>
<td>Other type of manure</td>
<td>65</td>
</tr>
<tr>
<td>Solid manure</td>
<td>65</td>
</tr>
<tr>
<td>Deep litter</td>
<td>45</td>
</tr>
</tbody>
</table>

Source: (Baltic Deal, 2012a)

**Better implement buffer strips near watercourses**

The use of buffer strips could be further implemented. Since September 2012, the Danish legislation requires that a 10 metre buffer zone is applied for fields next to open water streams and lakes larger than 100 m². The implementation of this measure has been reported as ‘complicated’ due to incorrect electronic mapping and malfunctions in official reporting websites. The national farmers’ unions and the government have also disagreed on what was to be considered as a natural open water stream (Jensen-
Skov, 2013). A better implementation of this measure would reduce leaching of agricultural soils to nearby watercourses and lake from water run-offs or erosion. This would reduce the nutrient load in watercourses and lakes which are in the Central Denmark region affected by eutrophication.

**Use perennial energy crops**

Perennial crops are established and bloom over at least a two-year period and have a permanent, deep root system, which ensures efficient nutrient utilisation. Cultivating perennial crops instead of annual crops may therefore reduce nutrient losses. According to the Ministry of Food, Agriculture and Fisheries research have shown that willow as perennial energy crop can reduce the leaching of nitrogen and phosphorus by 30-45 kilos nitrogen and 8-12 kilos phosphorus per hectare compared to annual crops such as cereals. This represents a drop by up to 70 percent in leaching of nitrogen (Danish Ministry of Food, Agriculture and Fisheries, 2010). In Denmark, N leaching from a sandy soil growing willow crop and fertilised with 120 kg N/ha was measured to be 1-7 kg N/ha. This correspond to a reduction of more than 60 kg N/ha (Børgesen & al, 2013). On lowland soils, conditions are more variable than for upland soils due to differences in drainage rates, mineralisation potentials and hydrological conditions, and it is not possible to give a reliable estimate of overall effects of selecting perennial bio-energy crops on N leaching. However, as an indication, it is considered that the leaching reduction from perennial bio-energy crops on lowland soils would be between 0 and 100 kg N/ha (Børgesen & al, 2013).

However, biomass production for bioenergy often necessitate changes to land use, with significant implications for related systems, including nutrient cycles. Even while the effects of using biomass for energy will vary greatly from location to location, it often involve further intensification (including fertiliser use and N input) of existing land uses. It could mean converting directly or indirectly non-cropped land (e.g. pasture land) into cropped land with significant interference in nutrient cycles (EEA, 2013a). In addition, growing perennial crops has raised concerns about biodiversity and landscape features. There is a fear that growing high crops such as willow will affect the visual presentation of the landscape. In addition, perennial energy crops are more suited to some fauna connected to closed landscapes, so there is the risk that it would disadvantage open landscapes species (Danish Ministry of Food, Agriculture and Fisheries, 2010).

**Use catch crops**

It is estimated that in Denmark N leaching on average can be reduced by 25 kg N/ha when cultivating well-established non-nitrogen-fixing catch crops after common agricultural crops fertilised with recommended amounts of mineral fertiliser or manure (Børgesen & al, 2013). The effect of catch crops is more important on sandy soils than on clay soils (Table 33). The mean effect of 25 kg N/ha relates to farms where the manure application rate corresponds to less than 0.8 livestock units per ha (LU/ha). Additionally, it is estimated that leaching reductions are 12 kg N/ha greater when catch crops are grown on land, which has more than 0.8 LU/ha. Thus, the average effect on farms with fewer than 0.8 LU/ha is 25 kg N/ha and on farms with more than 0.8 LU/ha is 37 kg N/ha. To achieve such leaching reductions, good management is needed with the right choice of catch crops and timely sowing and uniform establishment of the following crop (Børgesen & al, 2013).

<table>
<thead>
<tr>
<th>Table 33 – Estimated reduction of N leaching when growing catch crops on sandy and clay soil with less or more than 0.8 LU / ha, respectively in kg N/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Less than 0.8 LU/ha</strong></td>
</tr>
<tr>
<td>Clay</td>
</tr>
<tr>
<td>16</td>
</tr>
<tr>
<td><strong>Average : 25</strong></td>
</tr>
</tbody>
</table>

Source: (Børgesen & al, 2013)
Po River basin provinces are the most saturated areas for nitrogen and phosphorus in Italy. The impacts related to nutrient load in the Po River basin waters are mostly due to agriculture, especially in the Lombardy plain area and, to a lesser but notable extent, to sewage water from households and industry in western Lombardy.

The surface waters of the Po River basin are among the Italian surface water bodies with the highest nitrate concentrations. The nitrate ecological status of surface water was, on average, "moderate" in 2012 according to the status scale stated by the Water Framework Directive. In regards to groundwater, nitrate contamination is observed in the upper plain (concentration occasionally exceeding 50 mg NO$_3^-$/L). Nutrient contents of all lakes and rivers in Lombardy end up in the Po River that drains into the Adriatic Sea. Lombardy is a high contributor of the nitrogen load carried by the Po River. This discharge of nitrogen in the north-west of the Adriatic Sea contributes to the eutrophication of coastal water where algal blooms are observed. In addition, Lombardy is responsible for significant ammonia emissions (100 000 tons of NH$_3$ per year, among which 98 % comes from agriculture). Such emissions lead to acid rainfalls in alpine and subalpine areas of Lombardy. The resulting soil and water acidification affects terrestrial and aquatic ecosystems. Furthermore, some cases of freshwater eutrophication due to phosphorus are observed in northern lakes such as Lake Como, Lake Iseo, and Lake Lugano.

Lombardy is characterised by intensive agriculture, in particular livestock fed with imported concentrates. This leads to a high volume of manure produced in comparison with the needs of the crops in the region. In addition, manure nutrient content and efficiency are in general not accurately known thus inciting farmers to apply manure in high amounts to secure crop yields. The inefficient application of fertilisers and the wide use of surface irrigation are other causes of nutrient losses. Losses are also enhanced by the occurrence of rainfalls and droughts that increase run-off and leaching but also denitrification, the different types of soil textures encountered in the region and the dense and complex hydrology network.

In 2006, the “Programme for the protection and use of water” (PTUA) was approved at the Po River basin scale. Various measures of this programme aim at reaching good quality status of water bodies in the Lombardy Region. The investment required for the implementation of such measures and the achievement of water quality objectives was estimated at € 4.5 million. A measure from this programme specifically targeting nitrate pollution is the "Emergency Action Plan". This has financed some infrastructure and engineering investments to improve the management of livestock manure. A cost of € 67 million was reported for this plan for the period 2005-2008.

In Lombardy, the identified good practices include measures aiming at decreasing the local source of pollution by controlling the geographic distribution of the breeding activities, transferring manure, adapting feeding strategies, improving the nutrients fertilisation management plans for all agricultural sites and processing manure to ease its transfer and use. Another type of good practice relates to the reduction of the amount of nutrients lost during housing and storage by improving manure collection and covering the slurry tanks. A third type of good practices relates to the reduction of nutrient losses during application by using the most suitable application techniques and improving the irrigation systems. Lastly, measures such as the use of adequate tillage techniques and soil coverage with catch crops would help to increase the nutrient retention capacity of soil and decrease the nutrient losses.

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124 Authors: Mrs. Marion Sarteel, BIO by Deloitte (lead author); Mr. Giorgio Provolo, Milan University; Mrs. Alice Landowski, BIO by Deloitte; Mr. Clément Tostivint, BIO by Deloitte; Mrs Helen Ding, BIO by Deloitte.
Lombardy is located in the north of Italy and extends over 2.4 million hectares between the Alps and the Po River which runs from the western Alps to the Adriatic Sea (Figure 60). Lombardy covers about one third of the Po River basin and is one of the most important areas in Italy for water resources, including major rivers flowing into and out of large lakes. Its capital, Milan, is the second most populated city and the biggest industrial urban area of the country. The Po River basin presents a topographic-influenced climate, from alpine in the mountain zone to continental-warm in the flat basin area and Mediterranean on the coast (Fumagalli, et al., 2011; Raggi, et al., 2007; Salvetti, et al., 2006; European Commission, 2012c; Laini, et al., 2011; Cremonesi, et al., 2011; Region of Lombardy, 2006).

Lombardy is one of the leading agricultural regions of Italy with 939 000 hectares dedicated to agricultural lands in 2014 (Region of Lombardy, 2014a), representing 39.4 % of the regional territory. Lombardy accounted for 14.0 % of the value of the national agricultural production in 2013\(^\text{125}\) (INEA, 2014)\(^\text{126}\).

Arable lands, mainly localised in the plains, represented 77.3 % of Lombardy UAA in 2012 (Region of Lombardy, 2015a). They are mostly cultivated with maize (used for grain and silage), rice and wheat. In particular, rice represented 40.5 % of the surface dedicated to rice production in Italy in 2013 (Region of Lombardy, 2015a). Permanent crops and grassland, mainly located in hilly areas, represented 22.7 % of UAA (Region of Lombardy, 2015a).

Among the 54 300 Lombardian agricultural holdings in 2010, generally small in size compared to other selected saturated regions, 40.6 % conducted breeding activity (Region of Lombardy, 2014a). In 2014, Lombardy counted 4.66 million pigs, 1.53 million beef and dairy cattle and 32.4 million poultry (Region of Lombardy, 2014a). In 2013, pig, beef, dairy cattle and poultry production in Lombardy represented 40.1 %, 37.1 %, 26.0 % and 18.9 % of the national production respectively (in tonnes) (Region of Lombardy, 2015a). Lombardy livestock products are recognised for their overall good quality through

\(^{125}\) Agricultural output at basic price (OBP)

\(^{126}\) INEA: Italian Institute of agricultural economics
several distinctions including PDO, PGI and TSG marks\(^{127}\) that have been granted to over 250 products in the region (Region of Lombardy, 2015b). The resulting unique production methods notably emphasise the importance attached to the protection of the regional territory in livestock production.

### 5.3.1 Notable impacts of nutrient surplus

This section focuses on the impacts of nutrient excess that are specifically related to agricultural practices in Lombardy. The main regional impacts are caused primarily by nitrogen surpluses, mostly affecting water and air, but also by phosphorus, affecting water. Both nutrient surpluses not only result from agricultural production but also from industrial activities and urban settlements (EEA, 2005b). There is no identified environmental issue related to potassium in Italy.

The Po River basin provinces are the most saturated areas for nitrogen and phosphorus in Italy, with surpluses exceeding 150 kg N per hectare and 200 kg P per hectare in some municipalities of Lombardy in 2004 (Figure 61). Lombardy has some highly saturated areas, with a high livestock density and high amount of manure applied (ARPA Lombardia, 2013).

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\(^{127}\) PDO means “Protected Designation of Origin”, PGI refers to “Protected Geographical Indication” and TSG to “Traditional Speciality Guaranteed”, according to the EU schemes that protect and promote names of quality agricultural products and foodstuffs. More information available at:

http://ec.europa.eu/agriculture/quality/schemes/index_en.htm
Nitrogen load in water is mainly due to agriculture that accounts for more than 50% of the total nitrogen load to water bodies in the Po River basin (EEA, 2005b; Lynche-Solheim, et al., 2010). Currently, 62% of the regional UAA are designated Nitrate Vulnerable Zones (Region of Lombardy, 2014a).

**Surface water** – The surface waters of the Po River basin are among the Italian surface water bodies with the highest nitrate concentrations (Massarutto, 1999). Over the 2008-2011 period, 11.9% of the abstraction stations were recorded with more than 25 mg NO$_3$-L and 37.3% of the stations were recorded with a nitrate concentration between 10 and 25 mg NO$_3$-L (ISPRA, 2013). The ecological status of surface water was, on average, “moderate” in 2012 according to the status scale stated by the Water Framework Directive (ISPRA, 2013). For the period 2009-2011, only 28% of the monitoring points achieved the “good” and “high” ecological status (ARPA Lombardia, 2011). The quality status of the main watercourses in Lombardy depends on their geographical location and the related land use. While the surface waters and lakes have good or very good ecological status in the northern mountain area, the status is sometimes moderate or poor in the south of Lombardy where most farming activities are located, and very poor in urban and industrialised areas (see Figure 62) (Region of Lombardy, 2008).

In addition, freshwater quality is crucial for biodiversity and ecosystem balance. In Lombardy, environmental issues related to nitrogen surplus may particularly damage fish life in lakes and rivers,

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128 ARPA Lombardia: Lombardian regional agency for the protection of the environment
and thus disrupt both recreational and professional fishing activities that are well developed in the region (Region of Lombardy, 2008). Water ecosystem quality has thus been brought into focus with the Regional Act “Regulations on fish stock increase and fishing activities in Lombardy water bodies”, dated 2001. Within this framework, guidelines for fishing management and water restoration measures to protect and develop fish fauna have been implemented (Region of Lombardy, 2008).

**Groundwater** – Nitrate contamination in groundwater is observed in the upper plain\(^{129}\) with nitrate concentration occasionally exceeding 50 mg NO\(_3\)/L (see Figure 63) (Sacchi, et al., 2013; Region of Lombardy, 2008). Nitrate concentrations above 50 mg NO\(_3\)/L were recorded for 4.7 % of the abstraction stations of the region for the period 2008-2011 while 8.6 % of the stations recorded a nitrate concentration from 25 to 50 mg NO\(_3\)/L (ISPRA, 2013). However, no specific health problems due to the overload of nitrogen in groundwater have evidence (Provolo, 2014). Waters with a good environmental status can be found in the central area of the plain (see Figure 64) (Region of Lombardy, 2008), despite the intensity of the agricultural activity.

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\(^{129}\) Aquifers in the mountain area are not well known and their quality status has not been investigated a lot up to now. Most of monitoring stations are localised in the plain area due to the ease of access to the water bodies.
Nutrient contents of all lakes and rivers in Lombardy end up in the Po River that drains into the Adriatic Sea where it finally discharges all nutrients, including nitrates. Lombardy accounted for 43% of the nitrogen load carried by the Po River to the Adriatic Sea, resulting in a nitrogen load at the Po basin closure estimated at 140,000 tonnes N per year for the period 1985-2001 (Salvetti, et al., 2006). The additional nutrient load in north-western Adriatic Sea is primarily due to the pollution from the Venice lagoon watershed. These high nitrate discharges in the north-west of the Adriatic Sea thus contribute to the eutrophication of coastal water (Fumagalli, et al., 2011; Lynche-Solheim, et al., 2010; Raggi, et al., 2007; Delconte, et al., 2014). Some algae blooms observed in the Adriatic coastal area (Figure 65) have severely disrupted fish and crustaceous life and in some cases even caused extensive mortality of several marine species. Studies have identified the potential main toxic algae, among which *Ostreopsis ovata* and *Skeletonema sp* (Ferrari, 2013; ARPA Veneto, 2007). The last episode occurred during the summer of 2013 that presented favourable conditions, i.e. calm sea and elevated surface water temperatures higher than the seasonal average, accentuating negative effects of high nitrogen load. In addition to damages on the ecosystem, such eutrophication impacts sea products market as well as tourism due to unattractive odours and visible aspects and possibly restricted bathing (Ferrari, 2013; ARPA Veneto, 2007).

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130 This impact does not directly refer to Lombardy but to the Adriatic Sea. However, it is mentioned in this section since the origin of marine eutrophication is significantly situated in the Lombardy region.
Ammonia ($NH_3$) emissions to air

Lombardy emits the highest quantities of air polluting substances in Italy, including 100 060 tons of $NH_3$ per year, among which 98% comes from agriculture (INEMAR - ARPA Lombardia, 2014). Regional emissions of ammonia notably impact ecosystems (Thunis, et al., 2009). Indeed, high ammonia content in the atmosphere leads to acid rainfalls in alpine and subalpine areas of Lombardy. Acid rainfalls significantly increase from north to south, the southern area being closer to industrialised areas of Po Valley that emit other substances also leading to acid rain. The resulting soil and water acidification affects terrestrial and aquatic ecosystems and pollutes freshwater (Balestrini, et al., 2000). Indeed, notable damages are reported in forests due to removal of the protective layer of leaf surface (Bussotti, et al., 2014). Forest surveys conducted in alpine and prealpine areas of Lombardy showed that about 83% of the trees had damaged crowns\textsuperscript{131} (Balestrini, et al., 2000). Trees are therefore more vulnerable to attacks by parasites, diseases and insects. Furthermore, ammonia plays a role in the formation of dry particulate matter that can cause breathing trouble (Deutsch, et al., 2008; Wu, et al., 2008; Takai, et al., 2002).

Nitrous oxide ($N_2O$) emissions to air

In Lombardy, about 80% of the nitrous oxide in the air comes from agriculture. In 2010, approximately 13 200 tons of $N_2O$ were emitted per year representing 14% of the national $N_2O$ emissions (INEMAR - ARPA Lombardia, 2014). Due to its high global warming potential, $N_2O$ is a major greenhouse gas.

Impacts of phosphorus losses

In spite of the notable phosphorus surplus in the Po River basin (see Figure 61), phosphorus load does not lead to significant impacts in Lombardy, aside from some cases of freshwater eutrophication affecting northern lakes such as Lake Como, Lake Iseo, and Lake Lugano (ERM, 2000). Eutrophication is one of the intertwined factors (water temperature, and absence of water mixing also play a role) leading to algae blooms that colour water and make it appear muddy and unappealing. Algal blooms are observed seasonally, for instance in Lake Garda and Lake Como (Rex, 2013). Visible algal blooms can affect tourism since lakes of northern Italy are among the most visited locations of the country.

In deep lakes, eutrophication can lead to oxygen deficiency in the bottom water layers and reduce mixing of water. For instance, during the last four decades, in the northern basin of Lake Lugano\textsuperscript{132} the water column was stagnant and anoxic below 100 m (Holzner, et al., 2009). Such changes in mixing behaviours and oxygen concentrations have negative consequences on the entire lake ecosystem.

\textsuperscript{131} The crown of a tree is the branches, leaves, and reproductive structures extending from the trunk or main stems.

\textsuperscript{132} It should be noted that in Lake Lugano specifically, the external nutrient load derives from anthropogenic (85%), industrial (10%) and agricultural (5%) sources (UNECE, 2007).
5.3.2 Causes of nutrient losses

The impacts related to nutrient load in the Po River basin waters are mostly due to agriculture, especially in the Lombardy plain area and, to a lesser but notable extent, to sewage water from household and industry in western Lombardy. Uncertainties remain regarding the quantification of the impacts of livestock and crop productions on human health, biodiversity and air quality (Region of Lombardy, 2006). Several studies have been carried out about negative effects possibly due to farming systems, but neither the nitrogen losses from agricultural lands nor the threats specifically due to nitrates released by farms are well known or quantified in Lombardy. In this section, as far as possible, only the causes related to agriculture are tackled.

Farming system and agricultural practices

The main cause for nutrient surplus in Lombardy is the excessive use of organic and mineral fertilisers and improper fertiliser application. The results of the assessment carried out by ISPRA by applying the model ISONITRATE (LIFE06 ENV/F/000158) clearly demonstrate how the risk of pollution due to nitrate is related mainly to the combination of the use of mineral fertilisers and the production of manure from livestock activities (ISPRA, 2014). As it can be clearly deduced from Figure 66, the contribution of the non-agricultural activities on the risk of nitrate pollution of aquifers is significant only in the highly populated areas around major cities (Milan, Bergamo and Brescia).

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Figure 66 – Contribution of different agricultural and urban sources to nitrogen load in Lombardy (HI: Hazard Index)\textsuperscript{133}

\textsuperscript{133} The amount of nitrogen (kg in a year) is divided by the entire municipal area
Lombardy was in 2011 the leading region in Italy for the use of fertilisers. About 100 000 tons of fertilisers, from both organic and mineral sources, were applied that year, representing 20% of national consumption (ISTAT, 2011a). The main nutritive elements provided by fertilisers use in Lombardy are nitrogen (143 000 tons), potassium oxide (52 000 tons) and phosphoric anhydride (46 000 tons) (ISTAT, 2011b).

High production of manure that locally exceeds the carrying capacity of the territory – Livestock production in Lombardy, with a strong presence in the south-east of the region, is amongst the most intensive systems in Europe (Fumagalli, et al., 2011; Delconte, et al., 2014). As a result, large amounts of livestock effluents are generated. The corresponding nitrogen amount is estimated to be around 130 650 tons per year, with 60% coming from cattle and dairy cows, 28% from pigs and 10% from poultry (ERSAF, 2011). Farmers tend to dispose of the effluents as soon as possible since manure storage capacity is limited (ERSAF Lombardy, 2011). A minimum storage capacity requirement of 3 to 6 months is required in NVZ (depending on the type of manure and livestock category), but it is sometimes not sufficient to avoid applying fertiliser during inappropriate periods. This situation stems from the imbalance between animal breeding activities and land used for animal feed production. In fact, in the plain area of Lombardy, grazing is not practiced, and the feed required for animal production is produced only in part (generally roughage) in the region’s farms. Hence the use of concentrates and protein feedstuffs imported from markets external to the region is widespread (Provolo & Riva, 2003).

Lack of accurate knowledge regarding the content of livestock manure – Cropping systems in Lombardy are mostly based on cereals and forage and generally conducted according to intensive practices in plain areas while permanent crops and grassland can be considered as extensive (Castoldi, et al., 2009a; Lynche-Solheim, et al., 2010; Fumagalli, et al., 2011; INEA, Region of Lombardy, Milan University, DEMM, SMEA, 2013). Since manure nutrient content and efficiency are not accurately known, farmers are inclined to apply manure in high amounts, often based on the maximum amount of nutrients that can be applied indicated in the Nitrate Action Programme or the Code of Good Agricultural Practices, in order to be sure that plant nutrient needs are well satisfied. Farmers may also apply mineral fertilisers on the top of manure (ISPRRA, 2014). According to the legislation, farmers have to prepare a nutrient management plan based on average figures of nitrogen content in manure but these data might not represent the actual characteristics of their stored manure. Indeed, the nitrogen content in the manure stored in a slurry tanker might be affected by several factors such as the season and possible stratification of the manure during the storage (since the mixing operations are not always done). Therefore a uniform manure application is rarely achieved. Farmers are inclined to attach limited importance to the amount of nitrogen content in manure and hence commonly over-apply manure. This tendency may lead to excessive amounts of nutrients provided compared to plants’ needs, which notably results in nitrogen surpluses in the soils of the plain area of Lombardy. Unused nutrients, particularly nitrogen in agricultural lands of Lombardy plain, constitute the nutrient surplus in the soils of the region. According to the Regional Agency for Agriculture and Forest Services of Lombardy (ERSAF, 2011), the nitrogen load originating from livestock manure exceeds 170 kg N/ha/year134 in almost all of the livestock production areas of south-eastern Lombardy. In some municipalities, the limits of 250 kg N/ha/year (limit stated in the derogation decision 2011/721/UE for 2012-2015135) or even of 340 kg N/ha/year (maximum limit for the non-vulnerable zones) are exceeded. Therefore, nitrogen potentially ends up in water bodies by run-off or leaching or in the atmosphere after microbial transformations. To overcome these

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134 Maximum annual limit of nitrogen from livestock manure that can be applied on land, or above which a 250 is required, as established by the Nitrates Directive

135 For the period 2012-2015, Lombardy has a derogation to spread in a quantity equal to maximum 250 kg/N per hectare per year from cattle manure and treated pig manure in NVZ provided that farmers fulfil specific conditions
concerns, some techniques have been developed both to improve farmers’ knowledge on nutrient content (Provolo & Martinez-Suller, 2007) and to use record-keeping systems that aim to predict nutrient content in storage tank (Provolo & Riva, 2006). However, these solutions have not been implemented by farmers as they do not see clear benefits for themselves and neither law nor incentives compel them to implement such solutions.

**Improper fertiliser spreading** – Techniques for spreading fertilisers in the region may not comply with the best practices. In addition to the possible use of inappropriate fertilisation techniques (e.g. splash plate without incorporation), the timing of fertilisation might not be agronomically efficient. Manure is usually applied twice a year, in spring and in autumn (Provolo & Riva, 2003). The autumnal application is likely to lead to nutrient losses. Indeed, soils may be left bare and thus are particularly exposed to rainfalls in winter, while until springtime, winter crops (wheat, barley, rye or triticale in Lombardy plain area) do not catch enough nutrients to balance the amount of nutrients provided.

**Irrigation practices** – To a lesser extent, water management for irrigation potentially contributes to nutrient surplus in Lombardy for two main reasons: first the source and timing of irrigation water, then the choice of irrigation systems. Lombardy is a major water consumer: 582 000 ha were irrigated in 2010 (58 % of the UAA), while the share of the irrigable surface area of the UAA is 69 % (Region of Lombardy, 2015a). It mostly concerns maize and rice productions that require a high amount of water. In Lombardy, only 3.5 % of the 31 000 irrigable farms use irrigation advice services (5.5 % of the irrigable area) (INEA, Region of Lombardy, Milan University, DEMM, SMEA, 2013). The irrigation systems in Lombardy are notably based on surface irrigation, including flooding for rice crops. That kind of system particularly contributes to transfers of nutrients alongside with run-off and leaching water (Fumagalli, et al., 2011; ISTAT, 2006). Moreover, more than 80 % of the irrigation water comes from surface water in Lombardy, which decreases the natural watercourse flow. Consequently, an inflow of shallow groundwater may occur locally and potentially enriches surface water in specific parts of the river with surplus nutrients from the soil through which the groundwater has filtered. Such phenomena are even more pronounced in summer when 50 % of agricultural lands require irrigation water. This leads to a greater river flow decrease and higher risk of nutrient transfers to surface water (Delconte, et al., 2014; Sacchi, et al., 2013).

**Environmental conditions**

Natural factors modulate the effects of farming systems and agricultural practices.

**Concentration of crop and livestock productions in plains of southern Lombardy** – The topography is a major determinant of the land use of the region and the distribution of the potential sources of nutrient surpluses. In the north of Lombardy, the mountainous areas are not appropriate for anthropic activities, thus upstream rivers and lakes are less subject to water quality issues. In southern Lombardy, while the upper plain is dedicated to industrial activities and urban settlements, the lower plain is primarily dedicated to agriculture production. Lakes and midstream watercourses close to industries and cities are subject to degraded quality due to wastewater discharge from plants and households and to high frequentation rate around the large lakes, which result in high nutrient loads that naturally end up in the Po River. Downstream water bodies also drain large amounts of nutrients into the lower plain, largely as a result of the intensive agriculture (Raggi, et al., 2007; Civita, et al., 2007).

**Occurrence of rainfalls and droughts in Lombardy** – The average rainfall in Lombardy is 827 mm (ISTAT, 2009) and the two rainy periods are spring and autumn (Fumagalli, et al., 2011). Depending on the timing of fertiliser application, this rainfall can cause run-off or leaching of recently spread nutrients. In addition, exceptional rainfalls may occur in summer following periods of drought, resulting in floods over some areas of the Po River basin, including agricultural lands. The high level of urbanisation in the
region and the resulting lower soil permeability in densely built areas increases the flood risk (Raggi, et al., 2007). Such floods lead to water saturation of soils that might increase existing transfers of excess nutrients by leaching and run-off. Moreover, replenished shallow water tables create anaerobic conditions favourable to denitrification reactions, resulting in higher emissions of nitrous oxide. This process is intensified in the Po River basin in summer with increasing temperatures that enhance microbial production of nitrous oxide and lower gas solubility (Sacchi, et al., 2013).

**Stagnant meteorological conditions at the footstep of Alpine mountains** – The areas at the footstep of the mountains are characterised by frequent occurrences of stagnant meteorological conditions that partly explain high levels of pollutants in the air of the northern Po River basin (Thunis, et al., 2009). Air pollution in the upper plain is not only from industrial and urban origins but also from agricultural origin as ammonia and nitrous oxide released from agricultural lands and water bodies in the lower plain can disperse into the atmosphere of nearby areas (Pinho, et al., 2008; Ferm, 1998; Dosio, et al., 2002).

**Dense and complex hydrology network that transfers nutrient surplus** – The dense water network in Lombardy is partly responsible for highly populated settlements, industrial activities and agricultural production (ARPA Lombardia, 2013; Region of Lombardy, 2006). These anthropic factors include cities, industries and farming contribute to the nutrient content of these water resources. However, further research is required to better understand the impact of each anthropic factor on water quality and to better analyse the nutrient transfers and resulting impacts in saturated areas, which the natural complexity of the hydrology network in northern Italy makes all the more difficult. Aquifers in the Lombardy region show an increasing complexity from north to south (Region of Lombardy, 2008). Up until now, several studies have pointed out the movement of groundwater tables from the north to the south of the Po River basin, the extent of interactions between groundwater, and surface water, and provided evidence that they can cause nitrate enrichment of river water in some watersheds of the Po River basin (Delconte, et al., 2014; Sacchi, et al., 2013). Nitrogen in groundwater also induces nitrous oxide emissions due to denitrification that results from oxidation-reduction reactions with pyrite in anaerobic conditions. In the lower plain, denitrification is estimated to remove about 40–60 % of the initial nitrates from groundwater in maize fields, and up to 80 % in rice fields (Sacchi, et al., 2013), which also prevents groundwater eutrophication. However, knowledge is still missing about water-related phenomena, including recharge and flows of water tables, denitrification in groundwater, and emergence of springs (Cremonesi, et al., 2011; Laini, et al., 2011).

**Loamy soil texture and compacted structure** – Lombard soils mostly consist of loams, from silty-loam in the south-eastern part to sandy-loam in the western part of the region (Fumagalli, et al., 2011; Castoldi, et al., 2009b; Laini, et al., 2011). Sand texture presents high permeability. During rainfalls, such texture contributes to water infiltration through soils which in turn carry excess nutrients to groundwater (Region of Lombardy, 2006). This process particularly occurs in plain areas of western Lombardy dedicated to cereal crops and contributes to nitrate contamination in groundwater (Delconte, et al., 2014; Sacchi, et al., 2013). Lombardy also presents highly compacted soils (Joint Research Centre, 2012b), enhanced by silt texture that may be favourable to compaction and slacking. Compaction increases the risk of run-off which is particularly high in upstream parts of the Po River basin and especially in the province of Pavia and in the south of Brescia (Region of Lombardy, 2006).

### 5.3.3 Costs of the environmental and health effects

#### 5.3.3.1 Socio-economic description of the region

With about 10 million inhabitants in 2014 of which 70 % live in the urban area of the regional capital Milan, Lombardy represented 16.4 % of the Italian population (Eurostat, 2015d). With a GDP of € 358 billion in 2013, Lombardy contributed to 22 % of the national GDP in 2013. The regional GDP per capita
was 36% higher than the national average (Eurostat, 2015f). On a wider scale, more than 40% of the Italian workforce is employed in the Po River basin\textsuperscript{136} and it produces nearly 40% of the national GDP (Lynche-Solheim, et al., 2010).

The regional economy is mainly based on the industrial sector and tertiary services. Although agriculture contributes to the regional added value to a lesser extent, Lombardy is one of the leading agricultural regions of Italy and accounted for about 10.7% of the national agricultural added value, with 8% of the total national territory in 2013 (INEA, 2014; Eurostat, 2014c). In particular, Lombardy contributed 26.5% and 14% of the national livestock and cereal productions (in euros) respectively in 2013 (INEA, 2014; Eurostat, 2014c). Lombardy is also a strategic area for tourism, thanks to its water resources, including five of the largest Italian lakes and numerous rivers including the Po River, and to a diversified landscape, from mountains to hills and plains (European Commission, 2012c; Agenzia Nazionale per il Turismo in Italia, 2012; Region of Lombardy, 2008).

5.3.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in Lombardy and in the Po River basin, a range of literature was reviewed. Based on the classification of economic damages caused by environmental impacts described in Annex 14, costs found in the literature for each damage category (i.e. CRC, UVD, etc.) are presented below along with explanations on how these costs have been estimated.

It is important to note that all cost data reported in this section are taken from the primary studies found in the literature, and expressed in the value of the year when the study was conducted.

Note that some data provided in this section may cover a slightly different area from Lombardy. Depending on data availability and representativeness, the economic damage assessment that is presented in this section will as far as possible refer to the restricted perimeter of Lombardy and otherwise to the broader scope of the Po River basin.

Finally, it is important to underline that whilst agriculture is an important source of nutrient surplus it is not the only one, and the economic damages from nutrient surplus cannot be exclusively attributed to agriculture.

Clean up and restoration costs (CRC)

In Lombardy, excessive nitrogen load in water bodies is a major environmental problem and compromises water quality status. Actions have been taken through several Plans and Programmes in the region in order to reduce the pollution and the subsequent negative impacts caused by nitrogen, including nitrates.

In 2002, the Lombardy region signed the Framework Programme Agreement with the Ministry of Environment. Consequently, as reported by the Environment and Mobility Directorate of Milan in 2005 (Comune di Milano, Direzione Centrale Ambiente e Mobilita, 2005), interventions have been implemented with the purpose of, among other objectives, restoring the quality of surface and ground waters so as to make them suitable for drinking supply, wildlife and bathing, and minimising the pollution of water bodies caused by both nitrates from agriculture sources but also by dangerous substances and

\textsuperscript{136} Po River basin is spread over four Italian regions among which Lombardy.
non-purified wastewater from industrial and urban activities. The report mentions that about € 735 million was allocated for the Agreement actions, including restoration.

**Preventive measures to reduce nutrients concentration in water**

More recently, the “Programme for the protection and use of water” (PTUA\(^{137}\)) was approved at the Po River basin scale in 2006, pursuant to the River Basin Management Plan established in accordance to the Water Framework Directive. Various measures of this Programme aim at restoring good quality status of water bodies in the Lombardy Region through the designation of sensitive and vulnerable areas and the definition of intervention policies, the identification of areas for water conservation intended for human consumption, and the conservation and improvement of water bodies and associated ecosystems. The investment required for the implementation of above measures provided by the PTUA and the achievement of water quality objectives was estimated at € 4.5 million (ERSAF, 2011; Region of Lombardy, 2006).

As a measure from this regional programme regarding nitrate pollution more specifically, the Emergency Action Plan\(^{138}\) has financed some infrastructure and engineering investments that are expected to improve the management of livestock manure. Among the projects are: the formation of an agro-energy consortium contributing to manure transfer between farms and energy production plants, the construction of plants that favour nitrogen abatement or the construction of facilities that enhance nutrient efficiency through biological process. The amount of the financial contribution to eligible expenses depends on several criteria (type of project, size of farms, cooperatives or companies, etc.) (Republica Italiana, 2008; ERSAF, 2011).

**Nutrient removal**

In 2012, a regional reference specifies that among public expenditures for water resources, € 4.4 million were dedicated to interventions targeting the improvement of water quality in the water systems (ARPA Lombardia, 2002).

**Cleaning air and preventing air pollution**

Another primary environmental issue in Lombardy is air pollution. According to an assessment of environmental expenses in Lombardy in 2002, a total of € 2.4 million was dedicated to cleaning the air and protecting it from pollution caused by anthropic activities, including agriculture, in addition to guaranteeing physical security in industry (ARPA Lombardia, 2002). The literature review conducted in this study has pointed out the low data availability and the need of carrying out further research in this area.

More generally, with regard to all impacts affecting the region, the Lombardy region has established several regional parks in polluted areas that had been significantly degraded, with the purpose of restoring their natural characteristics. Although quantitative data have not been identified, there is no doubt that park creation and management require important economic resources and funds (Comune di Milano, Direzione Centrale Ambiente e Mobilita, 2005).

To sum up, Table 34 presents the clean-up and restoration costs that have been found for the Lombardy region or for the Po River basin.

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\(^{137}\) In Italian, PTUA : Programma di Tutela e Uso delle Acque

\(^{138}\) In Italian, Piano Straordinario di Interventi Urgenti
Table 34 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (M€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure for clean-up and restoration purposes</td>
<td>All actions implemented following the Framework Programme Agreement, including interventions for water quality restoration</td>
<td>735</td>
<td>Lombardy</td>
<td>2005</td>
</tr>
<tr>
<td></td>
<td>Actions for water quality improvement in the water systems</td>
<td>4.4</td>
<td>Lombardy</td>
<td>2002</td>
</tr>
<tr>
<td></td>
<td>Implementation of measures related to the “Programme for the protection and use of water” (PTUA)</td>
<td>4.5</td>
<td>Po River basin</td>
<td>2006</td>
</tr>
<tr>
<td></td>
<td>Infrastructure and engineering aiming at reducing effluent N content, according to the “Emergency action Plan”</td>
<td>67</td>
<td>Po River basin</td>
<td>Reported in 2011 for the 3 year period 2005-2008</td>
</tr>
<tr>
<td></td>
<td>Environmental measures including cleaning the air and preventing air pollution</td>
<td>2.4</td>
<td>Lombardy</td>
<td>2002</td>
</tr>
</tbody>
</table>

Use value damages (UVD)

Eutrophication in coastal areas of north-western Adriatic Sea is an important environmental problem that is due not only to pollution from the Venice lagoon watershed but also to discharge of nutrient surplus from the Po River. The resulting harmful algal blooms affect local economy through reduced revenues in the tourism and fishery sectors. On the one hand, negative effects of algae blooms, such as intense colouring of waters, repetitive foul-smelling odours and formation of foam on surface water, decrease the attractiveness of coastal areas. On the other hand, negative effects of toxic algae blooms lead to contamination of shellfish and death of fish species, which directly affects the fishing industry.

In 2006, Hoagland & Scatasta estimated the average annual economic effects of harmful algal blooms in the EU for the year 2005 at around € 166 million for commercial fisheries and € 720 for recreation and tourism (Hoagland & Scatasta, 2006; ARPA Veneto, 2011). The project ECOHARM estimated that the socio-economic impact of harmful algal blooms in Italy is around € 115 million per year, including the impact on tourism (€ 106.9 million) and in a lesser extent on commercial fisheries (€ 6.3 million) (EEA, 2005c).

Climate change is a global issue mostly related to industrial and urban activities rather than to the N₂O emitted as a result of nutrient surplus. Nevertheless, it should be underlined that climate change will affect alpine areas such as the north of Lombardy. Indeed, climate change contributes to increasing temperature and the retreat of glaciers and snowlines, thereby leading to a decrease in the attractiveness of alpine resorts and to greater costs for maintaining and preserving winter tourism infrastructures. An economic assessment conducted by Bigano et al. in 2007 estimated the loss of direct incomes from winter tourism in Lombardy at € 29.11 million¹³⁹ by 2030, which represents a decrease in incomes by 7.1 % with respect to the baseline scenario of “no climate change” established in 2006 (Sgobbi & Carraro, 2008).

¹³⁹ The loss is estimated by transposing the % reduction to the 2006 income
Table 35 presents the use value damages that have been found for the Lombardy region or the Veneto region also located in the Po River basin.

**Table 35 – Use value damages found in literature**

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Damage estimation</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduced revenues in tourism, fishery sector etc.</td>
<td>Decrease in tourism activities due to algae blooms from eutrophication</td>
<td>€ 106 900 000</td>
<td>Italy</td>
<td>Per year</td>
</tr>
<tr>
<td></td>
<td>Decrease in fishery activities due to algae blooms from eutrophication</td>
<td>€ 6 300 000</td>
<td>Italy</td>
<td>Per year</td>
</tr>
<tr>
<td></td>
<td>Loss of winter tourism due to climate change caused by GHG emissions</td>
<td>€ 29 110 000</td>
<td>Lombardy</td>
<td>Reported in 2007 for the prospective period 2006-2030</td>
</tr>
</tbody>
</table>

**Passive use value damages (PUVD)**

In addition to environmental issues, algae blooms in marine water and emissions to the atmosphere can also threaten human health and biodiversity.

Toxic algae blooms along the Adriatic coast give rise to public safety concerns and disrupt water ecosystems (Andricevic, et al., 2011).

Climate change and the related increase in temperatures and in floods and landslides partly caused by N\textsubscript{2}O emissions worsen biodiversity losses (Sgobbi & Carraro, 2008). The resulting occurrence of forest fires in alpine areas and displacement of natural ecosystems in plain areas represents passive use value damages that require further research to be economically estimated. Floods and landslides also injure human lives and contribute to the spread of water-related diseases.

Thunis, et al. (2009) estimated that atmospheric fine particulate matter, partly consisting of NH\textsubscript{3}, could be responsible for a loss of ten months of human life expectancy. However information was missing to economically quantify this loss of human life expectancy while taking into account social-economic characteristics of the Lombardy region.

Although these phenomena are to some extent related to the nutrient surplus in Lombardy, very limited quantitative information is available for assessing their magnitude in monetary terms.

**Policy action costs (PAC)**

Policy actions have been implemented to protect the environment of the Lombardy region. This includes scientific monitoring plans that have been drawn up to ensure good status of water bodies and control, for instance, shellfish toxins and plankton composition. As part of the Programme for Water Use and Water Protection established in parallel with the Water Deal signed by the Lombardy region, a monitoring system of watercourse quality has been developed. This tool aims at considering not only the chemical and physical status of surface waters, but also the biological and hydro-morphologic characteristics of the watercourse (Region of Lombardy, 2008). In addition, the project ECOHARM estimated that the monitoring and management costs of harmful algal blooms in Italy were around € 1.7 million per year (EEA, 2005c).
Table 36 – Policy action costs found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Monitoring plans</td>
<td>Monitoring system of watercourse quality as part of the Programme for Water Use and Water Protection</td>
<td>N/A</td>
<td>Lombardy</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Monitoring and management of harmful algal bloom</td>
<td>€ 1 730 000</td>
<td>Italy</td>
<td>Per year</td>
</tr>
</tbody>
</table>

Finally, it can be mentioned that the reduction of nutrient discharges from Adriatic rivers, particularly from the Po River, would decrease the appearance of algal blooms along the coast and reduce economic losses related to lower exploitation of marine resources such as tourism and fisheries. Andricevic et al. (2011) have proposed a rough estimate for the aggregated costs of N load reduction actions in the Po River basin. Considering a reduction cost of nitrogen loads of € 2.4 per kg of nitrogen\(^\text{140}\), they estimated the total cost to completely remove nitrogen load in the Po River basin to be around € 391 million (Andricevic, et al., 2011).

5.3.4 Good practices to reduce nutrient losses at farm level

In Lombardy, the identified good practices include measures aiming at decreasing the local source of pollution by controlling the geographic distribution of the breeding activities, transferring manure, adapting feeding strategies, improving the nutrients fertilisation management plans for all agricultural sites and processing manure to ease its transfer and use. Another type of good practice relates to the reduction of the amount of nutrients lost during housing and storage by improving manure collection and covering the slurry tanks. A third type of good practice relates to the reduction of the nutrient losses during application by using the most suitable application techniques and improving irrigation systems. Lastly, measures such as the use of adequate tillage techniques and soil coverage with catch crops would help to increase the nutrient retention capacity of soil and decrease the nutrient losses.

5.3.4.1 What has already been done in the region

In Lombardy, 62 % of the UAA has been designated as a Nitrate Vulnerable Zone (ERSAF Lombardy, 2014; Region of Lombardy, 2014a) and actions to reduce the pollution and negative impacts caused by excessive nitrogen loads in water bodies have gone through several regional plans and programmes.

Numerous measures implementing the Nitrates Directive and corresponding good farming practices have been implemented in the Nitrates Vulnerable Zones located in Lombardy under the Nitrates Action Programme 2012-2015 (Giovanazzi, 2014), and some of them have been extended throughout the region. These actions, like the minimum storage capacity requirement and the limitation of the maximum amount of nitrogen that can be applied to crops, are aimed at restoring the quality of water bodies contaminated by nitrates from agricultural activities and preventing such pollution. Under this programme, supervision and control of the compliance with the requirements regarding storage and use of manure is performed by the Region. As part of this activity, each year a sample of farms is selected for which an onsite inspection is carried out. In addition, with an amendment to the Regional Law

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\(^{140}\) The figure used by Andricevic et al. was obtained by estimating the cost of the agricultural measures that contributed to the reduction of the nitrogen loads in the Baltic Sea weighted by the contribution of each measure in the nitrogen load reduction (Schou, et al., 2006).
31/2008, in 2012 the Lombardy Region has introduced administrative sanctions at the regional level, in the case of non-compliance with the Nitrates Directive rules.

The “Programme for the protection and use of water” (in Italian - PTUA) was approved pursuant to the River Basin Management Plan (under the Water Framework Directive). Various measures under the Programme are aimed at restoring the good quality status of water bodies in the Lombardy Region.

Rural Development Plans developed under the Common Agricultural Policy are also a way in which actions to decrease nutrient losses on-farm and increase resource efficiency can be promoted. Specific measures have provided financial support for adopting new technology, updating equipment, changing crop rotations, ploughing using different methods, etc.

5.3.4.2 **Good practices to reduce the nutrient losses in livestock production**

*Control the geographic distribution of livestock*

In the following sections, a number of good practices are described to reduce nutrient losses. These practices concern livestock management, such as the feeding practices and the housing systems, and manure management such as the storage systems. However, the nutrient surplus in the region is also due to the high density of livestock in the territory. Indeed, as mentioned in 5.3.2, Lombardy counted 4.66 million pigs, 1.53 million beef and dairy cattle and 32.4 million poultry in 2014 (Region of Lombardy, 2014a). The average livestock density was 2.77 LSU/ha in 2010 (Eurostat, 2013i). Currently, the number of pigs is globally stable while the number of cattle has decreased by 20 % since 2007 (Eurostat, 2014b). However, the number of pig farms decreased between 2000 and 2010 while the average number of pigs per farm grew by 500 %, illustrating the intensification of the production systems (Bernardelli, 2012). Consequently, the quantity of manure produced exceeds the carrying capacity of the territory in some areas of the region in term of nutrient budget, that should be balanced, and in term of compliance with the requirements of the regional Nitrate Action Programme if the farm is located in NVZs.

Thus, in addition to good livestock management practices, the control of the geographic distribution of livestock through the limitation or the decrease of livestock density and production intensity in areas presenting an excess of nutrients would be needed in the long term in order to work towards a sustainable reduction of nutrient surplus.

In the short term, limiting or reducing livestock density in intensive areas and farms may cause major acceptability issues on the farmers’ side. Indeed, it would lead to a reduction of the quantity of meat and animal products and thus economic losses. Larger farms may be impacted less as they benefit from the economies of scale and have better access to capital. Arata, et al. (2013) showed that decreasing the number of dairy cattle to the point that the nitrogen production is in compliance with the nitrogen applicable on farms according to the Nitrates Directive results in a loss of income by up to one-third in Lombardy. This is due to a radical change of production systems in farms that produce a surplus of manure. Indeed, the milk quota was no longer binding because the nitrogen limit was more restrictive. Hence, this induces a decrease in milk production. The reduction in sales of livestock is a secondary cause (Arata, et al., 2013). The loss of revenue would be higher for farmers within the areas concerned by quality marks such as the PDO and the PGI marks where products may have a higher value-added than outside these areas. The farms may also partially reallocate their land, from fodder maize to

141 170 kg N/ha/yr, the derogation is not taken into account. Disposal of manure is not possible.

142 Note that on 31 March 2015 the EU milk quota regime has come to an end.
temporary grass. The reduction of concentrated feed purchases and veterinary services do not compensate the reduction of revenue and the possible additional expenses (Arata, et al., 2013).

Also, it may not be well accepted by the consumers, both national consumers and tourists. Indeed, many livestock products that are part of traditional Italian gastronomy such as Bresaola, Prosciutto and Grana Padano cheese are produced in the region. If the number of livestock decreases in these production areas, it may result in an increase of the price of these products. For context, in 2013, the livestock production (meat and livestock products) represented € 4.6 billion, including € 2.6 billion for the meat sector. Milk production accounts for € 1.7 billion and pig production € 1.2 billion (Region of Lombardy, 2015a).

**Transfer manure**

Considering that 62 % of the UAA in Lombardy has been designated as a Nitrate Vulnerable Zone (ERSAF Lombardy, 2014) which limits the content of N from manure that can be applied, this measure could be appealing to farmers whose limits are likely to otherwise be exceeded (limit stated in the derogation decision 2011/721/UE for 2012-2015).

Currently, only one study that analyses the effect of the Nitrates Directive for Italian dairy farms takes into account the options of creating a manure market (Arata, et al., 2013). The study analyses the effect of the transfer of the surplus of manure from dairy cattle in Lombardy and Emilia-Romagna to farms with a negative surplus. The quantity of manure above the quantity that can be applied in the field according to the Nitrates Directive is transferred to another farm. Since the threshold for the application of livestock nitrogen equals 340 kg N/ha in areas other than NVZ in Italy, farms outside of NVZ could absorb the manure coming from a vulnerable zone. Moreover, farmers with a surplus pay to get rid of manure, including the transportation costs from its farm to the receiving farm. As a result, the farmers that can receive the manure have increased their income due to the reduction of mineral fertiliser bought and the payment for receiving the manure. The income of the farmers with an initial manure surplus has decreased by up to 10 % which is far less than the decrease of income in the case of reduction of the herd (Arata, et al., 2013). However, the results can vary widely according to the manure price and the transportation distances. Beyond 20-30 km, the economic viability of importing comes into question. In addition, transporting manure may contribute to health issues related to transfers of pests or pathogens.

**Adapt feeding strategies to reduce nutrient load in manure**

The potential of further developing measures related to the adjustment of the feed ration is limited as it is already well developed in Lombardy. In the northern Italian regions, cattle production is characterised by the absence of grazing (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010; Provolo & Riva, 2003). About 90 % of the dairy cattle farms use the Total Mixed Ration technique that consists in weighing and blending all the feedstuffs required (forages, grains, supplements, vitamins and minerals). Cows are mostly grouped according to production phase. By contrast, improvements are still possible to increase the assimilation of nutrients by decreasing the crude protein content. Indeed, the mean content of crude protein is 15.3 % DM and could be reduced without compromising the milk yield (Italian Ministry of Environment, Land and Sea; Italian Ministry for

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143 For the period 2012-2015, Lombardy had a derogation to spread in a quantity equal to maximum 250kg/N per hectare per year from cattle manure and treated pig manure in NVZ provided that farmers fulfil specific conditions.

144 170 kg N/ha/yr, the derogation is not taken into account.

145 Dry Matter
agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia-Venezia Giulia, 2010). For pigs, the average crude protein content in diet is 14.9% (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia-Venezia Giulia, 2010). Decreasing the level of crude proteins in the diet can be possible, depending on the type of the final pig products. Indeed, in the region, the pigs can be bred for the production of very high quality ham such as Parma ham. Such pigs require specific feeding practices with limited flexibility regarding the diet composition and quantity. Therefore, a high variation in the protein content may be difficult without changing the yield or the quality of the meat produced required by the standard.

**Improve the collection of manure from livestock housing units**

This measure mostly concerns pig production. In the northern regions of Italy, including Lombardy, internal concrete floors with an external slatted alley is the most common type of housing for growers and finishers. Fully slatted floors and partially slatted floors represent each 22% of the housing systems used. Gestating sows and weaners are mostly on flat decks with fully slatted floors above a slurry channel (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia-Venezia Giulia, 2010). It appears that slatted floors are more broadly used in larger farms than in smaller farms (Provolo & Riva, 2002). Hence, a high share of the housing can be concerned by the measure.

For cattle production in the northern regions of Italy, housing on fully slatted floors is the main system (61% of heads) used for dairy followers and calves for beef production, followed by loose housing on a layer of bedding only in the resting area. For dairy cows, the main system is cubicle housing (53%) and loose housing on a layer of bedding (deep litter) (42%) (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia-Venezia Giulia, 2010). The use of litter can be implemented in a high amount of dairy farms but requires a consequent change in terms of manure management. For litter-based housing, the litter must be kept clean and dry and be regularly removed.

**Cover slurry tanks to prevent ammonia emissions**

In Lombardy, the slurry tanks are not always covered, even among newly built tanks. In 2010, only 33% of the slurry tanks were covered in Lombardy (Eurostat, 2013c). By limiting the contact between manure and atmosphere, covering slurry storages can reduce GHG and ammonia emissions to air from 40% (floating cover) to 80% (rigid store cover) (ADAS, 2011). Fixed covers are more efficient in emission reduction and in diverting rainfall water. Flexible covers can be implemented in another way by using plastic floating cover or chopped straw for instance.

The implementation of this measure on existing slurry tanks may pose logistical challenges and may not always be possible due to insufficient structural support, i.e. the existing slurry tank not being able to hold the added weight of a rigid cover. The cost of this measure depends on the type of cover. For instance, the cost of a plastic sheeting cover is € 1.25/m³/year while the cost of a tight lid roof or tent structure is € 8/m³/year (Co-chairs of the Task Force on Reactive Nitrogen, 2011). Note that covering cattle slurry may be relatively less beneficial as natural crust that often develops gives effective ammonia emission reductions.

**Process manure to eliminate, extract or concentrate the nutrients**

Processing manure is an interesting option for recycling manure and ease its transfer and use. The following techniques have been identified, amongst others, as relevant for the region.
Separate liquid from solid fraction

According to farmers unions and local authorities responsible for permits, solid-liquid separation techniques are widespread in the pig farms in northern Italy where nearly 40% of pig farms are equipped with a separation device. By contrast, it only concerns 5.5% of cattle farms (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010).

The most efficient separator allows for the removal of up to 70-75% of the phosphate from the solid fraction. However, this technique is seldom used considering the high investment (from € 30 000 to € 120 000) for maintenance costs and operational difficulties. Other techniques such as screw presses and sieve drum presses, which are slightly less effective, are usually preferred by Italian farms due to the simplicity and reliability of the equipment which can be adapted to a wide range of farms (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010).

Encouraging this processing technique is a good practice in Lombardy. Farmers are aware of the existence of this technique from which they could highly benefit. The renewal of the financial support provided by the Rural Development Programme 2007-2013 (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010) for the next Plan could encourage farmers to invest in that technique.

Nutrient concentration in digestate from anaerobic digestion treatment

Increasing the use of processed manure is another option that would allow a better management of livestock manure in the Lombardy region. One of the options for processing manure is the digestion in biogas plants.

Considering the amount of manure produced each year and the economic incentive for the biofuel supply sector implemented in 2001 (Green Certificates), the number of biogas plants has recently increased in Italy. In 2013, 40% of the biogas plants were concentrated in Lombardy, representing 361 plants with a total production of 294 MWe. The average energy production for biogas plants is 814 kWe. The most common type of plant produces between 750 and 1 000 kWe (57% of the plants), explained by the fact that the former maximum threshold for obtaining public aid for investment was 1 000 kWe. Only 4.4% of the plants exceed this size, though they represent 16.7% of the electrical energy produced (INEA, Region of Lombardy, Milan University, DEMM, SMEA, 2013). Biogas from livestock manure represents 49% of the bioenergy produced in 2011 (Adani, et al., 2014).

The investment costs of plants range from € 250 to € 700 per cubic meter or € 2 500 to € 7 500 per kW installed electrical power in co-generation. In Bologna in the Emilia-Romagna region, the investment cost for an anaerobic co-digestion plant using both manure and biomass (2 400 m³) is € 800 000 (Global Methane Initiative, 2007). In 2009, the value of the subsidies from green certificates for the electric energy sold to the grid was € 280 / MWh for the biogas produced by agricultural waste, breeding and forestall waste (ETA-Florance Renewable Energies; Environment Park Parco Scientifico Tecnologico per l'Ambiente).
In June 2012, the Italian government published a new Intergovernmental Decree revising the amount of the subsidies. Among other changes, the production of energy in small plants (< 5 000 kWe) receives more subsidies than in bigger plants, and livestock manure and agriculture waste are preferred to energy crops (Bolzonella, Fatone, & Cecchi, 2013).

While these subsidies enhance the production of bioenergy, it also results in competition for land use. Indeed the production of energy crop, namely maize, has significantly increased, to the detriment of crop used for feed that have to be imported as a consequence. In the Po Valley, the area of maize for energy accounted for 0.4 % of the maize area while it amounts 10.3 % in 2012. In Lombardy, the area cropped with maize for biogas production reached 39 071 ha in 2012, i.e. 18.2 % of the total maize area. The increase of feed costs that has consequently decreased the profitability of the livestock sector in Italy has also enhanced the development of the production of crops for energy (Mela & Canali, 2014). Therefore, this measure can be considered a good practice only if it does not entail a detrimental land use change. The increase of the production of intensive crops such as maize and land use change from pasture or extensive cropland to very intensive cropland would increase the risk of nutrient losses, in particular leaching.

Acidify slurry to limit the transformation of ammonium to ammonia

Several technologies are available for acidification of slurry, which shifts the ammonia-ammonium balance towards ammonium, and thereby reduces ammonia emissions to the air. Acidification can also be used during storage. The greatest effect is achieved by acidifying the slurry already in the stable, whereas there will be no reduction of methane emissions if acidification takes place immediately before slurry application in the field. Since the soil in Lombardy is generally not acidic, this technique would likely not cause significant acidification issues (European Commission, 2003).

Remove nitrogen from manure

Currently, the nitrification/denitrification treatment systems are used in a limited but increasing number of farms in Lombardy. A pilot project in Northern Italy tested the nitrogen removal from the liquid fraction of digested agro-waste from a pig farm through a nitrification/denitrification process by anaerobic ammonium oxidation. The project showed that up to 96 % of the nitrogen could be removed (80 % on average). N\textsubscript{2}O emissions accounted for between 3 % and 24 % of the total N removed, depending on the C/N ratio (Scaglione, Lorenzoni, Ficara, Cannziani, & Malpei, 2013). This process is not operational yet. Additional tests are necessary to control and decrease the N\textsubscript{2}O emissions that occur during the process. The fate of the nitrogen removed should also be further studied to understand the potential trade-offs. Nonetheless, this type of processing may be an interesting option that should be further explored.

5.3.4.3 Good practices to reduce the nutrient losses in crop production

Improve fertilisation management plans for all farms

The rules for the agricultural use of livestock manure and the limits on the maximum quantity that can be spread are defined in the Decree approving the Nitrates Action Programme published in 2011 (Deliberazione Giunta regionale 14 settembre 2011-n.IX/2208). According to the Nitrates Directive, and in line with the Regional Act, the maximum amount of N manure applied in the field must not exceed

146 Ministerial Decree of July 6, 2012 establishing new procedures for supporting electricity generation by plants using Renewable Energy Sources (other than photovoltaic ones).
170 kg/ha/yr in NVZ and 250 kg/ha/yr in Lombardy\textsuperscript{147} for cattle and pig farms that request it (European Commission, 2013d).

The Regional legislation already requires farmers to establish a fertilisation plan for the application of manure. Indeed, under the Decree approving the Nitrates Action Programme, all livestock farms located in Lombardy must seek authorisation to spread manure, except for small-sized farms\textsuperscript{148} and farms that produce only solid manure. Farmers must submit their manure management plans to the municipal administration to obtain the authorisation for the agricultural utilisation of manure, specifying the management and infrastructure plans to support the application (Provolo & Riva, 2003). Including inorganic fertilisers in the preparation of such a plan would raise awareness on the efficient use of organic and mineral fertilisers when they are applied together, or identifying situations when applying only mineral fertilisers is better, for instance for economic reasons. Including phosphorus in the fertilisation management plan is essential to decrease the amount of fertiliser applied. The rules implementing the Nitrates Directive in the region, with specific regard to the derogation conditions, already requires that there shall be no application of phosphorus to crops in the form of chemical fertilisers in the NVZ (Region of Lombardy, General Directorate of Agriculture, 2012). The participants of the regional conference\textsuperscript{149} have identified the balance of P application as a good practice to reduce the nutrient surplus.

**Use adequate tillage techniques to limit nutrient leaching**

In 2010 in Lombardy, conventional tillage was used on 87% of the arable land while conservation tillage represented only 3.3% of the arable land (Eurostat, 2013d). Extending the use of conservation tillage or even no tillage in the regional farms can be an efficient technique with a potentially strong effect on nutrient leaching on a regional scale. Lombardy soils mostly consist of loams, from silt-loam in the south-eastern part to sandy-loam in the western part of the region (Fumagalli, et al., 2011; Castoldi, et al., 2009b; Laini, et al., 2011). The soil texture in Lombardy shows a high bulk density (Joint Research Centre, 2012b). Hence, conservation tillage or no tillage could be an interesting option for implementation, in particular in the western part of Lombardy. Indeed, the soils of this area are sandier than the south-eastern part of the region and tillage has only a limited effect for reducing the nitrogen emissions to air in sandy soils (Rochette, 2011). Therefore reducing tillage in this region would allow increasing soil organic matter content while improving its structure. These elements increase the soil water retention capacity and thus decrease the risk of leaching. In the south-eastern region, the implementation of conservation tillage is possible, but no tillage at all is not recommended.

Non tillage techniques may be less easy to implement than conservation tillage. Indeed, it may require an important reorganisation of the field work since manure is usually spread before tillage and incorporated into the soil through ploughing (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010). It may also require investment in machinery. In addition, tillage is a common strategy to fight weeds in a crop field as it buries weed seeds in the deeper layers.

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\textsuperscript{147} For the period 2012-2015, Lombardy has a derogation to spread in a quantity equal to maximum 250kg/N per hectare per year from cattle manure and treated pig manure in NVZ provided that farmers fulfil specific conditions.

\textsuperscript{148} 8 tonnes of live weight if cattle or pigs, 3 tonnes of live weight if birds or small animals

\textsuperscript{149} As part of the dissemination part of the project regional conferences were held in selected regions. One was organised in Milan, Italy in November 2014.
of the soils thus limiting the germination. Non tillage techniques require specific attention to weed development and the implementation at farm level of new strategies to fight weeds.

Use appropriate manure application techniques

The Italian code of good agricultural practices\textsuperscript{150} suggests a range of measures related to manure application: band spreading, application of slurry diluted with irrigation water (e.g. with low pressure sprinklers or drip lines), trailing hoses, shallow and deep injection, etc. (Italian Ministry for agriculture, Food and Forestry Policies, 1999). Although a “fast incorporation” of manure is recommended, the Italian code does not provide requirements regarding the specific timing of incorporation.

Although participants from the regional conference recognised that manure incorporation is a good practice to reduce nutrient losses, the large majority of Italian farms do not incorporate immediately solid manure or slurry after application (resp. 96 % and 99 % in 2010) (Eurostat, 2013e). In northern Italy, manure is usually spread before tillage and incorporated into the soil with ploughing. Hence, the potential of improvement regarding incorporation is very high.

The appropriate application technique may vary according to soil type and crop. When the manure is applied before seeding, band spreaders (followed by incorporation) or injectors (open or closed slot) can be used. Band spreaders drag perforated hoses behind them, from which slurry is applied close to the ground. Injection systems slit the soil open and inject the fertiliser at different depths. On grassland, using a trailing shoe spreader helps provide uniformity of spreading and lowers emissions. Some band spreading or closed slot slurry injection machines for top dressing applications are also available. The Feedback from the regional conference highlights that consortium or cooperative formation is essential to tackle the high investment related to the purchase of such equipment.

Cover with catch crops

According to the Nitrates Action Programme implementing the Nitrates Directive in Lombardy, it is required in NVZ to reduce nitrogen leaching risk with cover crops or other suitable measures included in the Code of Good Agricultural Practice. In 2010, one fourth of the arable land was not covered during winter in Lombardy (Eurostat, 2013f). Increasing the area with cover crops could significantly decrease the nutrient loss.

This measure is adapted in the western part of the region because the effect of catch crops is larger on sandy soils than on other soils (Børgesen & al, 2013). The estimated cost for a hectare of cover crop is € 84 to € 180 (including all the machinery operations).

The use of catch crops but also legumes is recommended by the ERSAF Lombardy, the regional agency for agriculture and forest services. According to the website, the loss of nutrient can decrease by 50 % (ryegrass) to 75 % (Brassicaceae) thanks to the use of catch crops (ERSAF, 2013). In northern Italy, legumes are often used in rotation with grasses on grassland (Italian Ministry of Environment, Land and Sea; Italian Ministry for agriculture, Food and Forestry Policies; Regions of Piedmont, Lombardy, Veneto, Emilia-Romagna and Friulia Venezia Giulia, 2010). However, legumes also fix nitrogen from the atmosphere, increasing the amount of nitrogen introduced into soils if they are incorporated. Moreover, the use of a catch crop is sometimes preferred to the use of legumes since catch crops grow more easily during winter when they have less risk to “compete” with other crops in the region.

\textsuperscript{150} “Codice di buona pratica Agricola”, this code is developed in application of the Nitrates Directive.
Improving irrigation systems

Irrigation is widely used in Lombardy: 582,000 ha were irrigated in 2010, corresponding to 58% of the arable while the share of the irrigable surfaces of the UAA is 69% (INEA, Region of Lombardy, Milan University, DEMM, SMEA, 2013). This is much higher than the national average (19% of irrigated area with 64% or irrigable surfaces of UAA). In addition, Lombardy’s dense hydrologic network increases the risk of losses due to the proximity of the water bodies and drainage network, especially where water tables are high, as is the case in some areas of the region.

In Lombardy, only 3.5% of the 31,000 irrigable farms use irrigation advice services in 2010 (6% of the irrigable area). In the region, most of the farmers use surface flow and lateral infiltration (58% of the irrigated surfaces compared to 31% nationally). Sprinkling accounts for 26% of the systems used while submersion accounts for 15% for the rice production. Drip irrigation only represents 1% of the systems used (INEA, Region of Lombardy, Milan University, DEMM, SMEA, 2013). Hence, this measure has a high potential of development in the region.

Drip irrigation delivers water directly onto the soil surface or onto the root zone. It is the most efficient technique to prevent the excess of water. The water use efficiency is 70% to 95% higher than open systems (Hagin & Lowengart, 1996). Drip irrigation also allows saving the use of fertilisers through the fertigation technique. It also decreases compaction and issue of erosion by water that can cause a significant phosphorus loss.

The implementation of this type of system may face technical and acceptance barriers from farmers. Indeed, the use of drip irrigation is not common for cereals, although it is possible, including for maize. Thus, the implement of such techniques in Lombardy requires additional advisory and support programmes to provide assistance and support to farmers. Moreover, the success of this measure also depends on how the farmers are willing to change their crop management. Indeed, the implementation of this measure requires the placement of drip lines that may limit tillage operation. Moreover, some herbicides and top dressed fertiliser require sparkling and thus cannot use the drip irrigation system.

The change of irrigation system can be financially supported at national level. The modernisation of the irrigation infrastructures for water saving and rationalisation is part of the Axis 1 of the Rural Development Programme 2014-2020.
5.4 Murcia (ES)

Box 7 – Murcia case study - In brief

Murcia faces a number of environmental pressures stemming from different sources. The Albujón River and Segura River Basins, for example, have nitrate concentrations exceeding the value of 50 mg/L, mainly due to agriculture-related activities. Phosphorus overloads also result from agricultural run-off and urban wastewater and affect the composition of freshwater bodies, such as the Santomera reservoir, the Albujón River, the Segura River and the Mar Menor lagoon. This leads to fresh and marine water eutrophication in the rivers, the lagoon and the sea. In particular, nutrient overload in the Mar Menor lagoon has dramatically increased the jellyfish population during summer, thereby directly affecting biodiversity and tourism in the region. In addition, the local fishing industry has been negatively affected due to impacts on the feeding grounds of several commercial fish species. Murcia is also affected by high ammonia emissions, mostly due to agriculture.

In Murcia, intensive agricultural systems run the risk of nutrient losses to rivers and lagoons in the region. The causes of observed impacts are related to the excessive application of organic and chemical fertilisers, especially in horticulture production. Another major cause of the nutrient surplus and losses is the high and concentrated production of livestock, especially pigs, leading to a high production of slurry that is not efficiently reintegrated back into the regional crop production. While farmers have improved their practices to face water scarcity and the low soil organic matter content, some inefficiencies e.g. in the elaboration of a fertilisation plan, the water management or during manure storage have been identified. The Mediterranean climate of the region, inducing high temperature, seasonal heavy rainfall that can lead to nutrient leaching, and the soil texture of the region (clay) have a high impact on the nutrient fate and the agricultural techniques. Another factor contributing to nutrient loss and little soil water storage in Murcia is the low organic matter content in the soils.

The Murcia region has significant economic value in terms of tourism revenue. The Mar Menor in particular is one of the tourist destinations with the most visitors in the south-east of Spain. A specific example of cost related to the nutrient overload in the Mar Menor lagoon is related to the management of increasing opportunistic jellyfish in summer. In 2011, an intensive 6-day project was undertaken to remove jellyfish in the lagoon by boats, which led to a total cost of more than € 500 000. Moreover, in order to keep the jellyfish away from the bathing areas and limit the impact of the blooms on tourism, local authorities have installed nets on 43 km of coast at a total cost of more than € 400 000.

For Murcia, the identified good practices include measures aiming at decreasing the source of pollution by reducing the amount of nutrients excreted by animals as well as improving the application techniques or decreasing the amount of fertiliser used, for instance by processing manure or optimising the fertilisation management plans. Another type of good practice concerns the recycling of the nutrients contained in drainage water and in manure, for instance by promoting the use of organic fertiliser and easing its transport and sale through its processing. A third type of good practice relates to the decrease of the nutrient losses during storage and application through measures such as drip irrigation, the implementation of intercropping in orchards and the use of constructed wetlands. Lastly, practices aiming at increasing the soil organic matter content such as the use of manure within the limits set by the EU legislation instead of chemical fertilisers, the incorporation of soil residues, or soil coverage (in particular with nitrogen fixing crops) would help at improving soil structure, improving water and nutrient capture by soil and decreasing the amount of fertiliser needed.

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Murcia is a region of south-eastern Spain located on the Mediterranean coast (Figure 67). Besides high nutrient surplus, this region stands out due to its locally variable climate and its agriculture, mostly relying on intensive horticulture. The region of Murcia has a hot semi-arid Mediterranean climate.

The hydrographic network of the region is made up of one main river, the Segura River, and its tributaries (Figure 68). The Segura River is drained by several ephemeral watercourses flowing into the Mediterranean Sea after episodic heavy rainfall events, usually in autumn (Velasco, et al., 2005). The Albujeón temporary river (Skoulakis, et al., 2005), fed by numerous tributaries draining the agricultural area of Cartagena, flows into the Mar Menor lagoon. The Mar Menor lagoon is another important hydrographic feature since it is the largest natural lagoon in Spain. The Mar Menor is a saltwater lagoon, adjacent to the Mediterranean Sea. Its special ecological and natural characteristics make the Mar Menor a unique ecosystem and the largest saltwater lake in Europe (Velasco, et al., 2005).

The Mar Menor area, the irrigated farmland in the middle of the Segura valley and the Guadalentín Valley were designated as Nitrate Vulnerable Zones in 2001, 2003 and 2009 (Region of Murcia, 2014d) (Figure 68).

Source: (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), Hydrographic Confederation of the Segura River, 2014)

Figure 67 – Geographical location of the region of Murcia

Figure 68 – Map of Murcia showing the designated nitrate vulnerable zones (brown), the river networks (blue lines), the coastal area (dark blue) and the dam of Santomera (red triangle)
In 2013, 564,000 ha were dedicated to agriculture (Eurostat, 2014a), representing 50% of the total regional area. The region is a major producer of fruits, vegetables, and flowers for the rest of Spain and Europe. Fruits and vegetable production accounted for 12.6 and 13% of the national production in 2014 respectively (Region of Murcia, 2014a; Spanish Ministry of Agriculture, Food and Environment, 2014e) and represented 62% of the arable land. In 2014, 126,000 ha were used for the production of citrus and fruit other than citrus. In particular, almond production represented 51% of the surface dedicated to fruit production (Region of Murcia, 2014b). Also, 54,300 ha were dedicated to vegetable production (Region of Murcia, 2014b). The same year, 47,800 ha of land were allocated to cereals, mainly rye and oats (Region of Murcia, 2014b) (16% of the arable land) (Figure 69). Vegetable production is mainly located in the Campo de Cartagena and the Guadalentín Valley that are NVZ. Cereal production is mainly located in the north-west of the region. Citrus production is mostly located in the Segura Valley while other fruit productions are found in the entire territory. Olives and vineyards are mainly in the Altiplano (Region of Murcia, Council of Agriculture and Water, 2012).

Breeding only represents 10.5% of the holdings (Eurostat, 2013g). In 2014, the main livestock production is by far pigs, with 1.76 million head. The second main livestock is sheep with 0.55 million head (Eurostat, 2015g; Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014c). Breeding activities are mostly concentrated in the Guadalentín Valley (in particular in Lorca), in the Campo de Cartagena (in particular in Fuente Alamo), both designated as NVZs, and in the north-western area (in particular in Caravaca de la Cruz) (Global Atlas of the Murcia region, s.d.).

The agricultural system has moved towards an intensive use of land, natural resources and workforce which implies an important use of pesticides and fertilisers (organic and mineral). Since 1979, when the Tajo-Segura river diversion brought water to the Campo of Cartagena, the agriculture has changed from extensive dry crops to intensively irrigated crops (Velasco, et al., 2005). Water now mainly comes from groundwater (Baudron, 2013).
5.4.1 Notable impacts of nutrient surplus

This section focuses on the impacts related to nutrient excess that are specifically due to agricultural practices in Murcia.

In this region, the main impacts are mostly caused by nitrogen and phosphorus surpluses that affect the fresh (rivers and tributaries) and marine (mostly the Mar Menor lagoon) waters. Permanent and ephemeral watercourses transport significant amounts of nutrients to the Mar Menor. The consequences of such pollution in the region are fresh and marine water eutrophication, marine biodiversity imbalance (algae proliferation) and the excessive growth of opportunistic jellyfish populations, the latter leading to serious impacts on tourism (Perez-Ruzafa, et al., 2002). Potassium excess is not a major environmental issue in this region.

The average nitrogen budget in the region amounted to 51 kg N/ha UAA in 2004 (Leip, 2014) and decreased to 26.8 kg N/ha UAA in 2012 (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014a). The average phosphorous surplus was 6.5 kg P/ha UAA in 2012 (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014b). In soil, the excess of nitrogen and phosphorus is mainly due to agriculture. In water, while the nitrogen losses are mostly due to agriculture (Region of Murcia, 2014d), phosphorus mostly results from housing and industrial activities in some areas such as the territory near the Mar Menor (Garcia-Pintado, et al., 2007). In particular, the higher contents of nitrates in the Mar Menor lagoon coincide with the periods of the maximum agricultural activities while a higher content of ammonium and phosphorus could be found during times of the increased human population in the zone as the result of tourism (Alvarez-Rogel, et al., 2005).

Impacts of nitrogen losses

Nitrogen overload in freshwater

Surface water – In the region of Murcia, there are several watercourse networks. The most important one in terms of flow rate is the Segura River and its tributaries. The irrigated farmland in the middle of the Segura Valley was designated as a Nitrate Vulnerable Zone (Figure 70) for several reasons (European commission, 2002):

- It was identified as an affected body of water with clear signs of pollution;
- Concentration values above 150 mg NO₃⁻/L were occasionally reported;
- The area is intensely irrigated, mostly using gravity irrigation that creates high water flows at long intervals increasing the risk of nutrient transport;
- The agricultural system is intensive which implies an important use of fertiliser and increases the risk of nutrient excess.

In 2013, 57 % of the surface water of the Segura River basin did not achieve good ecological and chemical status stated by the Water Framework Directive (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2013b)

Within the Segura River basin, the Cartagena region is particularly affected by nutrient overload. The basin is characterised by a tight network of drainage channels. An area of 441 km² is drained by this network which corresponds to one third of the total surface of the “Campo de Cartagena” natural region (Velasco, et al., 2005).

During heavy rainfall events in autumn and spring, the Albujón River is fed principally by surface run-off (Velasco, et al., 2005). Thus, the tributaries discharge the nutrient-rich waters into the Albujón River. In normal local climatic conditions, the Albujón River is fed by point and diffuse sources. It is estimated that more than 50-55 % of the nitrate in surface waters is of agricultural origin in the watershed of the
Albujón in 2005 (Calatrava Leyva, et al., 2008; Garcia-Pintado, et al., 2007). In summer season, the Albujón has a dry river bed and nutrients are not transferred by surface water.

The average concentration of nitrate in the temporary Albujón River in 2003 was 72.7 mg NO₃/L (seven measures between September 2002 and October 2003) (Velasco, et al., 2005) and increased to 171.5 mg NO₃/L in 2008 (Pellicer, et al., 2009). This exceeded the Nitrates Directive threshold of 50 mg/L above which waters are considered as polluted.

In Spain, 1 % of the monitoring stations for water quality exceeded 50 mg NO₃/L for the period 2008-2011 and 5 % of the stations exceeded 25 mg NO₃/L (European Commission, 2013b).

**Groundwater**— The latest monitoring campaign (2013) of groundwater showed that 60 % of the tested samples from the water bodies in the Segura river basin have a nitrate concentration higher than 50 mg NO₃/L (Medio Ambiente Aqualogy, 2013) and 65 % of the groundwater did not achieve good ecological and chemical status (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2013b). In particular, 24 % of the water bodies show nitrate concentration higher than 150 mg NO₃/L. At the national scale, 23 % of the stations exceeded 50 mg NO₃/L in 2008-2011, and 42 % of the station exceeded 25 mg NO₃/L (European Commission, 2013b).

In Murcia, the majority of the groundwater that showed a nitrate concentration higher than 50 mg NO₃/L is located in the area in the south-east of Murcia, between the Segura River and the coast. Concentrations of more than 300 mg NO₃/L were observed in the Albujón watersheds while concentration of 134 mg NO₃/L were recorded in the Guadalentín Valley (Medio Ambiente Aqualogy, 2013).

**Figure 70 – Nitrate concentration of water bodies in Murcia in 2013**

Source: (Medio Ambiente Aqualogy, 2013)
**Marine water** – The nutrients transported by the rivers are discharged in the Mediterranean Sea. No study has been found about the impact of the Segura discharges into the sea.

Regarding the outflow of polluted water from the lagoon towards the sea, the Mar Menor lagoon acts like a nutrient reservoir. Studies have shown that the spatial distribution of the nutrient concentration in the lagoon is decreasing from the rivers’ mouths to the Sea (Velasco, et al., 2005).

**Ammonia (NH₃) emissions to air**

In 2012, 380 000 t of ammonia was emitted in Spain. Agriculture accounts for 95 % of these emissions (360 800 t) (Eurostat, 2014e). A recent study (Sanz-Cobena, et al., 2014) modelling the ammonia emissions from N fertilisation in Spain shows that the Murcia region was a hot spot of NH₃ with 3 % of the national NH₃ emissions.

The graph below (Region of Murcia, 2014c) shows that the mains sources of emissions (in pink on the graph) are mostly due to livestock farms. The large diffuse patches of dark blue colour correspond to agricultural fields where fertilisers are used.

![Figure 71 – Total NH₃ emissions in Murcia region](source)

One major consequence of nitrogen emissions to air is the direct deposition or precipitation of nitrogen on soils and water bodies (acid rain) thus increasing the risk of nutrient overload. No study has been found at this date to describe specifically the impacts of ammonia emissions from the region of Murcia.

**Impacts of phosphorus losses**

In Murcia, the increase in phosphorus concentration (>10 mg PO₄³⁻/L) in the watercourses around the lagoon follows a seasonal pattern associated to the increase of human population due to tourism in summer (Alvarez-Rogel, et al., 2005; LAGOONS, 2012). While phosphorus is also an agriculture-related water pollutant, it is estimated that in 2006 70 % of the total phosphate discharged in the Mar Menor lagoon was from urban point sources (Garcia-Pintado, et al., 2007). Phosphorus is found in organic fertilisers and also found in mineral fertilisers commonly used for horticulture, crops, flowers, etc. It is very likely that the phosphorus used for horticulture in the region of Murcia contributes to water pollution.
**Eutrophication of freshwater**

The average phosphate concentration in the temporary Albujón River was 1.74 mg PO$_4^{3-}$/L in 2008 (Pellicer, et al., 2009) (European average: 0.21 mg PO$_4^{3-}$/L in rivers (EEA, 2012c)).

In groundwater, the latest analysis campaign of water quality in the Segura watershed detected phosphate in five of the 58 control points. The highest concentration reached 3.43 mg PO$_4^{3-}$/L (Medio Ambiente Aqualogy, 2013). The Segura River and tributaries are affected by eutrophication due to an excess of nitrogen and phosphorus in freshwater. Phosphorus is commonly the limiting nutrient for eutrophication in freshwater. The photography below shows one of the consequences of eutrophication: massive algae proliferation in the Segura River (Figure 72).

![Image of algae proliferation in Segura River](source: Informacion.es, 2010)

**Figure 72 – Algae proliferation episode at Segura River in the city of Rojales (April 2010)**

The Albujón River water was eutrophic, with chlorophyll concentration of about 28 mg/L (Velasco, et al., 2005) in 2003. The dam of Santomera, located in the NVZ, is another example of freshwater eutrophication. Unlike other reservoirs in the area whose state cannot be attributed to agricultural pollution (the Argos reservoir for instance), it has been shown that the reservoir of Santomera is contaminated by nutrients specifically because of agriculture (European commission, 2002). The Cartagena River basin is also affected by nutrient overload.

During 2008-2011, 10 % of the Spanish surface freshwater was eutrophic or hypertrophic. The situation has been consequently improved compared to the period 2004-2007 during which 37 % of the surface freshwater was affected by eutrophication (eutrophic and hypertrophic water status) (European Commission, 2013b).

**Eutrophication of marine water and algae proliferation: the case of the Mar Menor lagoon**

In 2003, 2 010 t per year of dissolved inorganic nitrogen (93 % as NO$_3$) and 178 t per year of soluble reactive phosphorus were discharged in the Mar Menor lagoon from the Albujón watershed and the drainage effluent$^{152}$ (Velasco, et al., 2005). Velasco et al. (2005) showed that the nutrient concentrations in the lagoon decreased with the distance from the Albujón mouth. The study also reported that maximum nutrient levels were found after rain events in late summer or autumn (Velasco, et al., 2005) that increased run-off and nutrients transport.

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$^{152}$ Including two storm periods with high flows
The increase of nutrient concentration in the lagoon induces the development of phytoplankton (Velasco, et al., 2005). Eutrophication is due to both nitrogen and phosphorus load. However, although the lagoon shows high salinity levels, phosphorus is the main limiting nutrient in the Mar Menor lagoon (Velasco, et al., 2005). As for nutrients, the chlorophyll a concentration decreased with the distance from the Albuñón mouth and was higher in late summer and autumn (Velasco, et al., 2005). It is interesting to note that the chlorophyll concentration is partly regulated by the jellyfish population that feed on phytoplankton (Quispe Becerra, 2014).

The eutrophication phenomenon causes a reduction in the quantity of light that reaches the bottom of the lagoon and affects the survival of certain plant species living on the bottom of the lagoon. It allows invasive algae (*Caulerpa prolifera*) to develop and uptake much of the resources. In the 1970s, lagoon primary production was dominated by the seagrass species *Cymodocea nodosa*. However, since the 1980’s the macroalgae *Caulerpa prolifera* has begun to cover most of the lagoon beds, restricting the seaweed *Cymodocea nodosa* to small patches because of the intensity of the competition for resources and especially light (Lloret, et al., 2012; Terrados & Ros, 1991).

*Figure 73 – Evolution of the coverage of the bottom by the main algae in the Mar Menor lagoon*

**Jellyfish population increase**

The eutrophication of the water of the Mar Menor has led to a dramatic increase of jellyfish population (Region of Murcia, 2014d). In the Mar Menor lagoon, over 47 million jellyfish are reported each summer (LAGOONS, 2012). Indeed, Jellyfish eat large algae, protozoans and small crustaceans. By eating the algae, they have a direct effect on the nutrient load as algae uptake inorganic nutrients from water column. However, the simultaneous removal of grazers such as protozoans and small crustaceans reduces the predation pressure on smaller phytoplankton allowing them to flourish leading to phytoplankton proliferation (Perez-Ruzafa, et al., 2002).
The two main species of jellyfish found in the lagoon are not severely dangerous to humans. One of them is completely inoffensive and the other one’s sting is itchy and irritating. During periods of massive proliferation, the number of jellyfish in the shallow waters of the lagoon is very unpleasant for bathers. This negatively affects the number of visitors (Perez-Ruzafa, et al., 2002). Moreover, when jellyfish are beached because of onshore wind, they create visual and olfactory nuisances which may also negatively impact visitors and local population. Action plans to reduce the jellyfish population are being implemented.

5.4.2 Causes of nutrient losses

**Farming system and agricultural practices**

*Causes related to farming practices*

Excessive inputs of nitrogen and phosphorus fertilisers, in particular for horticulture production – Data on the nitrogen and phosphorus surplus of the different crop productions in Murcia shows that fodder crops presented the highest average surplus with 320 kg N/ha and 176 kg P/ha in 2012, followed by vegetables with a surplus of 145 kg N/ha and 92 kg P/ha. The absolute nutrient surplus at regional level is mostly due to vegetable production, while fodder crops only represent a small share of the regional surface area. Other crops (except sunflowers and olive trees) show a positive N budget (see Table 37). The nitrogen budget exceeded 70 kg N/ha for vegetables, flowers, fruit and tubers production and industrial crops in 2012 (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014a). The P budget was highly positive for fodder and vegetables. However, it was low or even negative for other crops. In particular, the P budget was negative for all perennial crops except fruit trees and for flower and sunflower production and industrial crops (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014b).

<p>| Table 37 – Surface area, P and N budget for different arable crops and ligneous in Murcia region for the year 2012 |
| Source: (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014a &amp; 2014b) |</p>
<table>
<thead>
<tr>
<th>Cereals</th>
<th>Legumes</th>
<th>Tubers</th>
<th>Industrial crops</th>
<th>Sunflower s</th>
<th>Fodder crops</th>
<th>Vegetable s</th>
<th>Flowers</th>
<th>Total arable crops</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total surface (ha)</td>
<td>49 199</td>
<td>2 114</td>
<td>2 368</td>
<td>1 015</td>
<td>132</td>
<td>1 221</td>
<td>50 350</td>
<td>344</td>
</tr>
<tr>
<td>P budget (kg/ha)</td>
<td>13.0</td>
<td>18.0</td>
<td>12.0</td>
<td>-2.2</td>
<td>-5.2</td>
<td>175.6</td>
<td>91.6</td>
<td>-163.7</td>
</tr>
<tr>
<td>N budget (kg/ha)</td>
<td>15.0</td>
<td>39.8</td>
<td>105.1</td>
<td>74.6</td>
<td>-11.3</td>
<td>320.4</td>
<td>144.9</td>
<td>92.9</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Citrus</th>
<th>Fruit trees</th>
<th>Almond trees</th>
<th>Olive trees</th>
<th>Vineyards</th>
<th>Others</th>
<th>Total ligneous</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total surface (ha)</td>
<td>39 250</td>
<td>27 243</td>
<td>71 499</td>
<td>32 202</td>
<td>32 202</td>
<td>1 181</td>
</tr>
<tr>
<td>P budget (kg/ha)</td>
<td>-13.9</td>
<td>10.5</td>
<td>-0.9</td>
<td>-0.7</td>
<td>-6.2</td>
<td>-17.7</td>
</tr>
<tr>
<td>N budget (kg/ha)</td>
<td>44.5</td>
<td>104.2</td>
<td>4.1</td>
<td>-14.1</td>
<td>14.0</td>
<td>9.8</td>
</tr>
</tbody>
</table>

The reason for this surplus is the excessive inputs of fertilisers, in particular nitrogen fertilisers. The flows of nitrogen were highest in spring and in summer, the seasons of highest fertilisation (Velasco, et
Fertiliser use in horticulture is particularly high because of the intensive nature of this sector and the high crop rotation intensity (Bomans, et al., 2005).

Farmers that participated in the regional conference\(^{153}\) indicated that the overall lack of water forces them to pay great attention to the amount of water applied which indirectly restrains the quantity of fertilisers that can be applied and assimilated by plants. However, they also highlighted the lack of updated information on the nutrient crop needs. By default, the farmers use data on the maximum amount of nutrients that can be applied indicated in the Nitrate Action Programme or the Code of Good Agricultural Practices. As a consequence, they consider that they often apply more fertilisers than is required for crops.

**High amount of manure produced, in particular slurry, that is not efficiently used for regional crop fertilisation** – The livestock density in Murcia is the second highest after Catalonia, the density is high with 1.4 units/ha UAA in 2010 (Eurostat, 2013a) (national average: 0.62 LSU/ha UAA; average EU-28: 0.77 LSU/ha UAA in 2010) (Eurostat, 2012c). The Guadalentín Valley is an intensive agriculture and farming zone, particularly for pig production. Lorca is the largest municipality regarding the pig production in Murcia with 4 million m³ of pig slurry per year (Llona, et al., 2004). In 2003, approximately 9 million m³ of slurry were produced in the Murcia region (Martinez-Pagan, et al., 2010). Since the number of pigs in Murcia decreased by 13% between 2003 and 2014 (Eurostat, 2015g), it can be considered that the amount of slurry decreased by the same amount.

According to Gomez et al. (2010), manure management from intensive livestock production in the Murcia region is recognised as an environmental hazard (Gomez, et al., 2010). In addition, the useable agricultural area has decreased during the last years due to urban development and abandoned fields while the mean size of livestock farms has increased (Espejo Marin & Garcia Marin, 2009). Hence, the surface area where the slurry produced can be spread is sometimes locally limited (Dauden & Quilez, 2003; Espejo Marin & Garcia Marin, 2009). Moreover, feedbacks from the regional conference highlighted that vegetable producers prefer not to use pig slurry to avoid any sanitary issues linked to pathogenic microorganisms. This contributes to the limitation of the surface where slurry can be applied. Lastly, the lack of communication between the breeders and the crop producers has been identified by the participants of the conference as another barrier to the use of manure by the crop producers. They indicate that crop producers have difficulties to find manure they can apply while breeders have difficulties to deal with their excess of manure.

**Inappropriate manure storage** – In 2009 in Murcia, 61% of the holdings stored the slurry in ponds or lagoons while 51% used slurry tanks\(^{154}\). In addition, 58% of liquid manure, 85% of the slurry and 50% of the solid manure were stored without being covered (INE, 2009). Storing slurry in ponds represents a risk for the environment since nutrients are leached when there is no barrier to the soil layers (Olivares, Faz, & Ramos, 2010). The building of impermeable ponds is mandatory (Olivares, Faz, & Ramos, 2010). Thus, farmers must build ponds with plastic lining or build the ponds in rocks considered to be impermeable. There is no published information about the number of farms which have these types of pond (Muñoz Garcia, 2014). Feedback from the regional conference indicates that many farmers mechanically compact the ponds’ walls, which partly reduces leaching risk but may not entirely eliminate it.

\(^{153}\) As part of the dissemination part of the project regional conferences were held in selected regions. One was organised in Murcia, Spain in November 2014.

\(^{154}\) Both ponds and slurry tanks can be used in a holding
In Murcia, slurry is stored openly on purpose in order to quickly dry the slurry and then obtain a stable material that can be sold. However, the absence of coverage leads to high ammonia emissions.

**Water management system that enhances or limits nutrient transfer** – The irrigation and the drainage system contribute to the transport of nutrients towards the Mar Menor lagoon and the Mediterranean Sea. Murcia is the region in Spain with the highest share of irrigated land with 144 000 ha UAA (INE, 2009). In particular, horticulture production requires a high amount of water.

*Wide use of drip irrigation that limits nutrient losses by leaching and run-off* – In theory, the high quantity of water provided during irrigation increases the risk of leaching due to a rapid water saturation of the soil, carrying labile nutrients and sediments. However, in Murcia the lack of water forces farmers to save water and to use precise irrigation systems. Indeed, the region of Murcia is mostly irrigated by drip irrigation (Lidon, 2014; Murcia Regional Statistics Centre, 2014a; Development institute of the region of Murcia, 2015). The low amount of water provided considerably reduces the risk of nutrient losses by leaching in Murcia compared to other European regions that do not face water scarcity (also see the causes related to water below).

*Water with high nutrient contents that may increase the nutrient surplus* – Until 2011, the irrigation water was mainly extracted from groundwater (Murcia Regional Statistics Centre, 2014a). In some areas, groundwater contains a high concentration of nutrient (nitrogen) from various origins. The use of water that is heavily loaded with nutrients, especially nitrogen, increases the amount of nutrients applied on the field. It may induce an excess of nutrients in the field if the amount of nutrients provided by water is not taken into account when farmers calculate the amount of fertiliser, which is the case in the region.

*Extended drainage system* – The extended drainage system in the Albujón area aims to collect and evacuate the residual rain and irrigation water from agriculture during the precipitation events. Thus, the water collected from agricultural plots in the region of Murcia contains a high amount of nutrients and is quickly transported towards the lagoon and the Mediterranean Sea.

**Environmental conditions**

Environmental conditions in the region of Murcia tend to enhance the impact of over-fertilisation and nutrient transport.

*Impact of the climate on nutrient uptake by plants* – The region of Murcia has a hot, semi-arid, Mediterranean climate. June, July and August in south-eastern Spain are characterised by a very low precipitation (Figure 74) and very high temperature (average of maximum temperatures in August between 1971 and 2000 exceeds 29°C) (AEMET, 2010). According to Nahar & Gretzmacher (2002), plant nitrogen uptake is significantly reduced by water stress. In summer, due to the high rate of evapotranspiration, plants might experience water stress despite the use of irrigation. While nutrient application in summer is often limited, it still may enhance the accumulation of nutrients in the soil that are not assimilated by crops. Under such assumptions, nutrient leaching and transport towards water bodies is amplified in case of heavy rains in fall and winter.

*Impact of air temperature on NH$_3$ emissions* – High temperatures are encountered in summer in this region and it has been shown that high temperatures cause higher emissions of ammonia (Skjøth & Geels, 2013). This study also showed that mineral and organic fertilisers are specifically sensitive to temperature regarding NH$_3$ emissions.

*Impact of the watershed on the nutrient losses and concentration* – The general high depth of aquifers averts the pollution of groundwater as a result of leaching. Regarding surface water, temporary rivers such as the Albujón River that successively show wet and dry conditions, which has a notable
effect on the nutrient flow. During the drying period, nutrient concentration in the river increases due to evaporation losses until the river dries up completely. While the level of water decreases, the water temperature increases, stimulating bacteria activity and thus mineralisation and nitrification, increasing ammonia and nitrate concentration. When the river is dried, there is very low risk of nutrient loss by leaching or run-off.

**Impact of the precipitation on nutrient run-off and soil erosion** – Low precipitation in Murcia (semiarid climate) limits nitrates leaching to the groundwater, rivers or lakes. Conversely, heavy rainfalls in autumn create flash floods which carry high loads of sediments and nutrients to the lagoon (Skoulikidis, et al., 2005; Velasco, et al., 2005). Soil erosion, that is higher when the soil is dry, contributes to the carrying of phosphorus in soil sediments towards the water bodies. Also, maximum nutrient levels in the Mar Menor were found after rain events, principally after the intense winter flash flood (Velasco, et al., 2005).

Impact of soil characteristics on nutrient run-off and leaching – In the region of Murcia the soil is mostly composed of loam and clay (LAGOONS, 2013). Horizons of loams in the soil accumulate nitrates avoiding groundwater pollution. On the other hand, fast wetting of a dry clay soil by high intensity rain enhance aggregate slaking and seal formation (Lado, et al., 1992). Thus, water tends to run-off, transporting soil particles linked to nutrients as well as soluble nutrients. Warrington, et al. (2009) showed that clay soil with important aggregate slaking presented a higher positive correlation between run-off rate and sediment load than clay soils with low aggregate slaking.

Despite the high quantity of CaCO$_3$ in Murcia’s soil (Cammeraat & Imeson, 1998), the low Soil Organic Matter (SOM) content and the salinity issues (Lidon, 2014) contribute to a weak aggregation of soil particles. The degree of soil aggregation and aggregate strength can be regarded as an indicator of the infiltration and water-holding capacity of the soil as well as of erodibility. Here, the soil structure increases the risk of nutrient losses by leaching and erosion. In addition, it induces a lower soil biological activity and reduces nutrient assimilation. Soil degradation is perceived as a major threat in the Mediterranean region due to changes in land-use and possible future climate change (Zdruli, 2014).

**The effect of water temperature and wind on phytoplankton proliferation** – In late summer and early autumn a proliferation of phytoplankton occurs as a result of increased inputs of NH$_4$ and the combination of high water temperature and strong winds. These conditions accelerate phosphorus exchange across the sediment-water interface, increasing the availability of water P for phytoplankton thus enhancing their development (Terrados & Ros, 1991).
5.4.3 Costs of the environmental and health effects

5.4.3.1 Socio-economic description of the region

Murcia represented 3.1% of the Spanish population in 2014 (INE, 2014a). With a GDP of €27.1 million in 2014, Murcia contributed to 2.6% of the national income (INE, 2014b). The nutrient surplus has resulted in significant environmental damages in Murcia, including the impacts on the Mar Menor, which is the largest lagoon on the Spanish Mediterranean coast (135 km²) with a high ecologic value. Indeed, the Mar Menor is a specially protected area under the EU Wild Birds Directive and presents a special importance for endemic fish species such as *Aphanius iberus*, which are threatened by the environmental changes (Velasco, et al., 2005).

The Murcia region in general and the Mar Menor lagoon in particular have significant economic value in terms of fishing and tourism revenues (Velasco, et al., 2005). The Mar Menor is one of the tourist destinations with the most visitors in the south-east of Spain, receiving an average of 200 000 tourists per year. Therefore, the nutrient surplus in the Murcia region and the eutrophication in the Mar Menor lagoon have not only an environmental impact, but also impacts on the local economic sectors such as tourism. Figure 75 illustrates the potential costs of polluted Mar Menor lagoon on the relevant economic sectors in the region.

![Diagram](image.png)

Source: (Martínez, et al., 2007)

**Figure 75 – Cross-impacts between the relevant socio-economic activities in the Mar Menor area, connected through the ecological state of the lagoon**

Note that some data provided in this section may cover a slightly different area from Murcia.

5.4.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in Murcia, a wide range of literature was reviewed. Costs were identified and are presented below, based on the classification of economic damages caused by environmental impacts presented in Annex 14.
It is important to note that all cost data reported in this section are taken from the primary studies found in the literature and expressed in the value of the year when the study was conducted.

**Clean up and restoration costs (CRC)**

**Jellyfish removal**

Nutrient overload in the Mar Menor lagoon has increased the population of opportunistic jellyfish in summer, and directly affected biodiversity and tourism in the region. To diminish the number of jellyfish near the beaches removal operations have been carried out. In 2011, an intensive 6-day project was undertaken to remove jellyfish in the lagoon by boats, which led to a total cost of € 532 000 (La Razon.es, 2011). In 2012, an average of 50 tons of jellyfish were collected each day from the lagoon (RTVE.es, 2013). Moreover, in order to keep the jellyfish away from the bathing areas and limit the impact of the blooms on tourism, local authorities have installed nets on 43 km of coast at a total cost of € 404 000 (Gomez, 2011).

Regarding the Segura River, the most important river in the region of Murcia, a clean-up and restoration project was launched to clean the Segura river at a total expense of € 12 000 000. The project includes the cleaning of the industrial discharges as well as the agricultural pollution between 1999 and 2003 (European parliament, 2004).

**Nutrient removal from water**

In addition, a study by Perni & Martínez-Paz (2013) estimated the costs of different measures to deal with pollutants due to agricultural practices in the Murcia region, including wastewater treatment and restoration projects. The costs of the measures were estimated based on the information available in the *Esquema Provisional de Temas Importantes* (Provisional Outline of Important Themes) reported by the CHS (CHS, 2008), (Grindlay, et al., 2011). In this study, Perni & Martínez-Paz suggested that amongst all the possible solutions, the measure with the least cost is the restoration of the *Rambla del Albujón* (the river bank) (Perni & Martínez-Paz, 2013). The cost is estimated over € 181 000 per year to remove the waste materials from the brook and reforest autochthonous species to favour the retention of nitrates and phosphates. In comparison, the most expensive measure is related to wastewater treatment, which is estimated at an annual cost of € 7 814 210. Detailed data on the cost of removing nitrates and phosphates were not identified. Other measures, such as expanding the Mojón desalinaisation plant and drain network to treat irrigation returns from the Campo de Cartagena would cost € 6 385 130 per year, whereas setting up perimeter wells to extract polluted groundwater to be treated in the treatment plant and used for irrigation purposes in the Campo de Cartagena would cost over € 2 318 200 per year. The latter feature may also take into account urban polluted water as well as pollution due to agriculture (Perni & Martinez-Paz, 2013).
Use value damages (UVD)

The project ECOHARM estimated that the socio-economic impact of harmful algal bloom in Spain was around € 178 million per year, including the impact on tourism (€ 124.3 million) and on commercial fisheries (€ 52.8 million) (EEA, 2005c). No studies have been found that estimate the total costs of the use value damages specifically for the region of Murcia.

Furthermore, as already mentioned, the nuisances caused by jellyfish (stings, visual and olfactory pollution) impacts the number of visitors in the area (Almeria verde, 2007) thus affecting the local economy. A study by Martinez-Paz et al. in 2013 aimed to evaluate the willingness to pay (WTP) for direct improvement of enjoyment of the Mar Menor (amenity value). Through a series of interviews, the study showed that the annual WTP per capita for improving the use value of the Mar Menor beach (no temporary closing of beaches, decrease of jellyfish population, improvement of the tourist image) is around € 13.65 (Martínez-Paz, et al., 2013).

In addition to tourism, the local fishing industry is negatively affected by the proliferation of the algae Caulerpa prolifera at the expense of less vigorous seagrass species such as Cymodocea nodosa. Indeed commercial fish species, mainly Sparidae and Mugilidae, prefer feeding on patches of the seagrass C. nodosa or unvegetated bottoms, which are now covered by a dense and continuous bed of the macroalgae C. prolifera (Verdiell-Cubedo, et al., 2007).

Table 39 presents the use value damages that have been found for Spain.

Passive use value damages (PUVD)

No studies estimating the cost of passive use value damages (such as the endangerment of marine species and other environmental externalities) in the region of Murcia were found. The project ECOHARM estimated that the total expenses for medical treatment, transportation and lost wages due

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155 Removal of waste materials from the brook. Reforestation with autochthonous species to favour the retention of nitrates and phosphates.
to illness related to harmful algal blooms cost are € 10 000 per year (EEA, 2005c) (Table 40).

The lagoon has special environmental value because of its characteristics (salinity, temperature, etc.) and the diversity of wetlands in the surroundings. Many allochthonous species (fauna and flora) are found in the area. Some plant species have adapted to the high salinity (Arthrocnemum macrostachyum, Sarcocornia fruticosa or Pancratium maritimum) for instance. There are also some species of high fishing interest such as Anguilla anguilla, Mugil cephalus and Sparus aurata. In addition, over 200 bird species have been located in the Mar Menor and its surroundings. The Mar Menor lagoon area is protected to preserve the current diversity of habitats and species (Martínez-Paz, et al., 2013). Nevertheless, nutrient pollution in the lagoon occurs and may affect the biodiversity and enhance changes in the natural habitats. Such damages are important to bear in mind while evaluating the impacts of nutrient excess on the environment. The study by Martinez-Paz et al. in 2013 showed that the annual WTP per capita for good environmental improvement in the Mar Menor (reaching a good ecological status on the long term) is € 24.27.

Table 40 – Passive use value damages found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Damage estimation</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Public health</td>
<td>Medical treatment, transportation and lost wages due to illness</td>
<td>€ 10 000</td>
<td>Spain</td>
<td>Per year</td>
</tr>
</tbody>
</table>

Policy action costs (PAC)

The Mar Menor area and the irrigated farmland in the middle of the Segura Valley are designated as Nitrate Vulnerable Zones (NVZ). The region therefore has to comply with several requirements set by the Nitrates Directive, including drafting an action programme to combat the nutrient problem.

Different programmes funded by the Consejería de Agricultura y Agua and developed by the Research Group Sustainable Use, Management and Reclamation of Soil and Water from the Universidad Politecnica de Cartagena and the agricultural cooperative FECOAM were carried out from 2001 to 2011 with the aim to demonstrate the sustainable use of the raw pig slurry based on its agricultural valorisation.

As part of the Water Framework Directive requirements, River Basin Management Plans (RBMP) must be drawn up for all river basins in order to reach the level of good water quality. For the Segura river basin (in which the study region is situated), the RBMP 2015-2021 is still pending and has therefore not yet led to the development of new measures.

The project ECOHARM estimated that the monitoring and management costs of harmful algal blooms in Spain were around € 0.9 million per year (EEA, 2005c) (Table 41). At the time of the study, no studies have been found to assess the costs of such policy actions in the region of Murcia.

Table 41 – Policy action costs found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Monitoring plans</td>
<td>Monitoring and management of harmful algal bloom</td>
<td>920 000</td>
<td>Spain</td>
<td>Per year</td>
</tr>
</tbody>
</table>
5.4.4 Good practices to reduce nutrient losses at farm level

For Murcia, the identified good practices include measures aiming at decreasing the source of pollution by reducing the amount of nutrients excreted by animals and improving the application technique or decreasing the amount of fertiliser used, for instance by processing manure or optimising the fertilisation management plans. Another type of good practice concerns the recycling of the nutrients contained in manure, for instance by promoting the use of organic fertiliser and easing its transport and sale through its processing, as well as the nutrients contained in drainage water. A third type of good practice relates to the decrease of the nutrient losses during storage and application through measures such as drip irrigation, the implementation of intercropping in orchards and the use of constructed wetlands. Lastly, practices aiming at increasing the SOM content such as the use of manure instead of chemical fertilisers, the incorporation of soil residues, or soil coverage (in particular with nitrogen fixing crops) would help at improving soil structure, improving water and nutrient capture by soil and decreasing the amount of fertiliser needed.

5.4.4.1 What has already been done in the region

Following the latest Code of Good Agricultural Practices developed for the Murcia region and published in 2003, Murcia has developed Nitrates Action Programmes at the local scale within the region. For instance, the order of the 3 March 2009 focused on the action programme for the aquifers of the regions of the Segura rivers (Region of Murcia, Council of Agriculture and Water, 2009). This programme and the related requirements take into account the local context including the farming systems, practices and the soil and climate conditions. It also provides the information necessary to elaborate the fertilisation plan at farm level. The River Basin Management Plan (RBMPs) 2009-2015 under the WFD is an important tool to improve the region’s water and environmental quality. Aside from waterlogging caused by pig slurry which is forbidden, no specific requirement is mentioned in the RBMP that refers to the Nitrates Action Program established for NVZs (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014d). Also note that no public funding is dedicated to tackle the diffuse pollution (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014d).

Due to the water scarcity that the region faces, Murcian farmers have implemented many practices that aim at saving water. Hence, a high precision in the water use for crop production has been developed. For instance, 72 % of the irrigating farmers already use drip irrigation in 2012 (Murcia Regional Statistics Centre, 2014b), limiting the risk of nutrient leaching and run-off due to the anthropic provision of a high amount of water. Regarding fertilisation planning, some tools such as the one developed by the Murcian Research and Development Institute for Agriculture and Food (IMIDA) have been developed to support farmers.

As regards breeding, the Government of Spain established basic standards of management of pig farms (RD 324/2000) and regulatory rules for awarding grants to projects that seek to improve environmental management of pig farms (RD 987/2008). Many farmers have already implemented specific techniques for improving manure collection. In 2009, on the 1 096 holdings that breed pigs, nearly 80 % already used partly slatted floor, which represents 74 % of the pig stock. Fully slatted floor is used in 14.5 % of the holdings (INE, 2009). Progress is still possible for manure storage and use. Several projects aiming at promoting the use of pig slurry as a fertiliser has been developed by several regional institutions such as the Consejería de Agricultura y Agua, the Research Group Sustainable Use, the Universidad Politécnica de Cartagena (Muñoz Garcia, 2014). Many initiatives have been developed to test and improve manure processing technologies. In 2008, the Spanish Government approved a plan of biodigestion of slurry. However, the Spanish government recently ended the subsidies provided for anaerobic digestion due to the tight budgetary context (Muñoz Garcia, 2014; Spanish Ministry of Agriculture, Food and Environment, 2015).
5.4.4.2 **Good practices to reduce the nutrient losses in livestock production**

*Use feeding practices to reduce the amount of nutrient produced*

Feeding practices allow decreasing the amount of nutrient excreted by (1) adjusting the feed (and hence the nutrients) to the animal’s needs, through multiphase feeding for instance, and (2) increasing the assimilation of nutrients by animals. In particular, increasing the assimilation of feed by pigs is a very effective practice. (Kaasik, 2012; Sutton & Beede, 2003).

This measure could have positive effects on ammonia emissions in Murcia considering the intensive pig production in the region and the high amount of slurry produced. In 2010, the Spanish Ministry of the Environment and Rural and Marine Affairs (MARM) published a national Best Available Techniques (BAT) guide for the pig sector that includes feeding practices. However, the guide is not very specific. While it recommends adjusting the amount of nutrients provided according to the growth and life stage, the BAT only proposes one single diet per animal type (sows, fattening stage and piglet) with a range of protein content in feed.

Farmers that participated in the regional conference stated that they are aware of the feeding practices to reduce nutrient excretion. However, no report on the level of use of these techniques is available for Murcia.

*Process manure*

In Murcia, breeders such as pig producers do not have sufficient land area to apply the manure or to transfer it to nearby crop producers. When the nearby producers grow horticultural crops, manure needs to be transported further because the crop producers often do not want to apply slurry to avoid pathogenic contamination. Hence, processing slurry and more broadly manure would allow obtaining a concentrated and stable fertiliser that could be more easily transported and sold. Some processing techniques can also reduce the amount of active pathogens in the transformed materials, possibly encouraging its use in horticultural production. This could be helpful to reduce the risk of contamination of soil and water by nutrients in some areas (for instance if nutrients (N and P) are already in surplus or already provided by another source). Technologies for removing the nutrients from slurry are less relevant in this region since crop producers lack organic fertilisers. Some farmers have installed treatment plants but this technology is not widely spread. Solid-liquid separation with an aerobic-anerobic treatment is the most common processing technology used in Murcia. Composting is also used. Other types of techniques are not developed in Murcia and may be tested. Some possible processing techniques that could be further promoted and implemented are mentioned below.

*Separate liquid and solid fractions of slurry*

Among the pig farms that process the slurry in Murcia, most of them have implemented a mechanical separator to produce solid manure and liquid pig slurry (Muñoz García, 2014; Life+, 2011). This measure is considered as a BAT according to the draft of the national guidance. The liquid fraction could be used for fertigation, which is mostly done in farms using this technique in Murcia. The use of the liquid fraction also reduces the risk of pipes clogging, which is an issue frequently observed in farms using this system. The solid fraction could be applied in soil and help increase the SOM content, or processed by composting, which is already done in the region for crop productions other than horticulture. The Veterinarian Faculty of the Murcia University tested an innovative separation technology called SELCO-Ecopurin® technology in an experimental small-scale pig farm (“Granja Guevara”) over five years. The technology allows recovering 90 % of the solid fraction of the slurry by flocculation based on ionic transfer using polymers (Martínez-Almela & Barrera, 2005). In addition to the initial investment (amount unknown), the running costs of this technology includes the energy consumption (8.5 KWh/m³ pig slurry) (GE, 2002) and the purchase of expensive flocculants. The project was not renewed. Today, the farm
still separates the slurry by using a screw press in the context of the on-going European project MANEV\textsuperscript{156} that aims at developing region-specific good practices to improve manure storage (Santos, 2014; Bernal Calderon, 2013).

**Compost solid manure and the solid fraction of the slurry**

Composting the solid manure or the solid fraction of the slurry allows the volume of the initial material to be reduced while obtaining a stable fertiliser that can be more easily stored, applied on the fields or sold. In the case of Murcia, composting is particularly relevant for breeders that can possibly generate revenue from the slurry that is produced and sold. In particular, the obtained material contains few pathogens and thus could be used by horticultural producers. This technique is currently used by a small share of farmers in Murcia. It is tested in research projects such as the projects led at the experimental farm Granja Guevara, and it is tested in demonstration farms such as the initiative conducted by the retailer Biocampo (Moreno Cornejo, 2014). The application of composts would mostly help farmers to increase the SOM content of soil in Murcia, more than as a fertiliser. Considering the C/N ratio of the compost, the addition of slurry or chemical fertiliser may be necessary. Thus, the nutrient content of the compost should be taken into account to avoid over-fertilisation.

**Recover nutrients from manure by precipitation**

This technology could be interesting to extract the ammonia and phosphate from the slurry or liquid manure in a very stable form and a small volume. It requires a separation of the solid-liquid fraction of these materials. Ammonia and phosphate can be precipitated together forming struvite. Phosphorus can also be precipitated as calcium phosphate (Flotats & Magri, 2011). The obtained products can be easily sold. The residue, rich in carbon, can be used to increase the Soil Organic Carbon (SOC) content with a reduced risk of nutrient surplus and possible losses. This is interesting in the context of Murcia considering the low SOM content. Nutrient recovery by precipitation has been tested for phosphorus during the project SELCO-Ecopurin® (Martines-Almela & Barrera, 2005). It is also tested for wastewater in treatment plants (Barat, et al., 2009) and in the on-going European project Efficientheat\textsuperscript{157} involving two partners from Murcia\textsuperscript{158}. It aims to “reduce the pig slurries volume, reduce waste management costs, minimise pollutants emission and optimise total process energy consumption” (European Commission - CORDIS, 2014)

**Use digested manure from biogas production**

In Murcia, the biogas production from pig slurry is considered as a very interesting option to valorise manure, including in horticulture since the pathogens do not survive this process. However, it is not economically viable for the moment. Indeed, numerous technical, economical and administrative barriers should be overcome before the wide implementation of this practice.

In Spain, the objectives of the national Renewable Energy Plan 2011-2020 is to attain a potential of production of 19 408 ktoe in 2020 (Spanish Ministry of industry, tourism and trade and the National Institute for energy diversification and savings (IDEA), 2010). The Region of Murcia has established a law on renewable energy and energy efficiency in 2006. The potential production of biogas for Murcia is estimated to 65.8 ktoe (IDAE, 2012). Currently, only a small share of manure is used to produce

\footnotesize{156} http://www.lifemanev.eu/index.php

\footnotesize{157} www.efficientheat.eu

\footnotesize{158} HRS Heat Exchangers, S.L.U., and Talleres Martinez Lorente, S.L.
biogas in pilot plants. The use of digestates as fertilisers has not yet been proved at the industrial scale in the region.

In 1998, the Spanish government included the treatment of pig slurries through cogeneration into the special electrical regulation\textsuperscript{159} (Espejo, 2005). In 2009, the Spanish government had provided financial aid for the development of the biogas production through the Plan for the Biodigestion of Slurry (Ministerio de Medio Ambiente, y Medio Rural y Marino, 2009) that can provide incentives for farmers to implement this measure. One cogeneration plant was constructed in Alhama de Murcia and another two in Lorca. However, expensive and complex systems such as cogeneration plants which use pig slurry mixed with petrol or gas have been demonstrated to be inefficient and unsustainable as a result of the high cost and the qualified workforce required, in particular considering the recent cuts in the subsidies (Santos, 2014). In addition, the administrative proceedings to obtain the licences are particularly long in Murcia. Recently, some cogeneration plants closed due to the reduction or the removal (depending on the type of plants) of the incentives in 2014 following the economic crisis that began in 2008-2009.

A technical barrier to the implementation of biogas plant in Murcia is the possible need for other substrates than pig manure. Solid manure is usually used for producing biogas. While anaerobic digestion is possible with only slurry, the use of other substrate increases the yield of methane production because it adds more dry matter. Nonetheless, energy crops are not produced in Murcia, in particular considering the high amount of water that would be required to grow these crops and the current low economic viability of such crops. The use of crop residues from cereals for instance is possible. However, it would require removing the residue from the fields and consequently to remove a significant source of nutrients and carbon that should be compensated for the next crop. In addition, the residues would need to be transported to the digestion plant, which can represent a long distance considering the area of production of the manure and the cereals.

However, several projects aiming at challenging the relevance and the feasibility of anaerobic digestion technologies are on-going in the region. Between 2007 and 2011, the research Probiogas project, which united 14 research centres, aimed to optimise the efficiency of the production and use of agro-industrial biogas in environments and demonstrated its feasibility and promotion in Spain (Probiogas, 2014). The CEBAS-CSIC led a project related to the valorisation of the digestate and participated in several other projects, including a demonstration plant project using citrus and breeding residues. The project “Integrated Pilot Plant for complete energy recovery of different municipal and livestock waste materials and by-products (METABIORESOR)\textsuperscript{160} is a project co-financed by the European Union LIFE+ program and the Autonomous Community of Murcia (Spain). It consists in developing a new industrial process to propose alternative ends of life for livestock by-products and waste from the farm to the slaughterhouse gate. The project aims at combining different processes in a pilot plant, such as hydrolysis of various organic by-products, methane fermentation, digestate recovery and biomass combustion for the treatment of different types of waste and by-products. The project Metabioresor will demonstrate the feasibility of this process and provide the technical and economic adjustments to its different phases allowing for the construction of full-scale plants. These plants could be used, for example, by municipalities, cooperatives and large pig farms for the management of waste and by-products. Return on investment would be ensured by the energy produced and environmental impacts would be reduced.

\textsuperscript{159} Decree 2828/1998

\textsuperscript{160} www.metabioresor.eu
Acidify slurry to reduce ammonia emissions in ponds

The acidification of slurry may be an interesting solution, in particular for slurry in ponds that are not covered. The measure consists in the addition of sulphuric acid to urine to decrease its pH. This measure decreases ammonia emissions during storage significantly, as well as during application. In addition, considering the high amount of CaCO$_3$ in the region's soils, the use of the acidified slurry as a fertiliser may not cause significant soil acidification.

Other methods are possible to reduce the ammonia emissions from liquid manure storage. The control of slurry temperature is not adapted to the regions due to the lack of proper storage vessels. In holdings with slurry tanks, cooling slurry may not be economically viable considering the possible high energy consumption to cool slurry considering the region’s climate.

**Ensure waterproofness of storage vessels**

Earth-banked stores or lagoons waterproofed with a geomembrane are considered as Best Agricultural Techniques (BATs) (Joint Research Centre, 2013). Otherwise, the building of a slurry tank is the best technique to avoid any losses through leaching and run-off. In addition, while covering the slurry ponds has high effects on ammonia emissions, the method is contradictory with the current use of manure by farmers in the region and could be difficult to implement. Indeed, farmers aim at drying the slurry to obtain a stable material that can be sold. Therefore farmers tend to increase the surface of the ponds and do not cover them to ease water evaporation. Consequently, it is important to use non hermetic covers in order to allow drying the slurry but also to avoid methane production, unless they were associated to biogas production. To reduce smells, predominant winds should be taken into consideration when locating the ponds.

For sheep breeding, solid manure storage in heaps could be improved by the installation of an impermeable (e.g. concrete) floor to avoid losses by leaching and heap coverage. Considering the significant impact of inappropriate storage manure management on the environment, in particular if manure is stored in the field, it is crucial to raise awareness on the environmental impacts of inadequate storage vessels and to enforce the regional legislation so that all the farms in the region install proper storage facilities. In the case of Murcia, these measures would concern a large share of farms since less than half of the holdings covered their manure. This leaves an important room for improvement in the region.

Note that one of the objectives of the European project MANEV is to monitor and evaluate the manure management strategies in the region of Murcia.

In addition, some recent surveys point out that the main release of gases from the pig slurry also happens when the slurries are moved during transportation (Faz, et al., 2010). In order to tackle this issue, a regional researcher that participated to the stakeholder consultation of this project suggested to the use of underground tubes.

**5.4.4.3 Good practices to reduce the nutrient losses in crop production**

**Prepare and improve the fertilisation plans to balance mineral and organic fertiliser use**

Murcia is the third Spanish region in terms of mineral fertiliser consumption per ha with about 120 kg N/ha, 60 kg P/ha and 40 kg K/ha for the period 2011-2012 (Spanish Ministry of Agriculture, Food and Environment, 2013a). Mineral fertilisers are very prone to leaching and run-off because of their high solubility. While the preparation of a nitrogen fertilisation plan is mandatory in NVZ, the plan could be better adjusted, extended to areas outside NVZ and prepared for phosphorus. In addition, the regional
climate results in high evaporation and irregular rainfall. In order to save water and to avoid nutrient leaching, spreading at the right time is required. The use of methods such as the ATMS could be useful.

In Murcia, soil and manure analyses are regularly performed. They are mandatory according to the regional Nitrate Action Programmes. However, the amount of nutrients in water, the amount of nutrient from the previous crop (especially in the case of legumes) and the amount of nitrogen from atmospheric deposition should also be taken into account, in particular considering the high nutrient concentration in the water for the irrigation. In addition, feedback from the regional conference indicated that information regarding the crops’ requirements should be updated. Farmers currently use data on the maximum amount of nutrients that can be applied as they consider obsolete, often over-estimating the crop’s needs. It is also essential to consider the possible nutrient losses through air emissions, run-off or leaching by establishing an accurate nutrient budget (Gomez-Garrido, et al., 2014). Information specific to Murcia are required considering the soil specificities, such as the CaCO$_3$ or salt contents, that can affect the nitrification process in soils treated with extremely high doses of raw pig slurry.

Horticulture is one of the most important agricultural activities accounting for over 75 % of the total arable land surface (in 2011) (Spanish Ministry of Agriculture, Food and Environment (MAGRAMA), 2014a). The areas dedicated to horticulture are also the areas that present the highest surplus of nitrogen and phosphorus. The surplus exceeds 100 kg N/ha for all types of horticultural crops and 80 kg P/ha for vegetables. Hence the preparation of a fertiliser plan should particularly focus on horticulture production, in particular vegetables for phosphorus. A study in Alicante (south-eastern Spain) showed that the reduction of the quantities of mineral fertilisers applied on horticultural crops reduced the loss of nitrogen but plant yield was not significantly affected (Pedreño, Moral, Gomez, & Mataix, 1996).

This measure is technically easy to implement but may require a high amount of additional information, know-how and changes in the farmers’ habits. Information necessary to elaborate the fertilisation plan at farm level is provided in the Nitrates Action Programme for the aquifers of the regions of the Segura river (Region of Murcia, Council of Agriculture and Water, 2009). Since 1995, the Research and Development Institute for Agriculture and Food in Murcia (IMIDA) has developed the Agricultural Information Systems of Murcia (SIAM) that provide free real-time information on the climate and a decision tool to help farmers to elaborate their irrigation and fertilisation program. While information is available for the entire Spanish territory, most of the users are from the region, with more than 27 000 website visits from the area around Murcia. However, most of the users are not the farmers themselves but rather communication companies and agri-food companies. Further communication is required to encourage farmers to use this system (Ayala, 2014).

**Preferentially use organic fertilisers**

Agricultural utilisation of manure or compost as fertilisers, within the boundaries of the existing environmental legislation, is interesting as it is a source of nutrients and organic matter and it improves the physical properties of soils (Gomez-Garrido, et al., 2014).

However, farmers that participated in the regional conference highlighted the lack of communication between breeders and crop producers. As a result, crop producers face difficulties to find organic fertiliser. In addition, they estimate that the use of organic fertiliser is not economically viable considering its high price and the cost of workforce to apply it. Consequently, they prefer using chemical fertilisers. Several programmes funded by the Consejería de Agricultura y Agua and developed by the Research Group Sustainable Use, Management and Reclamation of Soil and Water from the Universidad Politécnica de Cartagena (Murcia) and the agricultural cooperative FECOAM were carried out from 2001 to 2011 with the aim to demonstrate, promote and ease the sustainable use of the raw pig slurry based on its agricultural valorisation. This is the case, for instance, of the project on the “Valorización en común de purines para fertilización de zonas agrícolas y recuperación de zonas degradadas” and “Valorización...
Agronomía de purines para su gestión en común” (Muñoz García, 2014). Another idea proposed by the farmers that participated in the conference was to create a “manure bank”. The bank would ease the connection between “manure producer” and “manure consumer”. It would collect the excess of manure from breeders in one hand and gives the opportunity to crop producers to buy it on the other hand. This would increase the availability of organic fertilizer and decrease its price while valorising the high amount of manure produced.

**Use appropriate application techniques**

The conventional method for applying pig slurry in Murcia is the splash plate applicator. Surface-banding of pig slurry in an established crop in spring has become a very popular application method in recent years, because this technique is more easily compatible with slurry application by tanker (Gomez, et al., 2010). No more precise figures are available. Such techniques and the use of trailing shoe should be promoted in Murcia, although they entail high investment costs. A collective purchase of machines allows sharing the costs and benefits from the savings from fertilizer purchase.

**Favour incorporation of manure into the soils immediately after application**

This measure is interesting for the region as incorporation reduces the temperature of manure and thus reducing the risk of volatilisation. In addition, it is a relevant solution in areas where the local population is increasingly unhappy with the inconveniences caused by the smell when the pig slurry is applied, mainly in those cities located close to large agricultural zones such as Lorca or Cartagena. It is strongly advisable to incorporate the pig slurry immediately after application in order to avoid releases of NH3 emissions and run-off of nutrients in the case of heavy rainfall. Generally, the greatest reduction in NH3 emission is obtained when slurry is deeply incorporated immediately after application (Webb, et al., 2010). In the region, no specific deadline is required for the incorporation in the NVZ. This technique is not mandatory in other areas (Region of Murcia, 2009; Region of Murcia, 2011; Region of Murcia, Council of Agriculture and Water, 2009).

In Murcia, the share of farmers that incorporate solid manure and slurry was respectively 65 % and 66 % of the in 2009. This represented 76 and 74 % of the regional UAA (INE, 2009). According to Eurostat, 73 % of the Spanish holdings immediately incorporate solid manure or slurry after application (Eurostat, 2013h). The potential of improvement regarding incorporation is still interesting. More effort in terms of education is needed to maximise the reduction of the emissions of NH3. Moreover, a specific and appropriate mandatory timing for incorporation would be needed.

**Do not leave the soil uncovered**

This measure mostly concerns cereal producers and perennial crop producers and orchard and vineyard farmers. Since vegetables are produced in winter, their producers are not concerned by this measure. In Murcia, 41 % of arable land is left naked in winter, increasing the risk of erosion and soil deterioration, which results in nutrient losses in particular phosphorus. This figure has decreased slightly compared to 2009 when the 43 % of the UAA was left bare (INE, 2009). This also enhances nutrient leaching in cases of occasional heavy rainfall. Among the farms that cover their soils in winter, 30 % produced winter crops such as vegetables, 19 % used intermediary crops and 3 % covered with crop residues in 2009 (INE, 2009).

Several practices are possible such as the use of cover crops (e.g. grass, catch crops or legumes) and mulching. These practices allow increasing the SOM content, reducing the risk of soil particle losses and related nutrients and enhance the soil biological activity. While the effect of cover crops on the maintenance of soil particles is direct and immediate, the effect on SOM content and soil structure are long-term. In a context of water scarcity, the implementation of cover crops may not be accepted by farmers. Indeed, in the short term, water competition between the crops may decrease the main crop's
yield. However, studies in Andalucía show that early cover crop implementation and early removal in orchards would minimise possible yield losses (Ramos, et al., 2010; Gomez, et al., 2009b; Gomez, et al., 2009a). Moreover, some farmers using catch crops have even reported not to need to fertilise in the following period. Thus, adequate information on the choice of appropriate cover crops, in particular catch crops and legumes, adapted to dry climate should be provided. Grass, in orchards or vineyards for instance, also consumes water. It also gives a neglected appearance to the parcel and farmers may be afraid that the land would be wrongly considered as abandoned and the CAP subsidies consequently removed. Given the greening of the CAP as stated in Art. 46 of the EC Regulation No 1307/2013 this doubt can be ruled out.

One possible, more easily accepted technique could be to use crop residues. Mulching has the advantage of not consuming water and may also increase the main crop yield by maintaining soil humidity (Unger & Vigil, 1998).

**Incorporate crop residue**

In Murcia, the incorporation of crop residues can have a high effect on the average regional nutrient loss since nearly half of the regional arable land is used to grow cereals. Residues such as straw have high C/N ratio and help immobilising nitrogen that is slowly released, thus improving soil quality. The residue will help to increase the SOM content and improve soil structure and thus soil water retention. This measure is particularly appropriate for the region since crop residue incorporation and management is mainly beneficial for soils which are well aerated or contain high clay contents (Peyraud, et al., 2012). Feedbacks received during the regional conference showed that this practice has been increasingly used over the last 7-8 years, especially in citrus, almond and grape production. However, the utility of this measure is not yet clear for some farmers.

**Prefer conservation tillage**

In Murcia, conventional tillage was used on 88 % of the arable land use in 2010 (Eurostat, 2013d). While tillage increases aeration of the soil in the short-term, over the long-term it destroys the soil structure leading to compaction and anaerobic conditions, in particular in clay soils. Conservation tillage refers to a decreased frequency of tillage and thus less adverse effect to soil stability. The effect of conservation tillage on emissions depends on the type of soil as well as the organic matter supplied (more or less labile). Tillage has a high effect on the decrease of N\textsubscript{2}O emissions in clay soil. In addition, tillage decreases water evaporation by preventing capillary water flow, which is very interesting in Murcia. On the other hand, tillage decreases the SOM content and affects the soil microbiota influencing the humification/mineralisation equilibrium and natural processes. Conservation tillage could be an interesting compromise to still reduce N\textsubscript{2}O emissions while reducing the soil structure degradation and SOM loss. It may reduce the risk of erosion by water and thus phosphorus and other nutrient losses. However, it should not be implemented in areas where the soil is too easily compacted and saturated locally, which is a risk in the region considering the soil texture and the wide use of conventional tillage. In 2009, conservation tillage and no-tillage were implemented in 6 % and less than 1 % of the UAA respectively (INE, 2009).

**Favour drip irrigation systems**

Murcia is the region in Spain with the highest share of irrigated land. The irrigation and the drainage system contribute to the transport of nutrients towards the Mar Menor lagoon and the Mediterranean Sea. Horticulture production requires a high amount of water. The high quantity of water provided increases the risk of leaching due to a rapid water saturation of the soil, carrying labile nutrients and sediments. In Murcia, 16 % of the abstracted water is used through gravitational irrigation and over 83 % of the irrigation was drip irrigation in 2013 (Development institute of the region of Murcia, 2015). While this figure shows a large use of this efficient irrigation technique, this figure continues decreasing since
2005 in favour of gravitation irrigation (Murcia Regional Statistics Centre, 2014b). Further efforts are necessary to convert the remaining gravitation systems into drip irrigation, when possible.

Drip irrigation is one of the most adapted techniques for horticulture or arboriculture in Murcia because of the climate (very hot and dry in summer – semiarid climate). Indeed, this technique limits evaporation and increases water uptake efficiency by distributing water directly on the ground at the bottom of each plant. Also, the quantity of water distributed is precisely controlled.

**Re-use agricultural drainage water**

This measure is adapted to the region of Murcia as it responds to several combined issues such as water scarcity and nutrient pollution of water by agriculture but also industrial and urban wastewater.

In 2002, the University of Madrid hosted a research study developing a biological system to recycle water drainage. The aim is to eliminate the contaminating compounds by indigenous bacteria and to reuse the water in the irrigation also using excess fertilisers used for plant growth (Lopez Garbi, 2002).

In Murcia, a project of water reuse for irrigation was led on the Segura River. In this case, the Segura river water is decontaminated in a treatment plant and the clean water is reused for irrigation. This project mainly aims at reducing water point pollution and water scarcity reduction in a region where this problem is particularly important in summer (McCann, 2012). Currently, urban wastewater is almost entirely treated in plants and 100 hm$^3$ is used in agriculture (Ayala, 2014). However, a possible use of the residue as a fertiliser could be discussed. On another interesting topic, the project RiverPhy$^{161}$ (2012-2014) focuses on the rehabilitation of a heavy metal contaminated riverbed using the phytoextraction technique$^{162}$ in the Guadalentin River (Lorca, Murcia). The project is based on the phytoremediation technique that allows cleaning up pollutants (i.e. metals, salts and organics) from the environment. A similar project could be imagined for removing the nutrients from the drainage water.

Feedback from the regional conference suggests that another opportunity could be to reuse wastewater without removing the nutrients. This would allow decreasing the quantity of fertilisers that farmers need to buy. However, if the farmers do not want to use wastewater as a fertiliser, they must ensure that water complies with the legal requirement for drinking water before discharge it into the waterways. This technique is one of the topics of the on-going European project Carbrowth$^{163}$ that focus on horticulture production. Several regional organisations participate in this project.

**5.4.4.4 Other good practices**

The following measures do not particularly refer to farm level practices. Nevertheless, the measures may contribute to the reduction of environmental impacts of agricultural pollution in coastal waters in the region of Murcia.

**Further develop intercropping in orchards and develop agroforestry in the region**

In the Segura River basin, 20 % of the area is occupied by silvopastoral systems (forest-pasture-ruminants) and 40 % of the surface is occupied by agrosilvopastoral systems (livestock-cereal-rangeland). In the other 40 % of the basin (lower coastal areas) agroforestry systems do not exist.

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$^{161}$ Rehabilitación de un cauce fluvial contaminado por metales pesados mediante fitoextracción

$^{162}$ Use of plants to transport and concentrate metals from the soil into the harvestable parts of roots and above-ground shoots

$^{163}$ www.carbgrowth.eu
permanent crops are among the main production in Murcia, the coupling of fruit trees with cereals for instance or with sheep grazing would allow soil quality to be improved. In addition to the intercropping in orchards, landscape elements such as riparian buffer strips or windbreaks can also be relevant for horticulture or cereal production. Particularly in the Murcia region, water shortages in crop production could benefit from the trees’ shade, which lowers evapotranspiration rates. In the orchards in Murcia, this technique can also be applied in order to avoid barren soil between the tree lines despite social pressure to conform to traditional management. It may also decrease the land demand in case of the use of permanent crop land for other purposes.

**Protect and restore wetlands near the Mar Menor lagoon**

Alvarez-Rogel, et al. (2005) have shown that the wetland surrounding the lagoon acts as a filter to reduce nutrient concentrations before the polluted water flows into the Mar Menor. The marsh significantly helps to control the pollution of the coastal waters. The conservation of such natural filters may have positive impacts on the environment. Moreover, the marsh filters the highly polluted waters outflowing from the waterbeds canalisations. The use of coastal wetlands to treat the polluted waters, in the area of the Mar Menor lagoon in Murcia, is a management practice which should be considered by governmental and administrative decision makers in order to improve the conservation of the natural resources in the zone (Alvarez-Rogel, et al., 2005).

**Retain nutrients in constructed wetlands (CWs)**

The study of García García (2013) shows that this technique is very well adapted to arid areas such as the south of Spain. The studied wetlands removed 51 to 91 % of N contained in the wetlands according to the location and the type of wetlands (surface and subsurface wetlands164). In particular, the nitrate removal varied from 72 % at the surface to 93 % at the wetland subsurface. The nitrate removal was higher in summer, coinciding with the increase of the water temperature and the decrease of wetland discharge (García García, 2013). Depending on the wetlands, nitrogen can be emitted to air as N\textsubscript{2} or N\textsubscript{2}O if the denitrification process is incomplete. In that case, it may have a negative impact on climate change. However, the global impact of the wetlands on climate change should also consider the methane emissions and the carbon stored that vary according to the wetlands (VanderZaag, et al., 2010; Mander, et al., 2014). In the study of García García (2013), the phosphorus retention capacity of wetland was unequal since phosphorus was retained in one of the studied wetland and exported in the other studied wetland. Lastly, the increase in the wetland length did not cause a proportional increase in the nutrient retention (García García, 2013).

This system is mostly suitable for small and medium pig farms, since large farms produced an excess of slurry to be processed in a short period of time. Some research projects are currently testing and improving the effects of constructed wetlands on the liquid fraction of pig manure (Sánchez-García, et al., 2010; Caballero Lajarín, et al., 2012). This technique allows a reduction of N of almost 70 % and P about 80 %, caught by plants (Caballero-Lajarín, et al., 2009; Bayo, et al., 2012).

Due to the natural sedimentation of the solid material in the CW, the liquid fraction can be used for fertigation, although the remaining solid components and salts may contribute to the clogging of droppers. Additionally, considering the content of salts in the purified pig slurry after the CWs, it should not be used as fertiliser in a continuous way due to the risks of soil and plant salinisation. Conversely, CWs need a low qualified workforce, low investment and maintenance.

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164 The surface wetlands expose the water to the atmosphere while the subsurface wetlands maintain the water below the surface layer. Subsurface water flows can be horizontal or vertical.
5.5 North-Brabant (NL)

Box 8 – North-Brabant case study - In brief

North Brabant has the highest nitrogen and phosphorus surplus of the studied regions. The current production and application of nutrients in agriculture in North-Brabant is still imbalanced. The production of nutrients per hectare is significantly higher than the levels of application allowed by the legislation implementing the Nitrates Directive in the Netherlands. This imbalance is further illustrated by the current status of regional waters, which are still suffering from high nutrient loads, eutrophication, and the ammonia and nitrous oxide emissions which have an impact on natural areas. In 2013, less than 20% of the surface water bodies in the Meuse river basin (including North-Brabant) met the WFD quality standards for total nitrogen concentration in the Meuse River Basin Management Plan. The average phosphorus concentration was 0.23 mg P/L, which exceeds the current WFD objective in the Netherlands (0.15 mg P/L) in 2012.

In the region, the observed impacts are mostly due to the intensive livestock farming that induces a high production of manure. Considering the lack of profitable manure transfer and the lack of attractive and cheap techniques for processing, manure is applied on field potentially exceeding the crops needs. The application of inorganic fertiliser in excess is also a cause of nutrient losses, for instance in the form of ammonia and nitrous oxide emissions. Lastly, the losses of nutrients are enhanced by natural factors such as abundant rainfalls, the sandy soil and the regional water system that increase leaching risks.

The economic damage of the nutrient surpluses and its consequences are significant. In the river basin management plan for the Meuse, which includes North Brabant, for the period 2009-2015, the investments for preventive measures addressing diffuse sources (mainly agriculture) are calculated at € 65 million for the period 2009-2027. The restoration of sensitive Natura 2000 areas amounts to € 16 million for North-Brabant. The improvement of water quality in North Brabant would provide € 583 000 per year of additional revenues related to tourism and recreational activities and avoid a decline of housing prices due to poor water quality of 2 to 7% at national level.

In North Brabant, the first set of measures aims to reduce the local source of pollution by controlling the geographic distribution of livestock, adapting the feeding strategies, processing manure to facilitate its use and transfer and transferring manure. The second set of measures focuses on the control of nutrient losses by increasing manure collection through the reduction of the grazing period, preferring drip irrigation, improving the drainage systems and better implementing buffer strips near watercourses.

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North-Brabant is a region located in the south of the Netherlands. It borders the Zeeland and South-Holland provinces in the west, the Gelderland province in the north and the Limburg province in the east.

Belgium borders North-Brabant to the south, and there are a number of small, cross-border rivers like the Reusel, Beerze, Marke and Dommel (see Figure 76). In the western part, North-Brabant borders the Scheldt estuary. In the North, it is bordered by the Meuse river which (downstream part of the Meuse river basin). From an economic perspective, North-Brabant is centrally positioned between the port cities of Rotterdam (north-west) and Antwerp (south-west) and the German Ruhr area (east). There are four larger cities in the heart of the province: Breda, Eindhoven, ’s-Hertogenbosch and Tilburg (Figure 77). Eindhoven is the centre of the high-tech industry of the so-called ‘Brainport’ area, which is a concentration of manufacturing industries.

North-Brabant is the leading agricultural region in the Netherlands. In 2014, it had the largest UAA with 244 670 ha, representing 13.3 % of the national UAA (Central Statistics Office of Netherlands, 2015).
The region accounted for 17.5% of the value of the national agricultural production in 2012 (Eurostat, 2015h)\textsuperscript{166} and also has the highest number of farms in the Netherlands with 17.6% of the national number of holdings (Central Statistics Office of Netherlands, 2015).

Livestock is the main agricultural production in North-Brabant. About 30% of the intensive livestock in the Netherlands is located in North-Brabant (PBL, 2012). In 2014, 54.5% of the holdings in North-Brabant were grazing animal farms while 19.5% of the farms raised housed animals (Central Statistics Office of Netherlands, 2015). Farms in North-Brabant held 5,701,000 pigs in 2014, representing 47% of all pigs in the Netherlands. In addition, 33% of all goats (146,000) and 27% of all chickens (27.4 million) in the Netherlands were reared in the region. Finally, there were 671,000 head of cattle in 2014, which is 16.5% of all cattle in the Netherlands (Central Statistics Office of Netherlands, 2015).

In 2014, arable land represented 26% of the UAA with 63,430 ha in 2014 and 36% of the holdings in 2014, which is higher than the average national average (29.3%). The main products were potatoes (968 kt) and sugar beets (790 kt) in 2014. The same year, the main cereal was by far wheat (107 kt). The share of farms with horticultural production was 19.6% for open horticulture and 5.7% for production under glass (Central Statistics Office of Netherlands, 2015). Grassland and fodder crops covered 66% of UAA in North-Brabant with 161,670 ha in 2014. The remaining 8% was horticulture (Central Statistics Office of Netherlands, 2015).

5.5.1 Notable impacts of nutrient surplus

In North-Brabant, the main impacts are caused by nitrogen and phosphorus surpluses that mostly affect waters and natural conservation areas (Natura 2000 areas). Through run-off of nitrogen and phosphorus from agricultural land, waters from ditches to regional waters and lakes in natural areas are affected. Natural conservation areas are also affected by air pollution through ammonia volatilisation and nitrogen deposition. Potassium excess is not a major issue in this region.

In 2012, the surplus of nutrients in the Netherlands was 113 kg N/ha UAA and 6.1 kg P/ha UAA (own calculations based on Statistics Netherlands (2014c) data). There are no figures on the nutrient balances available at North-Brabant level (Leip, et al., 2013). In North-Brabant, 45% of the farms have a manure surplus, which is due to the fact that North-Brabant has a high share of intensive livestock farms (Statistics Netherlands, 2014c).

**Impacts of nitrogen losses**

The current application and supply of nutrients in agriculture in North-Brabant is imbalanced. In 2013, the production of N from animal manure – excluding N losses to air – were measured at 93,820 tonnes (380 kg N/ha UAA) (Statistics Netherlands, 2014c). The N losses to air were measured at 15,310 kg N. The production of nitrogen per ha is significantly higher than the levels of applications allowed by the Nitrates Directive for the Netherlands. Note that for about 5% of UAA in North-Brabant, i.e. fallow land and land with organic fertiliser crops, it is not allowed to apply manure according to Dutch Fertiliser Act and the legislation implementing the Nitrates Directive in the Netherlands (Statistics Netherlands, 2014d).

This imbalance is further illustrated by the current status of regional waters, which are still affected by eutrophication. In addition, there are ammonia emissions which have an impact on natural areas (PBL, 2010a). Agriculture is the main cause of both environmental problems (PBL, 2012).

\textsuperscript{166} Agricultural output, production value at basic price
The nitrogen surplus in the soil has decreased in the last two decades in the Netherlands (PBL, 2012). In the case of the sandy soils, as in the eastern part of North-Brabant region, the nitrogen surplus has hardly declined over the years (PBL, 2012).

**Excessive nitrogen load in freshwater**

In 2012, total supply of nitrogen to soil in Dutch agriculture was 547 million kg N to UUA (313 kg N/ha), which consisted of 308 million kg N manure application (176 kg N/ha), 202 million kg N fertiliser use (115 kg N/ha), and 37 million kg N other sources (21 kg N/ha). Excluding air emissions, the nitrogen uptake of crops was approximately 60 % of total nitrogen applied to soil and almost 10 % of nitrogen was transferred to surface waters through N-leaching and run-off. About 30 % of the nitrogen was accumulated in the soil and groundwater (own calculation based on Statistics Netherlands (2014c) data).

![Figure 78 – Nitrate concentrations of run-off water from a sample of farms in the Netherlands in the period 2007-2010](source)

Figure 78 shows the nitrate concentration in run-off water from a sample of farms. Farms in North-Brabant have concentrations exceeding 100 mg NO$_3$-L in their run-off water (either ditches next to plots or top layer of groundwater tables). The average concentration of nitrate in the sandy south region was 106 mg NO$_3$-L (Hooijboer & De Klijne, 2012) over the period 2007-2010. Therefore, the farms in North-Brabant contribute to the high nitrogen concentrations in the region’s waters.

In 2013, less than 20 % of the surface water bodies in the Meuse river basin (including North-Brabant) met the WFD quality standards for total nitrogen concentration in the Meuse River Basin Management Plan (Dutch Government, 2014b). In 2009, the groundwater quality in the areas with sandy soils – large part of North-Brabant – in the Meuse river basin did not meet the WFD objective for nitrates (Dutch
Government, 2014b). Agriculture is largely responsible for the high concentrations of nitrate in water. In addition, the nitrogen and phosphorus concentrations of regional waters in North-Brabant are also heavily affected by nutrient losses in upstream parts of rivers basins in Belgium, where there is also intensive agriculture (Dutch Government, 2009).

**Ammonia (NH₃) pollution in the air**

In the Netherlands, the agricultural sector is responsible for more than 90% of the national NH₃ emissions and as a consequence was also the largest contributor (42 % in 2012) to nitrogen deposition (Compendium voor de leefomgeving, 2014a). The ammonia emissions from agriculture have decreased since the early 1990s. In the last decade, the decrease is not as large as in the 1990s. In 2000, the ammonia emissions from agriculture amounted to 143 million kg NH₃ and then decreased to 105 million kg NH₃ in 2010.

More than half of the ammonia emissions stem from stables and manure storage (see Figure 79). The use of manure contributes to 34 % of total ammonia emissions. Use of mineral fertiliser and excretion during grazing represented with small fractions contribute smaller shares of the emission (13 % and 2 % respectively). The share of Dutch agriculture in acid deposition to soil and water was 31 % in 2012 (Compendium voor de leefomgeving, 2014b).

![Figure 79 – Ammonia emission from agriculture in the Netherlands in 2012](source: Compendium voor de Leefomgeving, 2014c)

In the areas with livestock farms, as in the eastern part of North-Brabant, the nitrogen deposition contributes to the acidification of the soil and it can reach 6 300 mol/ha (Compendium voor de Leefomgeving, 2014d) in the natural conservation areas, which exceeds the critical deposition values (i.e. 1 400 mol/ha for extremely sensitive nature such as peat land and marshes). This high nitrogen deposition is primarily caused by the high ammonia emissions from livestock farming. Ammonia is emitted close to the ground, so that its radius of spreading is limited.

In 2010, North-Brabant ranked second (with 1 850 mol/ha/year) in the list of nitrogen deposition after its neighbouring province of Gelderland (1 940 mol/ha/year). Almost half of this deposition comes from within the Dutch territory, the other half from the North Sea or other territories. About 70 % of North-Brabant’s deposition originating from the Netherlands comes from North-Brabant. For the most part (almost 80 %) this deposition is produced by the agricultural sector, other emitting sectors being public transport, energy, industry and waste disposal (Velders, et al., 2010).
The nitrogen deposition has an impact on nature and its ecosystems. It might make vegetation more vulnerable to droughts, storms and plant diseases (Velders, et al., 2013). In addition, the composition of the natural vegetation of natural areas might change due to modifications in the soil conditions caused by nitrogen deposition. Heathlands and dunes, for instance, are being converted into grasslands due to the changes in soil conditions caused by nitrogen deposition (Compendium voor de Leefomgeving, 2014d).

**Greenhouse gas emissions including nitrous oxide (N\textsubscript{2}O) pollution in the air**

In 2004, North-Brabant’s agriculture was responsible for approximately 5.5 Mton CO\textsubscript{2}-equivalent greenhouse gas emissions. This was about 20% of the national emissions of the agricultural sector (both at the farm level and elsewhere in the supply chain). The majority of greenhouse gas emissions were from methane and nitrous oxide. Animal sectors produced by far the highest emissions: the pig, dairy and poultry sector in 2007 emitted 2.0, 1.5 and 0.9 Mton CO\textsubscript{2}-equivalent, respectively (CLM, 2007).

Approximately 22% of North-Brabant’s agriculture GHG were due to N\textsubscript{2}O emissions. The agricultural greenhouse gas sources can be divided into three components: on-farm emissions, production chain emissions, and manure use (see Figure 81). The on-farm emissions include emissions from the use of nitrogen fertiliser (N\textsubscript{2}O), fuel use on the farm (CO\textsubscript{2}), digestion of the animals (CH\textsubscript{4}), manure storage (N\textsubscript{2}O and CH\textsubscript{4}), manure use (N\textsubscript{2}O) and stables (N\textsubscript{2}O). The production chain emissions consist of the production of nitrogen fertiliser (N\textsubscript{2}O and CO\textsubscript{2}) and fodder produced elsewhere (often abroad). The “manure use” component includes GHG emissions related to the use of manure produced in North-Brabant, but used elsewhere (other parts of the Netherlands). Almost half of the GHG emissions are...
produced on the farm. The use of fertiliser and fodder produced elsewhere is one third of the GHG emissions.

Figure 81 shows that livestock farming is largely responsible for the emission of greenhouse gases from agriculture in North-Brabant. Pig farms have the highest contribution with about 2 million tonnes CO₂-eq., which is more than one-third of the total greenhouse gas emissions from agriculture. Dairy cattle are responsible for a quarter of the greenhouse gas emissions. The assumption is that all animal manure is applied, and the emissions are assigned to the manure producing sectors (dairy cattle, meat cattle, pigs and poultry). The share of arable farming and horticulture is less than 5 %, because for the calculations of the results presented in Figure 81, CO₂ emissions of animal manure applied in arable farming are assigned to the manure producing farms (CLM, 2007).

**Impact of phosphorus losses**

The current application and supply of phosphorus in agriculture in North-Brabant is imbalanced. In 2013, the production of P from animal manure was measured at 39 410 tonnes (160 kg P/ha UAA) (Statistics Netherlands, 2014c). The production of phosphorus per ha is significantly higher than the levels of applications allowed by Dutch Nitrates Action Programme.

**Excessive phosphorus load in freshwater**

**Groundwater** - Phosphorus saturated soils are defined as soils with phosphorus concentrations in groundwater (the highest level of the groundwater table on average) exceed 0.15 mg/L, which is the maximum tolerable risk (MTR) value for P concentration in ground and surface water\textsuperscript{168} and also the current WFD objective in the Netherlands. For most soil types, this groundwater concentration 0.15 mg/L is considered as met if 25 % of the phosphates binding capacity, i.e. of the top soil layer (ground level

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\textsuperscript{168} The MTR value for phosphorus is the highest level of P-concentration in waters for which it is expected that there will not be any negative impact on the environment and/or human health according to the Dutch National Institute for Public Health and the Environment (part of the Dutch ministry of Health, Welfare and Sports), (National Institute for Public Health and the Environment, 2015)
and average ground water table level), is used (MNP, 2007; Compendium voor de Leefomgeving, 2014e).

Figure 82 – Share of phosphorus saturated areas in the Netherlands at the end of the 1990s

Figure 82 shows the share of phosphorus saturated areas in grid cells for the Netherlands at the end of the 1990s. More than half of the soils of agricultural areas in the Netherlands, including North-Brabant, were saturated with phosphorus. In North-Brabant, almost all grid cells have a share of 50% of saturated areas. In the eastern part of North-Brabant, the surplus is slightly higher than in the western part of North-Brabant. Phosphorus saturated areas are responsible for eutrophication of surface and ground waters in North-Brabant. Drinking water production from groundwater in North-Brabant is hardly affected by saturated areas, because groundwater is abstracted from sandy soils. The areas of groundwater abstractions are nature protected areas which are not affected by agricultural practices (Brabant Water, 2014).

**Surface water** - In 2010, Dutch agriculture emitted 3.7 million kg P to water (Statistics Netherlands, 2014e). This is 18.6% of the total phosphorus produced in the Dutch economy in 2010. There is no information on the share of North-Brabant in these emissions. The run-off of phosphorus is related to the soil type. (Hooijboer & De Klijne, 2012). The concentrations of phosphorus in run-off waters are negligible for the regions with sandy and loessial soils. For clay, the concentration was 0.23 mg P/L, and for peat soil it was 0.45 mg P/L in 2012.

Despite the fact that there are low P-concentrations in run-off water, moderately high P-concentrations of surface water can also be found in North Brabant along with moderate to low concentrations (see Figure 83). For ditches, low P-concentrations mean 0.22 mg/L or lower, and high P-concentrations mean 0.44 mg/L or higher (PBL, 2010b). For brooks a low concentration means 0.14 mg/L or lower and a high concentration means 0.19 mg/L. Finally, the low concentrations for canals mean 0.15 mg/L or lower, and the high concentrations mean 0.44 mg/L or higher. In 2009, slightly more than 20% of the surface
water bodies in the Meuse river basin (including North-Brabant) met the WFD quality standards for phosphorus concentration i.e. 0.15 mg/L (Dutch Government, 2009).

**Eutrophication in freshwater**

Phosphorus saturated areas have a high potential to cause eutrophication of surface waters and groundwater bodies that are linked to surface water bodies. The ground waters and surface waters in North-Brabant largely suffer from eutrophication (Groenendijk, et al., 2012) (Dutch Government, 2009). Moreover, natural conservation areas are affected by eutrophication and acidification of the soil through nitrogen deposition, which affects the natural vegetation and ecosystems (PBL, 2010a).

The high concentrations of nitrogen and phosphorus in surface waters cause algae blooms in the waters with low flow capacity. In combination with high temperatures, blue algae can appear in water courses with possibly human health risks (Pires, 2010). For freshwater, phosphorus is the dominant nutrient for algae bloom. In North-Brabant, ditches and lakes are prone to algae blooms, but these types of waters are hardly present in North-Brabant or are often not monitored. The regional water system in North-Brabant consists primarily of brooks, canals and rivers with higher flow capacity. The high flow capacity avoids the negative consequences of eutrophication (algae bloom) so nitrogen and phosphorus in waters with high flow capacity is transferred downstream i.e. Meuse river, and the North Sea coastal areas of the Netherlands. Algae bloom is not a major issue in North-Brabant.

### 5.5.2 Causes of nutrient losses

**Farming system and agricultural practices**

**Intensive livestock farming** - The primary reasons for the nutrient surplus are the large number of (intensive) livestock and the high production of manure, in particular due to many intensive meat and dairy cattle farms. Both types of farms use arable land to produce fodder for meat and dairy cattle. The production of nitrogen and phosphorus from manure in 2013 in North-Brabant was relatively high (Figure 84). In the eastern part of North-Brabant, there are many pig farms and the production exceeds the level of 300 kg and 120 kg per ha for nitrogen and phosphorus, respectively. In the north-western part of North-Brabant the production of nitrogen and phosphorus is significantly lower than in the eastern and
middle part of the region. The total production of nutrient in North-Brabant in 2013 was 93,840 tons of N (380 kg/ha N UAA) and 39,410 tons of P (160 kg/ha P UAA).

Figure 84 – Nitrogen (left) and phosphorus (right) from manure production (livestock) in the agricultural areas in the Netherlands measured in kg/ha in 2013.

Excessive application of organic and inorganic fertilisers on field - The arable areas are used to apply the animal manure up to the allowed regulatory levels of manure use according to the Nitrates Action Programmes for the Netherlands. The excessive application of manure, but also chemical fertilisers, compared to crop needs contributes largely to the nutrient (nitrogen and phosphorus) leaching and run-off to groundwater and surface water.

In 2009, the nitrogen surplus of dairy farms on the sandy soil in the south of the Netherlands ranged from 145-150 kg/ha N UAA (Van der Ham & Daatselaar, 2012). The phosphorus surplus of dairy farms in this region was 75 kg/ha P UAA. Both nitrogen and phosphorus surpluses for dairy farms showed a downward trend over the last two decades. For arable farms, the nitrogen and phosphorus surplus on sandy soils in 2012 were 128 kg/ha N UAA and 39 kg/ha P UAA (Van der Ham & Daatselaar, 2012). Although the nutrient surplus is decreasing, it still contributes to eutrophication of waters and acidification of soils through nitrogen deposition.

The nitrous oxide (N\textsubscript{2}O) pollution in the air is partially caused by agriculture and the use of chemical fertiliser in particular. Since 1990, there has been a gradual decline of the nitrous oxide (N\textsubscript{2}O) pollution from agriculture due to a gradual decline of chemical fertiliser use (LEI, 2013). CLM calculated the N\textsubscript{2}O emissions for 2007 at 1,200 kt CO\textsubscript{2}-eq. Emissions from fertiliser use were 200 kt CO\textsubscript{2}-eq., while emissions from manure storage amounted to 200 kt CO\textsubscript{2}-eq., and manure use accounted for 800 kt CO\textsubscript{2}-eq. (CLM, 2007).

Lack of profitable manure market - Given the current system of negative prices on the manure market in North-Brabant and the rest of the Netherlands, there is still an incentive for high levels of manure application. Indeed, due to the higher supply than demand, manure tends to be seen as a waste product rather than a valuable fertilising product. As a consequence, Dutch arable farmers in the northern and south-western parts of the country get paid when they accept pig manure. Pig farmers from the south-eastern Netherlands had to pay approximately € 18 per tonne to get rid of their manure surplus (Luesink, et al., 2014).

Lack of attractive and cheap alternatives for reducing manure surplus in North-Brabant - Manure processing capacity in North-Brabant (and the rest of the Netherlands) is still in development, and still has significant costs. The costs for total manure processing in the Netherlands range from € 20 to € 25 per tonne of liquid manure. For separation only, the costs range from € 2-8 per tonne (PBL, 2010a). In addition, most processes require large quantities of manure and are generally not suitable for on-farm implementation, except for separation.
Environmental conditions

Abundant rainfall – High rainfall in sandy soils may induce high levels of nutrient run-off, particularly if the rainfall occurs shortly after the application of the manure. The precipitation in 2013 was variable in North-Brabant (see Figure 85). In the eastern part of North-Brabant, the precipitation was from 630 to 770 mm/yr. In the western part of Brabant, there were higher levels of precipitation.

Regional water system – The water system in North-Brabant is rather dense. Next to the brooks and canals in the regional water system of North-Brabant, there is also a wide and dense network of small ditches (Dutch Government, 2009). These ditches are used for drainage purposes in case of periods of abundant rainfall in the summer, and infiltration of groundwater in the winter. In areas with sandy soils and drainage systems, there is an increased probability of run-off of nutrients to surface water. There is an increase in implementing controlled drainage systems to retain water in the top layer of the soil. It decreases the risk of nutrients leaching to groundwater, but it increases the risk of run-off to surface waters. The net impact on eutrophication is unknown.

Soil texture that increases leaching and run-offs risks – The soil type of a large part of North-Brabant is sand. The characteristic of this sandy soil in North-Brabant is that it cannot retain water for a long time. Nutrients are easily leached to groundwater especially in the periods of abundant rainfall.

5.5.3 Costs of the environmental and health effects

5.5.3.1 Social economic description of the study area

North-Brabant has a population of 2 490 000 in 2015, which is about 15 % of the Dutch population. The GDP of North-Brabant was € 96 billion (current prices) in 2013, representing 15 % of the national GDP (Central Statistics Office of Netherlands, 2015). In 2012, the share of the total value added of the
agricultural sector in North-Brabant was 18.6% (Eurostat, 2015c). North-Brabant’s industrial sector is relatively large: its share is 30% of all economic activities, whereas Dutch average is 25% (Central Statistics Office of Netherlands, 2015).

As aforementioned, the rivers in North-Brabant are affected by nutrient loads from upstream areas, such as Belgium and France and the Dutch province of Limburg. Agricultural activities in North-Brabant cause run-off of N and P, potentially leading to eutrophication of the water. This eutrophication of waters is the major impact of nutrient surplus in North-Brabant. The economic sectors affected are primarily the tourism and recreation sector. The drinking water production is only marginally affected by the nutrient surpluses today, because the groundwater wells are protected from agriculture influences.

5.5.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in North-Brabant, a wide range of literature was reviewed. Based on the classification of economic damages caused by environmental impacts presented in Annex 14, costs found in the literature for each damage category (i.e. CRC, UVD, etc.) are presented below along with explanations on how these costs have been estimated.

It should be noted that all cost data reported in this section are taken from the primary studies found in the literature, and expressed in the value of the year when the study was conducted.

Clean up and restoration costs (CRC)

Preventive measures to reduce nutrients concentration in water

The Nitrates Action Programme implementing the Nitrates Directive in the Netherlands sets out a number of requirements on manure use, and consequently, on nutrient surpluses. However, the measures of the Nitrates Action Programme 2014-2017 are insufficient to meet the water quality objectives of the Water Framework Directive. Therefore, there are additional measures in the river basin management plans. Note that the river basin management plans for Dutch river basins also take into account the improvements of water quality of upstream waters abroad.

For the Netherlands, the annual costs of measures to meet the WFD targets amount to € 390 million (PBL, 2008). The majority of measures have to be taken in the regional water system for a total cost of € 325 million per year, and the rest in national waters (Meuse, Rhine, Scheldt, and Ems river). North-Brabant is largely located in the Dutch part of the Meuse river basin. In the river basin management plan for the Meuse for the period 2009-2015, the investments for measures addressing diffuse sources (mainly agriculture) are calculated at € 65 million for the period 2009-2027 (i.e. € 3.6 million /yr during the period) (Dutch Government, 2009).

Restoration of nature due to N-deposition

In the Netherlands, 133 Natura 2000 protected areas are nitrogen-sensitive. In these areas, nitrogen deposition should be reduced to limit impacts to the environment. In order to achieve this, measures for ecological restoration, hydrology management, and other measures (such as research), are implemented. In North-Brabant, fourteen nitrogen-sensitive Natura 2000 areas can be found. Total costs for restoration are estimated at € 16.1 million (see Error! Reference source not found.), which includes ecological restoration of Natura 2000 areas (€ 3.5 million), hydrological measures to improve the ground water tables in the Natura 2000 areas (€ 12.3 million), and some additional measures (€ 0.3 million) (Dekker & Bruinsma, 2011).
The restoration costs for nitrogen deposition in North-Brabant are related to the damage costs of Natura 2000 areas in the province. Approximately, 34% of the nitrogen causing the deposition in North-Brabant (agriculture 26% and industries and transport 8%) originates from the region itself. The rest of the nitrogen originates from the rest of the Netherlands, North Sea or abroad. About 13% of the nitrogen has an unidentified origin.

![Figure 86 – Distribution of origin of nitrogen deposition in Natura 2000 areas in North-Brabant](image)

**Nutrient removal**

In North-Brabant, the company Brabant Water uses surface and groundwater resources for the production of drinking water. However, these surface and groundwater resources are protected areas and the nutrient concentrations of these water resources already meet drinking water standards, which are stricter than the standards for nutrient concentrations derived from the WFD. The study of KIWA Netherlands, the organisation that certifies water quality amongst others, claims that there will be no cost reduction for drinking water production if the nutrient concentrations of other groundwater resources would improve, because the objectives for water abstraction already meet the criteria for drinking water production (KIWA, 2008). The areas of water abstraction for drinking water production are already protected areas.

Although KIWA claims zero costs for improved nutrient concentrations, the purification of drinking water can involve nutrient removal to some extent. For the drinking water production in North-Brabant, the production costs amount € 0.97 / m³ (Dinar, Pochat, & Albiac-Murillo, 2015). Note that the purification of surface and ground water (i.e. removal of nutrients from the water amongst others) is part of the drinking water production process. It is difficult to disentangle the costs for nutrient removal in the production process. The removal of nutrients involve operational costs and capital costs.
Table 42 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Type of costs</th>
<th>Description</th>
<th>Costs</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
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<td>Water quality improvement through the implementation of preventive measures</td>
<td>Cost effective WFD-measures in regional waters</td>
<td>€ 325 million / yr</td>
<td>The Netherlands</td>
<td>Annually, between 2008 and 2027</td>
</tr>
<tr>
<td></td>
<td>Cost effective WFD-measures in national waters (“Rijkswateren”)</td>
<td>€ 65 million / yr</td>
<td>The Netherlands</td>
<td>Annually, between 2008 and 2027</td>
</tr>
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<td></td>
<td>Investments in measures for diffuse and point sources169</td>
<td>€ 65 million</td>
<td>Meuse river basin</td>
<td>2010-2027</td>
</tr>
<tr>
<td>Restoration of natural areas due to deposition of N</td>
<td>Ecological restoration (management, hydrology, other measures)</td>
<td>€ 16.1 million</td>
<td>North-Brabant</td>
<td>2012-2018</td>
</tr>
<tr>
<td>N removal</td>
<td>Total costs drinking water (from ground water sources)</td>
<td>€ 0.97 / m³</td>
<td>North-Brabant</td>
<td>2012</td>
</tr>
<tr>
<td></td>
<td>Operating costs</td>
<td>€ 0.60 / m³</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cost of capital, depreciation, taxes</td>
<td>€ 0.37 / m³</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cost reduction from better ground water quality</td>
<td>0</td>
<td>The Netherlands</td>
<td></td>
</tr>
</tbody>
</table>

Use value damages (UVD)

There is little known on the use value damages (UVD) of the nitrogen deposition, N leaching and P saturation of the soil. PBL summarised some economic damages effects but most of them are qualitative without any quantitative evidence (PBL, 2008). Tourism has suffered from damage due to the poor water quality in the Netherlands. This includes negative impacts on sport fishing and swimming in surface waters. One study on benefits of the WFD estimated that the improvement of water quality would provide € 583 000 per year of additional revenues related to tourism and recreational activities (swimming and sun bathing, water activities, etc.) (Twynstra-Gudde, 2007). This includes revenues of expenditures on food and drinks as well as on revenues of yacht basins. Although no evidence about the related damage costs was found in the literature, a 2008 study claimed that the damage for tourism was modest and the shipping sector hardly noticed any impacts (Ecorys, 2008). Brouwer et al. (2007) showed that the housing prices would be significantly higher (2-7 %) if water quality of surface water would improve (Brouwer, et al., 2007).

Table 43 – Use value damage found in literature

<table>
<thead>
<tr>
<th>Type of costs</th>
<th>Description</th>
<th>Costs</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loss of revenues (tourism)</td>
<td>Tourism and recreation</td>
<td>€ 583 000 per year</td>
<td>North-Brabant</td>
<td>2007</td>
</tr>
<tr>
<td></td>
<td>Declining housing prices due to poor water quality</td>
<td>2-7 %</td>
<td>The Netherlands</td>
<td>2007</td>
</tr>
</tbody>
</table>

169 In this case, the total investments of supplementary WFD measures are mentioned. The annual costs of these measures will be approximately 10 % of total investments (i.e. € 6.5 million). Moreover, the costs cover the Meuse basin in the Netherlands and North-Brabant is part of it.
**Passive use value damages (PUVD)**

Nutrient surplus can also lead to air pollution and consequently cause physical and economic damages to human health and ecosystems. There are no particular studies available for North-Brabant.

**Effects on ecosystems**

Brink & Van Grinsven (2011) estimated that the average damage cost of deposition of NH$_3$ on terrestrial ecosystem ranges between € 2 and € 10 / kg N in the EU27 (see Table 45) (Brink & van Grinsven, 2011). The lower bound is derived from the EU NEEDS project (Ott, et al., 2006), representing the cost of biodiversity and ecosystem restoration.

**Health effects of air pollution**

The economic losses can be both tangible and intangible, expressed in terms of direct lost market and non-market values (Brink & van Grinsven, 2011). Taking the cost of human health due to air pollution as an example, the market value losses are typically evaluated in term of direct expenditures incurred for medical treatment and hospitalisation, reduced wages and productivity due to the lower performance affected by pollution-related disease. On the other hand, the non-market value losses can only be estimated in a contingent market, in which a survey-based valuation technique can be used to assess the individual’s willingness to pay for avoiding the risk of being infected by the disease or death. Based on the use of different valuation techniques, Brink & Van Grinsven (2011) estimated average damage cost to public health and ecosystems in the EU-27 caused by NH$_3$, NO$_x$ and N$_2$O emissions respectively.

A Dutch study (CE, 2010) calculated the passive use value for NH$_3$ and NO$_x$ emissions (€ 27.6 and 10.6 / kg N respectively) which are above the upper limit of the estimations of Brink & Van Grinsven (2011) for the EU. In addition, CE (2010) also calculated the damage costs for P, which were € 1.78 / kg P. Given the surplus of N and P for North-Brabant regards the exact pollutant (NH$_3$, NO$_x$), a simple calculation of the shadow prices times the amounts of surpluses in North-Brabant would mean that the damage costs for ecosystems in North-Brabant would amount € 560 million for N and almost € 60 million for P (see Table 44). Note that we cannot simply add the amounts, because the total damage might be overestimated due to possible double counting.

The health costs are thousands of euros higher in North-Brabant than in the rest of the Netherlands due to the environmental issues related to nitrogen deposition (Twynstra-Gudde, 2007). Note however that these estimations of health costs are not necessarily related to the surplus of nitrogen and/or phosphorus in North-Brabant.

Based on the emissions surpluses of nutrient in North-Brabant and the Netherlands in 2012, the passive use value of ecosystems can be calculated for nitrogen and phosphorus. The damage costs for ecosystems in North-Brabant are more than half the damage to ecosystems in the Netherlands due to nitrogen surplus, see Table 44.
Table 44 – Passive use value damage for ecosystems in North-Brabant and the Netherlands for nitrogen and phosphates

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Surplus</th>
<th>Unit costs ecosystem from pollutant</th>
<th>Damage ecosystems Calculated</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>mln kg</td>
<td>€/kg</td>
<td>€ mln</td>
</tr>
<tr>
<td>North-Brabant</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N</td>
<td>53.0</td>
<td>10.6</td>
<td>562.5</td>
</tr>
<tr>
<td>P</td>
<td>33.3</td>
<td>1.78</td>
<td>59.3</td>
</tr>
<tr>
<td>The Netherlands</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>N</td>
<td>98.0</td>
<td>10.6</td>
<td>1,036.4</td>
</tr>
<tr>
<td>P</td>
<td>62.6</td>
<td>1.78</td>
<td>111.4</td>
</tr>
</tbody>
</table>

Source: Calculation LEI based on Statistics Netherlands for surpluses and (CE, 2010) for unit costs.

Table 45 – Passive use value damage found in literature

<table>
<thead>
<tr>
<th>Type of costs</th>
<th>Description</th>
<th>Costs</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Negative effect on ecosystems (deposition)</td>
<td>Unit damage costs for N to water</td>
<td>€ 5-20 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for NH₃-N to air</td>
<td>€ 2-10 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for NH₃</td>
<td>€ 27.8 / kg NH₃</td>
<td>The Netherlands</td>
<td>2010</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for NOₓ-N to air</td>
<td>€ 2-10 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for P to water</td>
<td>€ 10.6 / kg N</td>
<td>The Netherlands</td>
<td>2010</td>
</tr>
<tr>
<td></td>
<td>Willingness to pay for biodiversity</td>
<td>€ 1.78 / kg P</td>
<td>The Netherlands</td>
<td>2010</td>
</tr>
<tr>
<td></td>
<td>Willingness to pay for biodiversity</td>
<td>€ 13 000 000 / yr</td>
<td>North-Brabant</td>
<td>2007</td>
</tr>
<tr>
<td>Climate change</td>
<td>Unit damage costs for N₂O-N to air</td>
<td>€ 5-15 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for CO₂</td>
<td>€ 0.25 / kg CO₂</td>
<td>The Netherlands</td>
<td>2010</td>
</tr>
<tr>
<td>Increased health costs</td>
<td>Unit damage costs for N to water</td>
<td>€ 0-4 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for NH₃-N to air</td>
<td>€ 2-20 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for NOₓ-N to air</td>
<td>€ 10-30 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Unit damage costs for N₂O-N to air</td>
<td>€ 1-3 / kg N</td>
<td>EU27</td>
<td>2011</td>
</tr>
<tr>
<td></td>
<td>Health costs</td>
<td>Thousands euros of</td>
<td>North-Brabant</td>
<td>2007</td>
</tr>
</tbody>
</table>

Policy action costs (PAC)

Monitoring costs for Dutch policy to reduce nitrogen deposition to call a halt to deterioration of nature quality will increase. The additional monitoring costs are estimated at € 440 000 to € 770 000 per year on average (Leneman, et al., 2012).
<table>
<thead>
<tr>
<th>Type of costs</th>
<th>Description</th>
<th>Costs</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Additional monitoring costs</td>
<td>Monitoring costs for policy regarding the reduction of N deposition</td>
<td>€ 440 000 to € 770 000 / yr (average)</td>
<td>Netherlands</td>
<td>2013-2020</td>
</tr>
</tbody>
</table>

### 5.5.4 Good practices to reduce nutrient losses at farm level

In North Brabant, the first set of measures aims to reduce the local source of pollution by controlling the geographic distribution of livestock, adapting the feeding strategies, processing manure to facilitate its use and transfer and transferring manure. The second set of measures focuses on the control of nutrient losses by increasing manure collection through the reduction of the grazing period, preferring drip irrigation, improving the drainage systems and better implementing buffer strips near watercourses;

#### 5.5.4.1 What has already been done in the region

Mandatory measures set inter alia the maximum application levels for nitrogen and phosphorus taking into account the type of crop and soil. Other mandatory measures applicable within the region concern the design of livestock stables in order to reduce greenhouse gases (GHG) and ammonia emissions. In the different Nitrates Action Programmes over the years, mandatory measures to reduce manure and its emissions to water, soil and air have been established. The most important measures implemented are: manure transfer, sharpened manure storage regulation, mandatory manure processing for farmers with a manure surplus, low-emission housing for animals and low-emission application techniques. Not all measures are fully implemented yet, such as low-emission housing for animals, as there is a gradual implementation process. Note that all these measures are implemented nationally, although the impact in North-Brabant is significant due to its agricultural structure.

In order to reduce the nutrient surplus in North-Brabant and other parts of the Netherlands, manure is transferred from nutrient surplus areas to nutrient deficient areas. This has been an important measure to reduce nutrient surpluses in the last two or three decades. Manure transfer has been common practice in the North-Brabant region as well as the Netherlands as a whole for over 25 years.

With the Dutch Nitrates Action programme 2010-2013, the requirements of the manure storage capacity were changed due to a shortening of the manure application periods. Farmers with livestock have to be able to store their manure for a longer period of time (7 instead of 6 months) (CUMELA, 2010). This measure is continued in the Dutch Nitrates Action programme 2014-2017. Moreover, the rules for grassland farms derogations have been further tightened: for a grassland farms that would like to make use of the derogation at least 80 % of the total farm area has to be grassland. Also, farms with a derogation are not allowed to use chemical P-fertiliser anymore. Another measure is the reduction of the manure application threshold determined by the derogation in the sandy soil areas of North-Brabant.

In the Nitrates Action Programme 2014-2017, 2,000 km of fertiliser-free zones along waters near natural areas were designated. In North-Brabant, most river banks of the rivers originating from Belgium are designated fertiliser free zones (RVO/Dutch Government, 2005), which means that no manure or chemical fertiliser can be applied on agricultural land within 5 metres of the river bank. Farmers with a manure surplus are not allowed to apply the manure that was not applied on the manure free-zone on other parts of their land (RVO/Dutch Government, 2015). As a consequence, those farmers face higher costs for manure transfer or manure processing. Farmers using chemical fertilisers have reduced costs as they use less chemical fertiliser in the case of fertiliser-free zones. Note that the manure-free zone can be used to grow crops, but the revenues are lower than on land with application of manure or fertiliser.
With respect to reducing air emission, legislation for low emission stables for dairy and for intensive livestock is already implemented in the Netherlands (Dutch Government, 2005). Newly built stables have to fulfill environmental requirements in order to reduce nitrogen emissions to air. The environmental requirements are the incorporation of air cleaners for intensive livestock stables, and low-emission stable systems for dairy and meat cattle. In 2020, all stables for dairy, pigs and poultry have to meet the environmental requirements. Current policy is targeted to speed up the redesign of existing stables to meet the environmental requirements. Ammonia emissions and GHG emissions from stables will decline. For the protection of Natura 2000 areas, there is already an environmental permit system in place for farmers in North-Brabant to control for the nitrogen emissions to air.

In application of the dispositions of the Dutch Fertiliser Act, Farmers with manure surpluses in North-Brabant had to process 30% of their manure surplus in 2014 and 50% in 2015. The manure must either be processed into a manure product which can be exported abroad, or be combusted to generate energy, which involves incineration or gasification to the point that the ashes contain a maximum of 10% organic matter (Dutch Government, 2014a). In 2014, the Dutch Resolution of Fertiliser Use was changed to comply with the Nitrate Directive Programme 2014-2017. The rules for manure application techniques have been sharpened. On grasslands, it is only allowed to use low-emission manure application techniques which inject the manure directly into the top layer of the soil. On arable land, low emission manure application techniques are also effective and cheap methods for reducing ammonia emissions (PBL, 2009). The adverse effects on soil and on meadow birds are limited (PBL, 2009).

Deltaplan Agricultural Water management (in Dutch “Deltaplan Agrarisch Waterbeheer”, DAW) is a promising initiative of LTO Nederland (the Dutch Federation of Agriculture and Horticulture). An entrepreneurial and employers’ organisation, its objective is to contribute to resolving water issues in agricultural areas and to promote an economically sound and sustainable agricultural sector. Given these objectives, agricultural entrepreneurs and water boards work closely together on local water issues to improve water quality and quantity for groundwater and surface water. Farmers implement measures to reduce nutrient emissions and water boards guarantee the supply of fresh water in times of droughts.

5.5.4.2 Good practices to reduce nutrient losses in livestock production

Control the geographic distribution of livestock

The objective of this type of measure is to reduce the excessive use of manure in agriculture in North-Brabant (and the rest of the Netherlands). Such measures will contribute to a significant reduction of ammonia and GHG emissions. Although very effective, the measure would, however, entail significant economic cost for North-Brabant would be extremely high. A reduction of the number of animal heads will not only affect agriculture but also other economic sectors such as food processing industries and transportation in North-Brabant and the surrounding provinces.

The reduction of approximately 20% of the animal heads would result in a balance of supply and demand of manure in terms of phosphorus. It would also imply a loss of GDP of € 725 million for agriculture (Vrolijk, Blokland, Helming, Luesink, & Prins, 2010). If the impact on the whole agro value chain\textsuperscript{170} is considered, the loss of GDP would be estimated at € 2.3 billion (Vrolijk, Blokland, Helming, Luesink, & Prins, 2010). As a result, the reduction of the number of animal heads is socially challenging. Note however that there is still an production rights system in place for pigs and poultry in the Netherlands, so that the number of animals can be controlled. This permit system for pigs and poultry

\textsuperscript{170} Impact on farming, supplying and buying trade, transport, service provision.
will be continued after 2015. The system was to be abolished in 2015 because of the mandatory implementation of low emission housing for pigs and poultry in 2012, according to the general administrative orders for animal housing (Dutch Government, 2005).

For grazing animals, in the past the number of animal heads was controlled by the milk quota. The milk quota system is no longer in place since March 2015. However, the management of nutrients will be controlled by more strict regulation with respect to manure production and processing and by the introduction of a phosphate rights system for the dairy sector.

Overall, considering that the manure production exceeds the carrying capacity of the territory in several areas of the region, it is necessary to consider this measure in order to aim for a durable reduction of nutrient surplus.

**Adapt feeding techniques**

Feeding livestock is often not optimal in the sense that animals may get more nutrients than they actually need, and as a result the digestion process of the animal is inefficient. The digestion process of the livestock can be optimised by changing the diet composition and reducing fodder (PBL, 2010a).

The objective of this type of measures is to change the nutrient amount and nutrient content of feed, so that it will reduce the nutrient content of manure as well. Considering the large number of livestock in North-Brabant and the number of farmers with manure surpluses, measures targeting feeding techniques would have a significant impact in North-Brabant when implemented. It will contribute to lower ammonia and GHG emissions.

**Optimising fodder and diet composition for grazing animals**

For grazing animals, the amount of fodder and the diet composition can be optimised in order to optimise nutrient efficiency of the animals and consequently reduce animal manure and ammonia emissions. The amount of fodder can be reduced and by substituting grass for maize in the diet composition, the nitrogen component per unit of fodder can be reduced (PBL, 2010a). The milk urea will decrease to 20 mg per 100 g of milk (PBL, 2010a) which indicates more efficient use of nitrogen by the animals (PR, 1998). The environmental benefits from optimal fodder and diet composition result in a lower production of manure, i.e. the excretion per animal are reduced from 138 to 120 kg/yr N. Consequently, the nitrogen surpluses are smaller and the ammonia emissions are lower.

**Fodder for pigs with lower protein content**

In the case of pig rearing, farmers can switch to lower protein feed for pigs in order to reduce nitrogen emissions (PBL, 2010a). The environmental benefits are less manure and less excretion of nutrients. The ammonia emissions of pig production will decline as well (Smits, et al., 2012). Less manure implies lower costs for manure transfer, but the farmers will be confronted with higher feeding costs because low protein fodder elements are more expensive (Smits, et al., 2012) and (PBL, 2010a). Pigs with low protein diets (and amino acids) show a slight increase in carcass back fat depth (Carter, et al., 2012c), so that it is expected that revenues of pigs’ farmers implementing low protein diets might slightly decline. The costs will outweigh the benefits for most pig farmers, so that the income of pig farmers will decline. (PBL, 2010a). There are a few farmers who already use low protein feed for their pigs.

Both feeding measures are not specific for North-Brabant and can be implemented throughout the Netherlands. The main reason why they are suggested is the reduction of ammonia emissions and the impact of these emissions on protected natural areas. However, both measures reduce the excretion of nitrogen as well as the excretion of phosphorus.
Since neither measure is specific to North-Brabant, the implementation in North-Brabant would have disadvantages for dairy and pig farmers located in North-Brabant due to higher production costs compared to other regions.

**Reduce the grazing period to increase manure collection**

The share of grassland (and fodder crops) is about two-thirds in North-Brabant and thus there is a large number of grazing animals (mainly cattle, goats, and sheep). In order to collect manure from grazing animals more efficiently, the grazing period of animals could be reduced. In that way, the manure is not unevenly spread on grassland. In addition, the collected manure can be applied later on uniformly over grassland. However, to be efficient, this measure requires low emission stable systems. Stables need to be equipped with slatted floors, so that liquid and solid manure is separated directly to avoid ammonia emissions. The implementation of low-emission stables is in progress. This combination of measures has environmental benefits:

- Reduction of leaching and run-off by improvement of the use of manure;
- Less potential eutrophication results in improved conditions for biodiversity;
- Reduced ammonia emissions leading to less particulate matter formation and lower impacts on human health.

The application of manure is managed and controlled more efficiently, possibly resulting in higher yields when applied to crop fields or grasslands and lower costs from avoided purchase of inorganic fertilisers. In addition, more efficient collection techniques also make it feasible and attractive to recycle materials like phosphates from manure. Phosphorus abstracted from manure can be used to produce P-fertiliser so less phosphorus to produce fertilisers are imported. This could also contribute to reduce dependency on phosphorus resources mined in a limited number of countries. Moreover, more biogas from manure can be produced as well.

On the other hand there are investment and maintenance costs for more environmentally friendly stables and processing or transfer of manure (PBL, 2010a).

Although this measure is a good practice in the perspective of closing mineral cycles, it should be mentioned that public acceptance could be limited, since this measure affects the typical and culturally valued Dutch landscape, which includes grazing cows. Moreover, it might negatively affect animal health because of wet, more slippery floors which are more prone to bacteria and viruses. This is in conflict with a growing consumer demand for increased animal welfare.

**Process manure**

Manure processing is a way to convert surplus manure into products of higher value and/or products which are easier and cheaper to transport. Different types of processing are available, for instance the liquid manure mixture could be separated into a liquid and a solid fraction or manure combustion can be used for energy production. Separation of liquid manure into a liquid and solid fraction can be done by screening (mechanical process) or sedimentation (gravity process), among other techniques. For combustion, the higher the dry matter content of manure, the more energy can be recovered (dry matter content should be >50 %).

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171 Phosphate rock is in the list of EU Critical Raw Materials, for which supply security is at risk and economic importance is high" and quoting the "Communication on the review of the list of critical raw materials for the EU and the implementation of the Raw Materials Initiative (COM(2014) 297)
The techniques of manure processing are still developing. In North-Brabant, an experiment for the abstraction of nutrients from human urine and pig manure was conducted (Mulder, et al., 2011), the so-called SOURCE project. This system is developed as a treatment system combined with Waste Water Treatment Plants (WWTPs). Urine and manure were combined and purified in an efficient way at a Waste Water Treatment Plant. Residues were used to produce alternative fertiliser. Moreover, energy and heat of the SOURCE production process was exchanged between actors which saved costs and fuel use and therefore GHG as well. The results of the SOURCE experiment were promising, however with the SOURCE technique a maximum of 19 % of the manure surplus can be treated due to the capacity of waste water treatment plants.

A Dutch company that processes pig manure indicates that manure separation allows obtaining water (45 %), nutrient concentrated (liquid) manure (37 %) and organic matter (18 %). If the process is optimised, 80 % of the manure will be clear water, and the concentrated liquid manure will be fertiliser products that can be exported according to the Dutch Fertiliser Act (Eindhovens Dagblad, 2014). Another example is the initiative of regional water board De Dommel to construct a manure processing facility next to the wastewater treatment plant in Tilburg (North-Brabant) (Burgers, 2012).

Therefore, manure processing would be an attractive option to widen the opportunities of manure transfer and the export of fertiliser products. With manure processing, national legislation requirements of receiving countries can be met. Germany, for instance, has strict regulations with respect to poultry manure for health safety reasons. France only accepts manure in the form of compost. Another reason why manure processing is attractive is that it can be converted into product with high nutrient concentrations and exported.

As aforementioned, manure processing is expensive, and the farmer with a manure surplus is responsible for the cost. Most processes require large quantities of manure and are generally not suitable for on-farm implementation, except for separation. On the other hand, the farmer has reduced cost of fertiliser inputs and of electricity. The costs for total manure processing in the Netherlands range from € 20 to € 25 per tonne of liquid manure. For separation only, the costs range from € 2-8 per tonne (PBL, 2010a).

Manure processing leads to a number of environmental benefits. Nutrients of processed manure are more efficiently taken up by crops, so potentially lead to higher yields due to improved fertilisation. As a consequence, N and P leaching and run-off into water sources is reduced with consequent reduction of the environmental impacts. Moreover, the reduction of chemical fertiliser use is associated with lower manufacturing emissions, and the production of electricity reduces the need for fossil fuels and lowers GHG emissions. Less manure stored due to combustion avoids ammonia emissions and particulate matter formation, which reduces human health risks.

Transfer manure

Transfer of manure is already common practice in the Netherlands and also in North-Brabant, which is a net producer of manure. Approximately 45 % of the farmers in North-Brabant have a manure surplus.

Nutrients are redistributed between plots, farms or regions where the production of nutrients in manure exceeds the on-site need for nutrients in crop production or the legal threshold. Note that farmers with a surplus pay for the manure transfer, so applying it on their own land is much cheaper than transferring it to nutrient deficient farmers or regions. By transferring manure and its nutrients from a surplus location to a location in need of nutrients, potential conditions and impacts related to nutrient excess can be limited or reduced, e.g. run-off and leaching.
Manure transfer is a relatively expensive option for intensive livestock farmers compared to the application of manure on fodder crops’ plots since manure producing farmers are usually paying the manure receiving farmers for the application of the manure.

5.5.4.3 Good practices to reduce the nutrient losses in crop production

Drip irrigation systems

Currently, water irrigation and manure application techniques in arable farming are not efficient (PBL, 2012) and due to the sandy soil, arable farming is prone to nutrient leaching and inefficient water use. Therefore, the water use (and simultaneously manure use) in arable farming can be improved along with the improvement of the efficiency of irrigation systems, such as drip irrigation, timing of irrigation and combined fertiliser/irrigation systems (the so-called ‘fertigation’ systems). Efficient irrigation systems (like drip irrigation) require higher initial investments than less efficient irrigation systems like flood irrigation for instance (Tolk, 2013). Obviously, farmers can save on water use as well as on the costs. Moreover, drip irrigation systems are usually location-specific systems and less flexible than others less efficient irrigation systems (Tolk, 2013). Currently, drip irrigation is only implemented for high value (tree-growing) crops, and not for arable crops like vegetables, because it is considered to be too expensive. Due to the high initial investments, the costs of drip irrigation are higher than the costs of less efficient systems. The cost saving of reduced water use is small due to low water prices in the Netherlands. The cost of drinking water is less than €2 per cubic metre, and for irrigation water farmers pay the costs for equipment, pumping (fuel costs) and an abstraction permit (in the case of groundwater).

In North-Brabant, the area used for fruit production is small, while a more significant share of arable land is used for vegetables. However, more controlled (efficient) irrigation systems have a positive effect on nutrient leaching and run-off, and in this respect these systems are interesting for arable crops (vegetables) as these crops are prone to nitrogen leaching and nitrogen run-off (PBL, 2012).

Improve the drainage systems

North-Brabant soil is largely sandy soils, which are prone to low groundwater tables in the case of droughts. This affects arable farmers in North-Brabant, but also livestock farmers with grasslands and fodder crops production. Controlled drainage systems at the plot level can be an interesting measure. With controlled drainage systems farmers have the ability to store water in the soil and control the groundwater table at plot level. The controlled drainage system differs from a regular drainage system in that it is installed deeper underground and that it includes a drainage basin. It requires higher investments because the density of drain pipes is higher, drains are placed deeper in the underground and the system requires a basin to control the groundwater tables and drainage system.

The environmental benefits of controlled drainage systems include:

- Reduced leaching and run-off via an adapted groundwater table;
- Less eutrophication potential resulting in improved conditions for biodiversity;
- Reduction of soil salinity as waterlogged soils prevent leaching of the salts imported by the irrigation water;
- Improved air quality due to less manure/fertiliser use and ammonia emissions, reducing particulate matter formation which poses risks to human health.

The benefits to the farmer are lower costs for water and fertiliser use. Potentially, the average crop yields will be higher because the crop growth is less affected by water shortages or water-logging. The farmer can manage the groundwater table, and as a consequence he/she can arrange good conditions for work in the field using machinery. The annual costs of the system are higher because of the high initial investments.
**Better implement buffer strips near watercourses**

For arable and grassland, buffer strips can be applied next to water courses in North-Brabant. In addition to the already existing fertiliser free zones in river banks designated according to the Dutch Nitrates Action Programme 2014-2017. Buffer strips are advocated in the River Basin management plans of the Water Framework Directive. However, farmers can implement them voluntarily, and in North-Brabant there is a subsidy scheme for the implementation of buffer zones in place. A buffer strip is an area of land with permanent or non-permanent vegetation on the edge of agricultural land and adjacent to surface water. On this land, neither nutrients nor pesticides are used. Buffer strips trap sediments and enhance the filtration of nutrients and pesticides by slowing down run-off from arable land that could enter the local surface waters. Based on current practices, width requirements buffer strips range from 3 to 12 m. North-Brabant has already implemented regulations, but the measure is not common practice yet. Neither nutrients nor pesticides may be used on the buffer strip; depending on the type of crop, different machinery may be required. The environmental benefits of buffer strips are:

- Reduced use and run-off of nutrients and pesticides leading to lower N and P surpluses;
- Increased agro-biodiversity;
- Reduced erosion and contamination of soil (preserved natural resource base of farm due to reduced soil losses).

In the eastern part of North-Brabant, 62 ha of agricultural land have been converted to buffer zones, which led to a reduction of the nutrient load by 3%. Expansion of this measure would increase the reduction in nutrient load.

The benefits of the farmer are the yields of biomass from buffer strips, which can be used as fodder or a renewable energy input. The disadvantages include costs of planting vegetation of the buffer strip, management of the buffer strip (getting rid of unwanted vegetation), and loss of income for taking land out of production. In contrast to manure-free zones, the manure or fertiliser not applied on the area of the buffer strip can be applied on other parts of the farmer’s land.
5.6 Southern and Eastern Ireland (IE)\textsuperscript{172}

Box 9 – Southern and Eastern Ireland case study - In brief

Southern and Eastern Ireland is affected by eutrophication due to overload of nitrogen and phosphorus, potentially causing health effects and by ammonia emissions to air causing acidification and health issues. Algal bloom, particularly in coastal areas, may negatively affect biodiversity, health and economic activities such as tourism and fisheries. While only a small share of freshwaters in the South-East are classified as eutrophic, three-quarters of the waterbodies are considered to be at risk from eutrophication. The region features a high number of large farms and high density of livestock. Overall, more than half of the surface water bodies are suffering pressures from diffuse sources, including run-off from agriculture caused by tillage activities and nutrient leaching from farmyards, manure stores and fields. Southern and Eastern Ireland is characterised by intensive agriculture accounting for 98\% of the national ammonia emissions. In particular, emissions from manure excreted by livestock in Ireland represent 86\% of the ammonia emissions followed by emissions from fertiliser application. Most of the emissions and nutrient losses arise from the intensive livestock breeding spending a long time outdoors that leads to high quantity of manure deposited on pasture and high quantity of manure and inorganic fertiliser applied on fields. Inappropriate application techniques and tillage and ploughing also have an effect on nutrient losses. The effects of the practices are enhanced by natural factors such as abundant rainfall, the geology and soil structure and to a lesser extent soil erosion.

With a total Gross Value Added (GVA) of € 132.3 billion in 2013, the region contributed 82\% of the country's GVA. In addition, the Food Harvest 2020 programme is aiming at increasing the value of primary output in the agriculture, fisheries and forestry sectors by € 1.5 billion. This represents a 33\% increase compared to the 2007-2009 average. Agricultural activities are responsible for 31\% of the total number of polluted river sites. The pollution of rivers and coastal waters with nutrients lead to eutrophication and algal blooms. These blooms are detrimental to tourism, fisheries, aquaculture and shellfisheries. Nitrate concentrations in rivers in the Southern and Eastern region are generally below 11.3 mg nitrogen per litre (equivalent to 50 mg NO3-/L). However during winter many rivers were found to exceed the 25 mg/L NO3- guide level set by the EU Directive on Drinking Water. Irish agricultural outputs were worth € 7.1 billion and its inputs cost € 5.5 billion in 2013. By increasing the efficiency of agricultural inputs on-farm by 1\%, global savings of € 55 million could be achieved.

For Southern and Eastern Ireland, measures have been selected to address the causes of nutrient losses, which mainly relate to the agricultural pressure related to the high livestock density. The measures also take into account the fact that in Ireland and in the Southern and Eastern region, the predominant manure system is slurry based, followed by farmyard manure system. The first set of measures focuses on the reduction of the source of pollution through the conversion of agricultural land to organic farming, the transfer of manure to other farms, the improvement of preparation of the fertilisation management plan. The use of digestate from anaerobic digestion can also help to ease the transfer and use of manure while also valorising the high amount of manure produced in the region through the production of energy. Another type of good practice relates to the reduction of the amount of nutrients lost during housing and storage by improving manure collection and covering the slurry tanks. The third set of measures concerned to the prevention of nutrient losses during manure application on field or on pasture. The measures relate to the lengthening of the grazing season or the adoption of rotational grazing, the choice of the best suited application techniques, the incorporation of manure into the soil, spreading fertilisers at the right time and stage of crop growth, the use of adequate tillage technique and the consideration of climatic and geographic conditions. It is important to highlight that some of these measures are already being successfully implemented in some farms and that further improvement will depend on the sharing of these ‘success stories’. The role of knowledge transfer in Southern and Eastern Ireland is considered very important based on discussions with stakeholders at the Regional Workshop held in Portlaoise on 28 October 2014.

\textsuperscript{172} Ms. Victoria Cherrier, AMEC (lead author); Mr. Ben Grebot, AMEC; Mr. Giorgio Provolo, Milan University; Ms. Marion Sarteel, BIO by Deloitte; Dr. Helen Ding, BIO by Deloitte.
Southern and Eastern Ireland is a NUTS 2 classification region of Ireland (Figure 87). The region covers an area of around 36,544 km², accounting for 52.3% of the national territory in 2014 (Eurostat, 2014f). The economy of the region is based primarily on agriculture, manufacturing and services, tourism, fishing and aquaculture. The agriculture and food sectors account for a significant proportion of output and employment in the Southern and Eastern Region.

The NUTS 2 region spreads across four different river basins districts that do not match exactly the boundaries of the region. The River Basin Districts (RBD) are: the Eastern river basin (IEEA), the South-Eastern river basin (IESE), the South-Western river basin (IESW) and the Shannon river basin (IEGBNISH) (Figure 88). The South-Western RBD is fully encompassed in the NUTS2 region, but only the southern parts of the South-Eastern, Shannon and the Eastern RBDs are included. All of the RBDs are characterised by a high number of small water bodies, with an average length overall of 5 km. Similarly only 2% of the lakes (209 out of 12,206) are greater than 0.5 km² (European Commission, 2012e). Ireland has adopted the “whole territory approach” under the Nitrates Directive, with obligatory measures in place in its whole territory.
Irish soils are generally rich in soil organic matter, in particular in the wetter parts of the country. However, a range of soils are present in the Southern and Eastern Ireland region which includes some of the best drained soils in Ireland. Variations can be observed in the organic content of soils due in part to the influence of the climate. The organic content of Irish soils has been decreasing due to practices such as tillage activities (Environmental Protection Agency, 2008). Tillage at a national level occupies a small proportion of the utilisable agricultural area (UAA) (9-10 %) and is mostly practiced on dryer soil types with lower organic matter content. On average, Irish tillage soils have an organic matter content above the 3.4 % threshold, which had been set under the GAEC (cross-compliance rules).

The national Soil Organic Carbon (SOC) stock has been estimated at 2.4 Gt which compares with the current global estimate of SOC of 1 500 to 1 550 Gt (Environmental Protection Agency, 2008). However, most of the SOC is contained in the peat soils (62 %) and the remaining 38 % SOC is spread between the other soils (Figure 89).

The SOC content is commonly considered as low, below 2 % (3.5 % or Soil Organic Matter (SOM) content), possibly resulting in decline of soil structural stability and disruption of the nutrient cycles (Eckelmann, et al., 2006; BIO by Deloitte, 2014).

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173 The RBD coloured in pink are international RBD shared between several countries, here Southern and Eastern Ireland

174 Note the South-East Ireland region is not the same as the Southern and Eastern NUTS 2 region. Rather the South-East Ireland region (referred here as “South-East”) is an area within the Southern and Eastern NUTS 2 region, covering the counties of Carlow, Kilkenny, Laois, Tipperary, Waterford and Wexford, and Waterford city. The region includes parts of three RBD: the South-Eastern RBD, the Eastern RBD and the South-Western RBD. See the available map in (Environmental Protection Agency, 2013a).
The Southern and Eastern region accounted for 55.6% of the total area farmed in Ireland with 2.5 million hectares out of 4.5 million hectares in 2013 (Eurostat, 2015a). Livestock production is by far the main type of farming in the region. In 2014, 62% of the livestock in Ireland was kept in the region: there were over 3.9 million cattle in the Southern and Eastern region (compared to 2.5 million cattle in the Border, Midland and Western region), of which 900,000 are dairy cattle (Eurostat, 2015g). Specialist beef production represented more than half of the holdings in 2010 (Central Statistics Office Ireland, 2012). The second main production is dairy cattle, with nearly 81% of Ireland’s milk being produced in the region in 2012 (Central Statistics Office Ireland, 2014a).

Arable land accounted for 29% of the total agricultural land in the region in 2014 (Eurostat, 2015a). While around only 16% of the holdings produce crops (Eurostat, 2015a), the region accounted for almost 80% of the national cereal production in 2012 (CSO, 2014c). However, only 6% of the holdings are specialist tillage. The main crop production is by far barley, then wheat and maize silage (Central Statistics Office Ireland, 2012). Within the region, variations can be observed in farms, for example in the south the average size of the farms are 53.9 ha UAA and 73.3 livestock units for grazing livestock. In comparison, the south-east farms are smaller with an average size of 48 ha UAA and 64.2 livestock units for grazing livestock in 2012 (Teagasc, 2012). By comparison, at national level, the average size of farms are 46.2 ha UAA and 60.3 livestock units for grazing livestock (Teagasc, 2012).

175 In 2007, the national Irish livestock density was 1.4 LSU/ha UAA according to Eurostat.
Whilst there are fewer farms in the Southern and Eastern region than in the rest of Ireland, the farms themselves are larger and keep a higher number of livestock. Southern and Eastern Ireland is also characterised by the high density of its agriculture with over 1.7 LSU per ha UAA (Eurostat, 2013a).

5.6.1 Notable impacts of nutrient surplus

This section focuses on the nutrient loss impacts that are specifically related to agricultural practices in Southern and Eastern Ireland. In this region, the main impacts are caused by nitrogen and phosphorus surpluses that mainly affect water and air. Note that nutrient losses to water, soil and air do not only result from agricultural production but also from other sources such as sewage. Potassium excess is not a significant issue in this region.

In 2012, the regional nitrogen budget was 145 kg N/ha and the phosphorus budget was 5.6 kg P/ha (Buckley, 2014). According to a study performed by Teagasc, the evolution of nitrogen budget at a farm
reduced by 14% compared to 2006, which is mainly due to reduced imports of synthetic nitrogen fertilisers. This represents for farmers an average savings of €1,423 per annum (Buckley, 2014). Over the same period, the phosphorus balance has reduced from 11.7 kg/ha to 5.6 kg/ha, which represents an average saving of €897 per annum. These reductions have been reached through an improvement in phosphorus use efficiency from 60.7 to 79.6%.

**Impacts of nitrogen losses**

**Nitrogen load in freshwater**

Southern and Eastern Ireland has a high nitrogen leaching and run-off ranging between 20 and 50 kg N/hectare (Velthof, et al., 2014). The Irish Environmental Protection Agency assesses the quality of freshwater and marine waters in Ireland which includes measuring levels of nitrates and phosphates in groundwater, rivers and lakes. Overall levels of nitrates in water (in groundwater, rivers and lakes) are higher in the South-East than in the rest of Ireland (Environmental Protection Agency, 2013a).

**Surface water** – Nitrate concentrations in rivers in the Southern and Eastern region are generally below 11.3 mg N/L (equivalent to 50 mg NO3-/L). The South-East has the highest river nitrate levels within Ireland. In 2012, 59% of the rivers in the region had a concentration above 25 mg/L NO3-, compared with 58% in 2011. In particular, many rivers were found to exceed this concentration during winter. This is explained in part by rainfall during winter which is likely to increase nitrogen leaching. Furthermore in 2012, 30% of rivers had a concentration of ≤ 1.8 mg/L nitrate as N (Environmental Protection Agency, 2013a).

Ammonia in rivers can occur naturally from the microbiological decomposition of nitrogen compounds in the organic matter. In addition, ammonia is naturally excreted by fish and other aquatic organisms. As a result, it is not uncommon to detect ammonia concentrations in rivers, usually inferior to 0.02 mg/L of ammonia as N. The limit for good status is 0.065 mg/L N (Environmental Protection Agency, 2013a). Ammonia is toxic to fish even in very low concentrations; however ammonia in rivers is present as ionised ammonium (NH4+) which is less toxic. In 2012, the average concentrations of NH4+ in freshwaters ranged from 0.01 to 0.944 mg/L as N in the region. However, in 2012, 16% of the monitoring stations have measured ammonium concentrations above 0.065 mg/L N which is comparable to 2011 monitoring results where 15% of the monitoring stations reported concentrations above the limit for good status.

From information in the river basin management plan for South-Eastern Ireland basin district, 53% of the rivers and 38% of the lakes did not reach the good ecological and chemical status (South-Eastern River Basin District, 2010). In 2012, there were 900 river sites with a less than good status in all of Ireland with 100 of these in the South-East region. For 61 of these river sites, agriculture was identified as the principle source for not achieving good status as the intensive land use increased the risk of nutrient loss and enrichment of the water potentially leading to eutrophic conditions in standing water bodies (Environmental Protection Agency, 2013a). Furthermore, 72% of the rivers in the South-Eastern RBD, 26% of the rivers in the South-Western RBD and 50% of the rivers in the Shannon RBD were at

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176 The farm gate approach restricts analysis to imports and exports of nutrients over which the farmer has direct control (through the farm gate)

177 1.8 mg/L N is a surrogate standard used by the Irish EPA as indicating good ecological status for rivers, since it is considered that the Drinking Water Directive’s Guide Level of 5.65 mg/L N is not stringent enough for ecological purposes. The surrogate standards have been derived from measurements of average nitrate concentrations at over 1,500 high and good ecological status sites (Environmental Protection Agency, 2013a).
risk of not meeting the good status requirement because of pressures from agriculture (South-Eastern River Basin District, 2010).

The report on the implementation of the Nitrates Directive found that 12 % of the lakes were eutrophic, this has remained stable from the previous reporting period (European Commission, 2013b).

Excessive nutrient load from agriculture and other sources and subsequent algal blooms result, in some instances, in a reduction of output capacity or even temporary closure of drinking water treatment plants. For instance, Irish Water reported a cut-off of water supply to wider Greystones area during night time (from 8 p.m. to 6:30 a.m.) when an algal bloom affected the Vartry water treatment plant at Roundwood (Irish Times, 2014). In particular, algal blooms blocked filters and reduced capacity from 65 million litres per day to 40 million litres per day. A research project is on going to determine the causes, the impacts and the measures to deal with the algal proliferation in this reservoir (end of the project: 2018) (Environmental Protection Agency, 2015).

For Ireland, the latest report on implementation of the Nitrates Directive found that no freshwater stations exceeded 50 mg nitrate per litre during the 2008-2011 period. In addition, only 0.7 % of the freshwater stations exceeded the concentration of 25 mg nitrate per litre during the same period. For the majority of the surface waters (61 %) the concentration of nitrate remained stable during the 2008-2011 period. Feedback from the regional conference178 highlight that this does not necessarily mean that no improvement has been made during the period, but rather that a ‘lag time’ can be observed between the introduction of a management practice and the observation of improvements from the practice due to the length and complexity of some water cycles (European Commission, 2013b).

Groundwater—For Ireland, the latest report on the implementation of the Nitrates Directive (2008-2011) found that no groundwater monitoring station exceeded the concentration of 50 mg nitrates per litre and around 10 % fewer monitoring stations reported groundwater exceeding 25 mg of nitrates than during the previous reporting period (2004-2007). Overall during the 2008-2011 period, 74 % of the groundwater showed decreasing concentrations of nitrates, 21 % were stable and for 5 % the concentration of nitrates increased (European Commission, 2013b). The national 2012 monitoring indicated that nitrate concentrations remained highest in the South and South-East179 of Ireland in the period 2007 to 2012 (Environmental Protection Agency, 2013a). In the South-Eastern RBD 97 % of groundwater resources have ecological and chemical good status (South-Eastern River Basin District, 2010). Overall, nitrates in groundwater are not considered to be a notable issue for Southern and Eastern Ireland region.

An important factor to consider, with regards to nitrate levels in groundwater bodies, is the lag time and that the legacy of past management may still affect slow moving aquifers in the present day. Therefore, it is difficult to disentangle current management practices and their impact from past ones. The stakeholder consultation of this project suggest that stabilisation of the levels could indicate that there is no further deterioration due to past management (i.e. legacy) or it could indicate that a new balance in nitrate levels has been reached and is being maintained with current management.

178 As part of the dissemination part of the project regional conferences were held in selected regions. One was organised in Portlaoise, Ireland in October 2014.

179 As mentioned above, note the South-East Ireland region is not the same as the Southern and Eastern NUTS 2 region.


**Eutrophication in marine waters**

Point sources of pollution from wastewater treatment plants and diffuse sources from agriculture are significant contributors to nutrient loadings, and the subsequent eutrophication of transitional and coastal water bodies. In the south-east, eutrophication is higher than the national average as 76% of the estuaries and 33% of the coastal waters do not have a ‘good’ status in 2008 (i.e. moderate, poor or bad) (South-Eastern River Basin District, 2010).

While phosphorus is likely to be the limiting nutrient for plant growth in freshwater, nitrogen is considered limiting in saline waters (Environmental Protection Agency, 2013a). The Slaney and Barrow/Nore/Suir estuaries are located in the South-East (Figure 88) and present some of the highest winter nitrogen values nationally. In 2011, the load represented 24 300 tonnes of N, which is 25% of the national load into transitional and coastal waters (Environmental Protection Agency, 2013a).

The Irish Marine Institute monitors algal blooms off the coast of Ireland through its Phytoplankton Monitoring Programme. Algal blooms, but also foam associated with the blooms, pose no direct threat to humans, though they can lead to extreme anoxia (lack of oxygen), causing mortality in marine organisms. It has been observed that wild fish tend to avoid algal blooms which can threaten fisheries. In addition, algae can pose a risk through consumption of exposed shellfish. However, the quality of shellfish waters is generally considered as very good, with 37% of shellfish production areas in Ireland classified as class A or seasonal class A and 54% as class B in 2009 (Environmental Protection Agency, 2010).

**Ammonia (NH₃) emissions to air**

In 2013, a total of 107 800 tonnes of ammonia were emitted at national level (23.1 kg NH₃/ha UAA\(^{180}\)). Agriculture is responsible for almost all of these emissions (>98%).

It is estimated that the animal manures produce 87% of the ammonia emissions in agriculture, with chemical fertilisers accounting for the remainder. The cattle sector is the main contributor, responsible for 74% of the total ammonia emissions in 2010 (Mounsey, 2012). Most of the emissions arise from the livestock housing and the application of manure (respectively 54% and 24% of the total NH₃ emissions). This is followed by emissions from inorganic fertiliser application (7% of total emissions), pig rearing (6.3%), and sheep and poultry rearing (each around 2%). Whilst there is no specific estimate for the Southern and Eastern Region, the EPA estimated that approximately 17% of the nitrogen contained in animal wastes and 2% of nitrogen contained in chemical fertilisers is lost to the atmosphere as NH₃ (Environmental Protection Agency, 2014).

**Nitrous oxide (N₂O) emissions to air**

In 2012, the Environmental Protection Agency (EPA) estimated that the contribution of agriculture to GHG emissions had increased by 3.1% between 2011 and 2012. This increase is due to a 4.4% increase in the number of cattle and a 9% increase in the number of sheep. At the national level, the agriculture sector is responsible for 32% of the total GHG emissions (58.5 Mt CO₂eq.), and N₂O emissions account for one third of the GHG emissions from agriculture. In 2007, Ireland had the third highest contribution of agricultural N₂O (9.9%) to the total national GHG emissions compared to the EU average (5.9%).

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\(^{180}\) Considering that 104 080 t NH₃ are due to agriculture.
Impacts of phosphorus losses

Excessive phosphorus load in freshwater

Surface water – River water quality monitoring has shown increased eutrophication in most Irish rivers since the 1970s. It is interesting to note that until mid-1990 the use of phosphorus fertilisers increased, this has since been reduced due to the inclusion of phosphorus in the national Nitrates Regulations (Environmental Protection Agency, 2013a). Agriculture is not the only source of phosphorus leaching to freshwater as municipal and industrial effluent discharges are also important contributors.

Phosphorus concentrations of >0.2 to <0.5 mg PO$_4^{3-}$/L are found in the rivers of the South-Eastern, the Eastern and the Southern part of the Shannon RBDS (Lynche-Solheim, et al., 2010). The Irish Environmental Protection Agency found that the orthophosphate levels in rivers have been stable in the recent years. In the South-East in particular, 2012 saw an improvement in the levels, with an average annual orthophosphate in 36 % of river sites exceeding the threshold for good status, against 45 % in 2011 (Environmental Protection Agency, 2013a). These levels are still higher in the South-East than for the rest of Ireland and especially in the areas of land under tillage (Carlow, Kilkenny, Laois, Tipperary, Waterford, Wexford) (Environmental Protection Agency, 2013a).

A 2013 review of the suspected causes of pollution from phosphorus and nitrogen for the Southern and Eastern river basin district found that 42.7 % of suspected pressures were due to agriculture, followed by urban wastewater 18.8 %, mixed rural influences 17.6 %, urbanisation 7.8 % and industrial activity 6.7 % (Environmental Protection Agency, 2013b).

Groundwater – In the river basin management plan adopted within the Water Framework Directive, Ireland has used the threshold value concentration of 0.035 mg P/L when assessing the contribution of orthophosphate in groundwater to rivers. The proportion of monitoring locations with an average orthophosphate concentration >0.035 mg P/L was 5.3 % in 2012 (Environmental Protection Agency, 2013a). Overall in comparison to 2001 levels, the average orthophosphate concentration in 2012 had decreased slightly, which may be a reflection of the reduction of the use of phosphorus in agriculture. However, the proportion of monitoring locations with average concentrations >0.015 mg P/L has increased since 2009, with one monitoring location (in Wexford county) having an average concentration of >0.050 mg P/L in 2012 (Environmental Protection Agency, 2013a).

Coastal waters: Over 1000 km$^2$ of transitional and coastal waters in 16 water bodies were monitored in the South-East in 2012. Of the 16 estuarine and coastal water bodies in the south east, one was classed as eutrophic, three as potentially eutrophic, eight as intermediate and four unpolluted. The non-polluted water bodies represent 25 % of the river bodies in this river basin, which compares unfavourable with the national average of 57 % of water bodies non-polluted. A report from the Sea Lettuce Task Force found that 10 000 tonnes of sea lettuce accumulates each year on the beaches located on the Cork County coast (Sea lettuce task force, 2010).

In addition, occasional toxic algal blooms occur in some inland water bodies. Thus, warning signs due to algal blooms have been placed at Ballyalla Lake and Lough Graney while frequent algal blooms on Lough Derg have made one of the country’s top pleasure lakes off limits to bathers for up to a month each year. Bathers seem to be complying with the warning signs particularly following the recent deaths of two dogs after drinking water on the North Tipperary shoreline (The Independent, 2005). Recently, the public has been warned not to drink from, fish or paddle in Lough Leane, the biggest of the Killarney lakes, after the emergence of algal scum at a number of locations (The Irish Times, 2015).
There has been no improvement in the quality of these waters since the previous assessment conducted in 2011 and it is noteworthy that the five main rivers\(^{181}\) in the South-East contribute to 25\% of the pollution from nitrogen and phosphorus to the coastal waters (Environmental Protection Agency, 2013a).

### 5.6.2 Causes of nutrient losses

In Southern and Eastern Ireland, two main groups of causes for nutrient losses have been identified. The first one relates to farming systems and practices (e.g. use of inorganic fertilisers), which are under the control of the farmer or land manager. The second group relates to other factors such as climate and soil conditions that cannot be directly managed but rather must be taken into account.

**Farming system and agricultural practices**

*Intensive dairy and beef farming with high manure production*— The Southern and Eastern region is the most intensive agricultural area of Ireland. Irish dairy and beef sectors are very important for the national economy. Exports of Irish dairy products were valued at €2.7 billion in 2011 and total milk production was 5.4 billion litres in 2011. With the end of the EU milk quota in 2015 and in line with the Food Harvest 2020, it is expected that the production of dairy products will continue to grow.

In the Southern and Eastern region there are less livestock holdings than in the other NUTS 2 regions of Ireland (Border, Midland and Western). However, holdings in the Southern and Eastern region include nearly 40\% more head of cattle than in the other regions (Table 48). Thus, it can be perceived as important sources of pressures on the environment (Buckley, et al., 2013).

| Table 47 – Comparison of holdings and total head of cattle in Ireland NUTS 2 regions (Eurostat 2014) |
|--------------------------------------------------|--------------------------------------------------|
| **Number of holdings** | **Cattle (number of heads)** |
| Border, Midland and Western | 68,390 | 2,551,900 |
| Southern and Eastern | 58,740 | 4,054,680 |

In the Southern and Eastern region, dairy farms tend to have the highest stocking densities and fertiliser inputs of grassland systems in Ireland (Teagasc, 2012). The nitrogen use efficiency has been reviewed for different types of farming. In 2012, it varied from 14.9\% for cattle rearing to 56\% for specialist tillage farms. For livestock farms, the most efficient use of nitrogen are from sheep rearing (nearly 36\%). Larger inputs of nitrogen are for dairy farms (155.8 kg N/ha) compared to beef farms (47.5 kg N/ha). Similar trends were observed for phosphorus\(^{182}\) (Buckley, 2014).

Thus, the dairy farming industry exerts a significant influence on nutrient balance and dairy farms are spread throughout the region. In the region, the intensive production of manure is one of the main sources of surplus nutrients. Slurry distribution across the farm landscape and its utilisation in conjunction to chemical fertilisers may potentially cause nutrient imbalances. The main imports of nutrients (both nitrogen and phosphorus) onto dairy farms are chemical fertilisers (during growing season) and concentrate feedstuffs (mainly during winter indoor season)\(^{183}\). Because of the high levels of agricultural activities in the region, the water bodies are particularly vulnerable to nutrient enrichment – from phosphates and nitrates in particular. A decline in the cattle population was observed after 1998.

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\(^{181}\) The Barrow, the Blackwater, the Nore, the Slaney and Suir

\(^{182}\) It is important to note that Ireland is one of the few Member States that has included measures for phosphorus in its implementation of the Nitrates Directive.

\(^{183}\) Expert review
as well as a decrease in fertiliser use, which contributed to a downturn in ammonia emissions in the period 2000 to 2011. However, the EPA notes that recent increases in cattle numbers and fertiliser uses have seen ammonia emissions increase in 2013 by 1.9kt. This is linked to a 19 % increase in synthetic fertiliser use in the same year (Environmental Protection Agency, 2014).

The proposed expansion of the agriculture sector, as set out in national policy entitled Food Harvest 2020\textsuperscript{184,185}, is a significant economic opportunity for the country and will bring large increases in farm outputs over the coming years. The majority of the planned expansion of dairy farming will be focused in the east and south of the country. This means that the existing pressures from agriculture are likely to be further increased and need to be managed. An assessment of the potential environmental impact of the increased food production envisaged by the Food Harvest 2020 was published in 2014. The assessment found that the changes envisaged under the Food Harvest 2020 would lead, before mitigation, to “a slightly negative impact in the environmental characteristics biodiversity, flora and fauna; water quality; air quality and climatic factors and to a neutral/imperceptible impact on soils and landscape”. Furthermore, the report concludes that the use of technology and best production methodologies can deliver enhanced environmental outcomes and could potentially, if implemented in addition to mitigation measures, reverse negative impacts on environment. However, the report identified a knowledge and skills deficit at both farm advisor and primary producer level and recommended the development of a skill programme for 570 approved farm advisors (Farrelly, 2014).

Livestock spending a long time outdoors leading to high quantities of manure being deposited directly into the soil in pasture – The livestock production, especially for cattle, in the Southern and Eastern region is a grass based system. It is characterised by the longest grazing period of the country with more than 300 days (Figure 92). Among the success factors of the Irish farming sector there is the availability of cheaper feed from outdoor grazing. For comparison, the cost of grass is much lower than the costs of buying feeds or concentrates (€ 80 per tonne of dry matter for grass versus € 265 per tonne of concentrates feed).

\textsuperscript{184} The Food Harvest 2020 is a roadmap adopted by the Irish government aiming at developing Ireland’s potential in providing food resources. It is based on the following objectives: the Irish agri-food and fisheries industry must put smart thinking at the core of both its strategy and action. Irish companies must seek new markets, develop new product streams that meet changing consumer demand, as well as finding new ways to assert Ireland’s environmentally friendly credentials to target the premium end of the market with high-value products.

\textsuperscript{185} Note that Food Harvest 2020 has been replaced by the Food Wise 2025 programme since July 2015.
Research carried out under the National Development Plan\textsuperscript{186} found that grazed grassland had a higher nitrate-removal capacity through denitrification in groundwater, compared to areas under tillage farming (Jahangir, 2010). A higher percentage of denitrification in the tillage site resulted in N\textsubscript{2}O instead of N\textsubscript{2}, which could be indirectly contributing to greenhouse gas emissions upon discharge to surface water (Jahangir, 2010). The proportion of time spent outdoors by animals is a cause of nutrient losses because manure deposited outdoors cannot be collected as easily as manure from housing. Furthermore, it requires greater levels of management in order to avoid atmospheric pollution.

**Impact of tillage and ploughing on nutrient leaching** – The Southern and Eastern region also has the most intensive agriculture practices, including tillage. Tillage at a national level occupies a small proportion of the utilisable agricultural area (UAA\textsuperscript{187}) (9-10 %) and is mostly practiced on dryer soil types with lower organic matter content. The organic content of Irish soils has been decreasing due to practices such as tillage activities (Environmental Protection Agency, 2008). On average, Irish tillage soils have an organic matter content above the 3.4 % threshold, which was set under the GAEC until 2014 (cross-compliance rules). This is also reflected in the higher-than-national average nitrate concentrations seen in the South-East region rivers (Environmental Protection Agency, 2013a). In areas

\textsuperscript{186} Funded by Ireland’s Department of Agriculture and Food (DAFM)

\textsuperscript{187} Utilisable agricultural area is defined in Ireland as the area under crops and pasture plus the area (unadjusted) of rough grazing. It is the total area owned, plus area rented, minus area let, minus area under remainder of farm. Source: http://www.agriculture.gov.ie/publications/2008/compendiumofirishagriculturalstatistics2008/notes/
under tillage, also depending on the timing, ploughing of agricultural land is a factor affecting nitrate concentrations in rivers since tillage increases the soil drainage. Ploughing is restricted in Southern and Eastern Ireland, and more broadly in Ireland, from July until December as part of the implementation of the Nitrates Directive. It is deemed that the temperature of the soil in December and January are low enough so that mineralisation of nitrogen during these months would remain minimal. The highest annual river nitrate concentrations in areas with high tillage/ploughing normally occur in the months of January/February and the lowest concentrations in July/August (Environmental Protection Agency, 2013a).

In addition, it was identified that land drainage work can lead to silt leaching to the rivers, carrying N and P nutrients with it (University College Dublin, 2014). The Environmental Impact Assessment (Agriculture) regulations (SI 456 2011) sets thresholds for drainage. The Good Agricultural and Environmental Conditions and recently introduced measures related to the implementation of the Nitrates Directive also contribute to reducing (silt) leaching\textsuperscript{188}.

\textbf{Inappropriate application techniques for manure} – In Ireland and in the Southern and Eastern region, the predominant manure system is slurry based, followed by farmyard manure system. A national survey of manure management practices found that 87 % of all farms produced and spread slurry and/or farmyard manure in 2009 (Hennessy, et al., 2011).

Slurry is applied by tanker and splash plate on 97 % of all Irish farms (Hennessy, et al., 2011), a method which does not distribute slurry evenly over land and often results in over application of slurry (Ryan, 2005). It also leads to higher volatilisation of nitrogen than other application techniques. For farmyard manure, side discharge is the most common method and is used by 57 % of the farmers. Rear discharge is the next most common method and is used on 27 % of the farms. This application technique leads to more volatilisation and nutrient losses due to uneven spreading.

\textbf{High amount of fertilisers used (inorganic and organic) for crop production} – According to modelled data, the Southern and Eastern Ireland region is characterised by a very high phosphorus fertiliser application (Figure 94), and high nitrogen fertiliser application (Figure 93) (Grizetti, et al., 2007).

\textsuperscript{188} Expert review
The data collected for the South and South-East Ireland on a survey conducted between 2000 and 2003 on the use of fertilisers found that fertiliser uses were higher than average amounts. In addition, dairy farms were found to have the highest usage of fertiliser, cattle farms the lowest for both N and K, and sheep farms had the lowest P usage (FAO, 2007b; Coulter, 2005). Recent national research has found that there is tendency to over-apply both chemical and organic fertilisers to land, in particular on specialist dairy and tillage farms (Buckley, 2010). It also shows that on a proportion of dairy farms there was a positive farm gate balance of phosphorus; however, this does not account for the requirements to build-up phosphorus in the individual fields on a farm. Nutrient management efficiency across specialist dairy and tillage farms was investigated as part of a national farm survey. Results indicated that, compared to the most efficient benchmark farms, the average over application of chemical fertilisers ranged from 22.8 to 32.8 kg N/ha and 2.9 to 3.51 kg P/ha (Environmental Protection Agency, 2012).

Information available suggests that, since 2007, chemical N fertiliser sales continued to fall while chemical P sales continued to rise in Ireland (Environmental Protection Agency, 2012). The decline in the use of inorganic fertilisers is explained by an increase in the price rather than an optimisation of their use (Lalor, 2010a). It was stated that a reduction in the price of inorganic fertilisers could increase the consumption again\(^{189}\). This was observed between 2012 and 2013, the Environmental Protection Agency identified that national emissions of ammonia increased by 1.9 kt, primarily due to a 19 % increase in inorganic fertiliser use.

The over-application of fertiliser significantly impacts nutrient losses and water quality. The review of a survey on fertiliser use, based on nationally stratified survey data from 2004 to 2008, found a reduction in the use of fertilisers during the period considered (Lalor, et al., 2010), especially for phosphorus. In particular, the review found a consequent and steady decrease in the usage of fertiliser consumption at national level with an average level of fertiliser applied to grassland in 2008 of 86 kg/ha of nitrogen and

\(^{189}\) Expert review

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[189] Expert review
5 kg/ha of phosphorus. It is important to note that the decreasing use of fertilisers could be explained in part by an increase in their prices in 2008-2009 (Lalor, 2010a).

The application of manure is spread throughout the year during the open season. The national territory is sub-divided into three zones, taking into account soil type, rainfall and length of the growing season. The open seasons are different for each zone. The Southern and Eastern region falls mainly in Zone A and B where spreading is prohibited respectively between 15 October to 12 January and 15 October to 15 January for all organic fertilisers excluding farmyard manure. For farmyard manure, spreading is prohibited between 1 November to 12 January (zone A) and 1 November to 15 January (zone B) (Teagasc, 2011b). However there is a tendency for spreading slurry in spring and the farmyard manure in autumn. The spring application of slurry leads to higher use efficiency of the nitrogen and farmyard manures take longer for incorporating in the soil. In addition, farmyard manures may not be suitable for spreading on grassland that will be used for grazing early in the year due to potential grass contamination (Mounsey, 2012). At a national level, a scheme has been established in order to provide grant aid for farmers to invest in low emissions equipment. There is no information available on the uptake of this scheme, however it was expected to be modest as the aid was quite restricted and the equipment required is costly. It is important to note that in the Southern and Eastern region, low emissions application techniques are mostly supplied by contractors\(^{190}\).

In recent years the closed period for slurry spreading has been modified, shifting the dates in order to accommodate adverse weather conditions. As the storage capacities for slurry were not sufficient for the added slurry collected from housed livestock, it has been applied to the land whilst the crops were not requiring fertilisation.

The use of fertilisers also has an impact on N\(_2\)O emissions as they arise naturally from the soil and from the application of nitrogen based fertilisers. The nitrogen contained in the manure is also a source of N\(_2\)O emissions (Teagasc, 2005).\(^{191}\)

**Environmental conditions**

Natural factors modulate the effects of farming practices and especially fertiliser inputs.

**Abundant rainfall** — The figure below maps the average annual rainfall over a 30-year period, and highlights the west as the wettest part of Ireland.

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\(^{190}\) Expert review

\(^{191}\) Teagasc is currently conducting research in order to develop more accurate emission factors for N fertilisers. Currently, Ireland use a default value of 1 % as stated in the IPCC.
The high number of wet days, combined with the seasonality of rain, contributes to nutrient run-off and leaching from seasonally water-saturated soil. In the Southern and Eastern region, there is a large variability in mean rainfall from south-west (>1 800 mm per year) to north-east (>700 mm per year) across the region. The average annual number of wet days (days with more than 1mm of rain) ranges from about 150 days per year along the east and south-east coasts, to about 225 days per year in parts of the west (Met Eireann, 2014).

The combination of frequent and abundant rain with free-draining soils which are under tillage and ploughed regularly, leads to an increase in nutrient run-off and leaching from agricultural land (both N and P). Rainfall occurring shortly after the application of manure also leads to increased nutrient run-off.

**Nutrient transfers increased by geology and soil structure** – Soil structure and geology play a significant role in nutrient transport and storage. In the south-western part of the country, soils are often waterlogged (gleysols) due to the slow movement of water there. Most of the central and eastern part of Ireland is covered by ‘brown earth’ (cambisols) which is a shallow layer of fertile soil. At a national level 64 % of the cambisols are used for agriculture (4.4 million hectares) (Ask about Ireland, 2014). In the west and south-west parts of Ireland mainly podzols dominate which are generally poorer soils for agronomic purposes with high lime and fertiliser requirements (FAO, 2007b) (Figure 96).
Since the dominant agricultural use of land in the region is dairy farming, which has the highest usage of fertiliser relative to other agricultural sub-sectors, this combines with the geology of the area to influence nutrient loading (Coulter, 2005).

**Phosphorus losses through erosion / sedimentation** – Phosphorus attached to eroded sediment is transferred from agricultural land to water bodies (Daniel, 2002). Whilst soil erosion is not a major issue in Southern and Eastern Ireland, erosion and sedimentation are increased by land drainage and tillage near watercourses. They are also increased by livestock access to stream and rivers banks, causing bank erosion (SWAN, 2013). The predominance of grassland farming in the Southern and Eastern region, with both dairy and beef cattle spending a large proportion of their life outdoors, may contribute to increased sedimentation from river banks in this manner. However, it is important to note that most paddocks on dairy farms are fenced with posts and wire, giving livestock a limited access to streams.\(^{192}\)

In addition, much of the phosphorus added to soil in the form of agricultural fertiliser or animal slurry tends to accumulate in the top layers of soil and the surface soil layer can over time become saturated with P thus increasing the risk of run-off.

### 5.6.3 Costs of the environmental and health effects

#### 5.6.3.1 Socio-economic description of the region

The region is relatively more densely populated\(^ {193}\) than the overall country and hosts 3.35 million inhabitants, accounting for 73 % of total population largely due to the presence of the capital city, Dublin.

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\(^{192}\) Expert review  

\(^{193}\) Population density in the region is 92 inhabitants per km\(^2\) versus 65 inhabitants per km\(^2\) on average across the Republic of Ireland
in the region (Central Statistics Office Ireland, 2014b). With a total Gross Value Added (GVA) of € 132.3 billion in 2013, the region contributed 82 % of the total country’s GVA (Eurostat, 2015c). The region's GDP per capita in 2013 was € 42 800 (Eurostat, 2015f).

The region benefits from international tourism, which includes attracting around 70 % of total overseas visitors to Ireland and 80 % of the revenue generated by the overseas tourism sector (4.6 million visitors and € 2.5 billion respectively). Of note is that Dublin is a tourism hotspot, attracting about 40 % of all overseas visits to Ireland or 52 % of all overseas visits to the region (National Tourism Development Authority, 2014). While no specific data exist on the number of international visitors engaged in water-related recreational activities, international tourist survey suggests that 52 % of the surveyed international tourists engaged in walking/hiking activities and further 9 % in angling.

Note that some data provided in this section may cover a slightly different area from Southern and Eastern Ireland. In addition, it is important to note that whilst agriculture is an important source of nutrient surplus it is not the only one, and the economic damages from nutrient surplus cannot be exclusively attributed to agriculture.

**5.6.3.2 Review of economic damages**

In order to better understand the economic damages associated with nutrient surplus in the Southern and Eastern Region, a wide range of literature was reviewed. Based on the classification of economic damages caused by environmental impacts presented in Annex 14, costs found in the literature for each damage category are presented below along with explanations on how these costs have been estimated.

It should be noted that all cost data reported in this section are taken from the primary studies found in the literature, and expressed in the value of the year when the study was conducted.

**Clean up and restoration costs (CRC)**

**Algae removal**

Algal blooms on beaches on the east coast of Ireland are monitored by the Marine Institute as part of the Phytoplankton Monitoring programme and are commonly detected during summer months in coastal areas (Marine Institute, 2014). For instance, an outbreak detected in 2013 in the Wexford County resulted in discolouration of the water and foaming on the beach in windy conditions. These events did not pose any threat to human health from recreational contact with water or consumption of fish exposed to the bloom. The beaches remained safe and open for use (Marine Institute, 2013).

A report from the Sea Lettuce Task Force describes Cork County Council’s activities targeting **clean up** of the areas affected by the presence of sea lettuce. Four options and their associated costs were presented for the use of the algae once removed: use as fertiliser for farms located in the vicinity of the affected beaches (€ 16 per tonne), use as soil enhancer for capping landfill cells (measure identified as not suitable), transport to a composting facility (€ 34.50 to € 67 per tonne depending on distance of the composting facility) or transport and disposal in landfill under appropriate agency approval (€ 260 per tonne) (Sea lettuce task force, 2010). According to the Environmental Protection Agency, Agricultural run-off and sceptic tanks were identified as the primary sources of the nutrients that led to the sea lettuce (Irish Independent, 2014).

**Water quality restoration**

Algal blooms can have a significant adverse impact on **public water supply**. However, no specific data on the cost of this impact was available for the region. Excessive nutrient loading and resulting algal
blooms, lead, in some instances, to a reduction of output capacity or even temporary closure. This has left some areas with no water available and Irish Water had to deploy water tankers to serve the area. In the absence of the data on the costs incurred, other direct valuation methods, such as the Avoided Damage Cost and Substitute Cost Methods\(^{194}\) can be used. These methods can be used to estimate the costs arising from a loss of an environmental resource, in this case, a water supply source. The Averting Behaviour (or Expenditure approach) is one of the methods and is based on analysing the cost incurred in defending against the negative impacts of environmental degradation such as costs of using an alternative water source (bottled water, supply from other sources), installation of water filters etc. (Gorlach, 2003). The method is based on estimating the costs of behaviours to prevent or mitigate adverse impacts of pollution i.e. costs of providing goods and services that substitute for the environmental loss (Abdalla, 1994).

Taking the example of the interruption of water supply in a local wastewater treatment plant, the following estimates can be calculated. The average price of bottled water can be used to estimate the likely scale of the costs associated with an interruption of water supply. Information available suggests that 41 % of total public water supply constitutes water that is unaccounted for (largely due to leakage), 39 % is supplied to domestic customers and the remaining 20 % to commercial users (PWC, 2011). Using the reported average price of € 1.50 per litre of bottled water and calculated loss of 14.75 million litres per day\(^{195}\), results in estimated costs of alternative supply were at € 22.1 million per day (Irish Times, 2008). The estimate, however, is likely to be an overestimate of the costs due to included commercial and sanitary use (e.g. use of baths). Using the shares above, domestic water use only is calculated at 9.75 million litres per day. In Ireland, domestic water consumption in average sized household is 145 litres per person per day, of which drinking water is accounting for 3 % and dishwashing for 22 %. Use of shower, toilets and washing machine is accounting for 72 % of domestic water consumption with the remaining 3 % being attributed to external uses (Department of the Environment, Community & Local Government, 2014). Using the reported average price of € 1.50 per litre of bottled water and calculated loss of 2.44 million litres per day (for drinking and dishwashing purposes) results in estimated costs of alternative domestic supply at € 3.7 million per day (Irish Times, 2008). Note that these costs estimates do not include the price that would be paid for the same volume of water used under normal conditions so are overestimates.

Furthermore, presence of algal blooms in an abstraction source would result in increased chemical and energy costs.

\(^{194}\) In some sources, Averting Cost Method is discussed as part of Revealed preference methods.

\(^{195}\) This excludes the estimated 41 % of losses due to leakage
Table 48 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure for clean-up and restoration purposes</td>
<td>Costs of bottled water to substitute the loss of water supply due to algal blooms (capacity reduction from 65 to 40 million litres per day of which 59% was domestic and commercial consumption). Average costs of bottled water - € 1.5 per litre</td>
<td>€ 3 656 250 (low estimate incl. only drinking and domestic) – € 22 125 000 (high estimate incl. commercial and sanitary uses)</td>
<td>Loss of water supply from one water treatment plant</td>
<td>2014</td>
</tr>
<tr>
<td></td>
<td>Treatment of algae removed from beach</td>
<td>€ 16 to € 240 per tonne depending on treatment techniques</td>
<td>Cork County</td>
<td>2009</td>
</tr>
</tbody>
</table>

Use value damages (UVD)

Excessive nutrient loads may also adversely affect a wide range of economic activities and sectors, including, in particular, aquaculture and shellfisheries, recreational uses and tourism, industrial water abstractors.

In the context of aquaculture and shellfisheries, algal blooms may result in marine organism mortalities. Typically, rather than removing and processing the algae, toxic algal blooms result in a temporary closure of lakes and shellfisheries, the aim of which is to ensure protection of human health through restricted access. Some of the species causing algal blooms in Ireland are toxic to fish and shellfish and pose a risk to human health (including through consumption of exposed shellfish).

Closure of shellfisheries due to failing water quality standards result in a loss of production and associated costs. For instance considerable shellfish toxicity observed in 2008 resulted in widespread closures of shellfish production areas (Environmental Protection Agency, 2010). In Ireland, Pollution Reduction Programme is prepared for each designated shellfish water. Nationally, there are 64 designated shellfish waters and each Plan considers key pressures on water quality, e.g. agricultural pollution and wastewater discharges and a program of action. Conditions of waters in the shellfish growing area are monitored by the Marine Institute.

It has been observed that wild fish tend to avoid algal blooms and this phenomenon has been shown to have an adverse effect on sea angler catches, with some fishermen also reporting clogging of the nets (Marine Institute, 2013), resulting in income reduction and additional operational costs.

While no explicit cost estimates were found in the literature in relation to the damage caused by eutrophication on shellfisheries and aquaculture in the region, data on the gross output value of these sectors are available (see Table 49).

Table 49 – Annual Gross Output Value in Southern and Eastern Region

<table>
<thead>
<tr>
<th>RBD</th>
<th>Gross output value (€)</th>
<th>Seaweed Harvesting</th>
<th>Inland commercial fishing</th>
<th>Aquaculture</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eastern</td>
<td>0</td>
<td>40 684</td>
<td>0</td>
<td>40 684</td>
<td>40 684</td>
</tr>
<tr>
<td>South-Eastern</td>
<td>1 000 000</td>
<td>448 739</td>
<td>9 932 164</td>
<td>11 380 903</td>
<td></td>
</tr>
</tbody>
</table>

No data is available on whether and to what extent the value generated by the sectors is negatively affected by excessive nutrient loads in the region.

In the context of *recreation and tourism*, eutrophication can have a range of adverse impacts, such as potential health risks associated with the exposure to toxic algal blooms and loss of tourism revenue due to nuisance caused (e.g. visual). Every year, the region welcomes about 4.6 million overseas visitors and generates €2.5 billion in revenues. Data on the annual value of water for different recreational uses are available (see Table 50, based on expenditure data).

<table>
<thead>
<tr>
<th>RBD</th>
<th>Gross output value (€)</th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Seaweed Harvesting</td>
<td>Inland commercial fishing</td>
<td>Aquaculture</td>
<td>Total</td>
<td></td>
</tr>
<tr>
<td>South-Western</td>
<td>2 666 667</td>
<td>1 754 626</td>
<td>31 675 551</td>
<td>36 096 844</td>
<td></td>
</tr>
<tr>
<td>Shannon River Basin</td>
<td>1 333 333</td>
<td>616 750</td>
<td>2 505 411</td>
<td>4 455 494</td>
<td></td>
</tr>
</tbody>
</table>

Source: (CDM, 2004)

However, no data is available on whether and to what extent the value generated by the recreational uses is negatively affected by eutrophication and algal blooms in the region.

The microbiological quality of *bathing waters* is generally very good with 97% of the designated bathing areas in Ireland (131 out of 135) achieving ‘sufficient’ water quality status in 2013\(^{197}\). While a range of factors contribute to localised poor quality, such as inadequate sewage treatment and pollution accidents, monitoring results indicate that there is little risk to bathers’ health from pollution in designated bathing areas of the country (Environmental Protection Agency, 2010). However, bathing waters compliance is assessed based on the *E.Coli* and *Enterococci* monitoring only.

Damages associated with a disruption to recreational use of beaches are two-fold. On one hand, closure of the beaches could result in losses to dependent industries, such as catering and hospitality. The extent of the damage caused by beach closure can be estimated based on the information on the average spend of recreational visitors and the number of users affected. Data available on the number of domestic holiday trips and associated expenditure (National Tourism Development Authority, 2014) suggests that the average expenditure per short holiday trip (1-3 nights) in 2013 was €194. Assuming the average duration of 2 days results in an average daily spend of €97 per trip. No data, however, is available to indicate the number of businesses adversely affected by the algal blooms through the

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\(^{197}\) In EPA (2014) *The Quality of Bathing Water in Ireland*. An overview for the year 2013 indicated that 4 bathing waters (1 inland and 3 coastal) failed to comply with the minimum mandatory standards and were classified as having ‘poor’ quality status.
Reduced visitation rates. Indeed, occasional and localised presence of algal blooms in water bodies have in the past resulted in warnings being posted advising people not to swim in the water bodies affected. Available information suggests that frequent algal bloom on Lough Derg is posing a serious threat to tourism in terms of visitor numbers and revenues as it is putting one of the country’s top pleasure lakes off limits to bathers for up to a month each year (The Independent, 2005). While the number of recreational users adversely affected by algal blooms is not known, the study on the activities engaged in by domestic, i.e. Irish holiday makers suggested that 22% of all domestic tourists engage in watersports, while further 4% engage in angling activities (National Tourism Development Authority, 2014).

The nuisance caused to recreational users themselves can be estimated based on the Willingness to Pay for improving water bodies to bathing waters standards or for avoiding a deterioration in water quality below the set standards. For instance, Georgiou et al. (1998) estimated the WTP for a beach in East Anglia (UK) for preventing deterioration in water quality below mandatory standards at €2.41 per visit\(^\text{198}\). It should be noted, that a number of affected recreational users might choose to visit alternative beaches situated within a short travel distance, thereby minimising adverse impact. In such instance, the use of alternative beaches may be associated with additional travel costs. However, no data is available on the number and duration of beach closures and the number of affected recreational users in the region.

Secondly, poor water quality can have an adverse impact on recreational angling which is one of the top sources of revenue in the shoreline villages and towns. The data available suggests that in 2012, the total annual value of recreational fishing to the Irish economy was in the region of €0.75 billion. This is split between €555 million in direct spending on angling and €200 million of indirect spending. Recreational angling was found to directly support 10,000 existing jobs, many of which are located in the most peripheral and rural parts of the Irish countryside and along the coastline (Inland Fisheries Ireland, 2013). Dissolved oxygen, total suspended solids, temperature, and water quantity are important water quality parameters for fish and, therefore, fishing. There is, however, a lack of quantitative evidence about the link between water quality and extent of recreational angling activity. In other words, the value of the damage caused by excessive nutrient loads to angling or the extent of potential increase in number of angler days per year as a result of water quality improvements is unknown.

For instance, concerns were raised regarding the impact of algal blooms on angling tourism at Lough Derg suffering from frequent algal blooms. As algal bloom seems to persist the locals fear that anglers and, in particular, those coming from abroad will not be returning due to extremely low fish takes. The total willingness-to-pay (WTP) given current water quality, for salmon anglers in Ireland is estimated at €262 per day (Goodbody Economic Consultants, 2008). As regards the demand for water-based leisure activity in Ireland, the national consumer surplus is estimated at €26.3 m for sea-angling day trips, €38.9 million for boating, €233 for swimming and €61.5 for other beach/sea trips (Goodbody Economic Consultants, 2008). No region’s specific disaggregation of the national consumer surplus is available.

There is, however, an extensive body of research available suggesting that water quality affects the value of property adjacent to or in the immediate vicinity of a waterbody and that property values increase with improved water quality and decreased algal blooms and vice versa.

A summary of studies looking at water quality indicators and property value highlight that waterfront properties can lose value if the water quality reduces specifically as a result of an increase in turbidity (reduced clarity), algal blooms and unpleasant odours (Dills, 2003). In addition, there can be benefits

\(^{198}\) £1.18 per visit (2001 prices) converted to €1.90 per visit (2001 prices) and €2.41 using 1.27 inflation index
from reductions in nutrients, coliform bacteria and toxic contaminants. An OECD study shows that within close proximity of areas that benefitted from improved water quality, property values were found to be 11% to 18% higher than properties next to water bodies with low quality (OECD, 2011). In particular, this added value is estimated to be 10-40% (mean nearer 20%) for residential properties in the UK (Wood, 1999).

One Irish study reported Willingness to Pay values for adjusting agricultural practices to achieve improved river and lake quality (and to remove green algae from rivers and lakes in particular) at €115–249 per person per year (O’Leary, 2004).

**Passive use value damages (PUVD)**

**Water pollution – Negative effect on health**

Cyanotoxins\(^{199}\) possess an irritating potential and in human exposure cases also act on the gastrointestinal system via oral or respiratory routes. In particular, hay fever-like symptoms, pruritic skin rashes and gastro-intestinal symptoms are most frequently reported (Stewart, 2006). Exposure of recreational users to polluted water, therefore, could result in illness, which carries with it the possibility of working days being lost, the cost of hospital admissions and treatments etc.). Such risks would be relevant to both immersive water activities, such as bathing, surfing and diving, as well as to limited contact water activities such as kayaking, canoeing etc. While health related costs could be estimated using the costs of healthcare provision, no information seems to be available on the number, severity and duration of water-borne diseases in the region or Ireland in general.

One study assessed the increased incidence and damage costs associated with nitrate in drinking water. The authors estimate that 2.8 million people are exposed in Ireland to NO\(_3\) concentrations above 25 mg/L in Ireland leading to the loss of healthy life years at €3 million per year with the unit health damage cost from N-leaching from agricultural land at €0.1 per kg N (Brink & van Grinsven, 2011). In the absence of the region specific results and assuming equal distribution of exposure among population, health damage costs in the region are estimated at €2.2 million per year.

**Water pollution – Negative effect on ecosystems**

Eutrophication can have diverse impacts on ecosystems and biodiversity. As mentioned above, algal blooms have been detected along the east coast on a number of occasions in the course of previous years. For instance, an outbreak detected in 2013 in County Wexford resulted in discolouration of the water, and foaming on the beach in windy conditions (Marine Institute, 2013).

The production of foam, and in some extreme cases anoxia, can result in marine organism mortalities.

Although it is clear that algal blooms lead to economic cost falling in the PUVD category, only patchy quantitative data is available. No regional-level estimates of the magnitude of these damages could be found. There are very few estimates available on the values falling within this category.

**Air pollution**

Excessive nutrient loading can also lead to air pollution and consequently cause physical and economic damages to human health and ecosystems in the region.

\(^{199}\) Cyanotoxins are produced by cyanobacteria whose reproduction is increased in environments with high nutrient concentrations.
In particular, harmful substances in the air can cause both immediate and long-term damage to health. According to the EPA, air quality in Ireland is generally good; however, levels of some pollutants in Ireland are at concentrations that may impact on health including nitrogen dioxide in cities from traffic emissions. Levels of PM10 and PM2.5 are also of concern due to the ability of these small particles to penetrate deep into the respiratory tract.

A study (2011) reports the average unit damage cost for health impacts in Ireland by airborne NH$_3$ at €3/kg N and at €12/kg N for NO$_x$ (Brink & van Grinsven, 2011). In addition, the study estimates the average damage cost of deposition of NH$_3$ on terrestrial ecosystem between €2 and €10/kg N and the cost of N$_2$O emissions on public health between €1 and €3/kg N in the EU27 (Brink & van Grinsven, 2011).

**Policy action costs (PAC)**

The costs of measures aimed to reduce nutrient pollution from water and sanitation and agricultural sectors are discussed in the previous sections. In particular, implementation of the Good Agricultural Practices Regulations (2006) required €2 billion investments since 2006 to prevent pollution while water and wastewater services cost approximately €1.2 billion per annum. Furthermore, targeted plans are also developed for designated shellfish waters and bathing waters.

No specific plans seem to exist in the region that would target algal blooms specifically. A range of monitoring programmes exists, including national phytoplankton monitoring programme executed by the Marine Institute since the 1980s. No published cost data were obtained for the Programme.

**5.6.4 Good practices to reduce the nutrient losses at farm level**

For Southern and Eastern Ireland, measures have been selected to address the causes of nutrient losses, which mainly relate to the agricultural pressure related to the high livestock density. The measures also take into account the fact that in Ireland and in the Southern and Eastern region, the predominant manure system is slurry based, followed by farmyard manure system. The first set of measures focuses on the reduction of the source of pollution through the conversion of agricultural land to organic farming, the transfer of manure to other farms, the improvement of development of the fertilisation management plan. The use of digestate from anaerobic digestion can also help at ease the transfer and use of manure while also valorising the high amount of manure produced in the region through the production of energy. Another type of good practice relates to the reduction of the amount of nutrients lost during housing and storage by improving manure collection and covering the slurry tanks. The third set of measures concerned to the prevention of nutrient losses during manure application on field or on pasture. The measures relate to the lengthening of the grazing season or the adoption of rotational grazing, the choice of the best suited application techniques, the incorporation of manure into the soil, the spreading at the right time for stage of crop growth, the use of adequate tillage technique and the consideration of climatic and geographic conditions.

**5.6.4.1 What has already been done in the region**

A combination of mandatory and voluntary measures aims to address the nutrient surpluses in the Southern and Eastern Ireland region. The Directive establishes a maximum application standard of 170 kg N/ha from livestock manure (i.e. around 2 livestock units per ha). However according to Decision
grassland farmers in Ireland may apply for a derogation from this limit to allow them to stock at levels up to 250 kg organic N/ha (2.9 livestock units per ha), but this obliges them to meet more stringent recording and reporting requirements. This derogation has principally been used by dairy farmers (Buckley, et al., 2013).

The European Union (Good Agricultural Practice for Protection of Waters) Regulations 2014 (S.I. No 31) (1), which implement the EU Nitrates Directive, establish a number of measures which are compulsory throughout Ireland. They represent the Third National Action Programme adopted by Ireland under the Nitrates Directive and are also part of the cross-compliance requirements for EU funded schemes. The measures include rules on the timing and amount of fertiliser application, as well as maximum fertilisation rates for N and P application to grassland, tilled land, horticulture and fruit crops in order to limit fertiliser over-application. The measures emphasise management of livestock manure and fertilisers and require that monitoring takes into account the impact of the agricultural activity on water quality. Inspections are conducted to verify that the requirements are respected and during the 2008-2011 period a total of 4 % of the Irish farms were visited (European Commission, 2013b).

In addition, in the region the role of knowledge transfer is very important. The region benefits from a range of very active networks and tools which have been created in order to assist farmers in, inter alia, reducing nutrient losses. For example:

- **The Farm Advisory System (FAS)**: almost 700 advisors have been trained by the Department of Agriculture, Food and the Marine (DAFM) and support Irish farmers in their day to day activities;

- **Agriculture Catchment Programme**: the ACP monitors the effectiveness of the measures in Ireland’s action programme. Research findings from this programme will contribute to future policy decisions. It is implemented by Teagasc and funded by DAFM, this programme has provided free advice and soil surveys in 6 catchments. Nutrient management plants were also drafted for farms in order to help farmers use their nutrients in a better way;

- **Incentivised discussion groups to accelerate knowledge transfer amongst farmers have been organised and have involved around 15 000 farmers. These are:**
  - **Beef Technology Adoption Programme**: this initiative encourage cattle farmers to adopt new technologies to improve their business;
  - **Beef Genomic Scheme**: this programme supports farmers with improving the genetic quality of their livestock;
− Dairy Efficiency Programme\(^{205}\): this programme encourages dairy farmers to share knowledge on improving efficiency of their farms; and

− Sheep Technology Adoption Programme\(^{206}\): this initiative encourages farmers to adopt new technologies to improve their business.

• Beef Carbon Navigator\(^{207}\): this tool has been developed by Teagasc and Bord Bia (the Irish Food Authority) to help farmers set improvement targets in key management areas and visualise the results for their farms in terms of environmental and economic performance;

• Bord Bia’s Beef and Lamb quality assurance scheme\(^{208}\);

• BETTER (Business, Environment and Technology through Training Extension and Research) Farm Programme\(^{209}\): this tool has been developed by Teagasc and aims at transferring knowledge from research into practical advice for farmers;

• Smart Farming\(^{210}\): aims at improving farm returns with better resource management.

Smart Farming is implemented through the Irish Farmers’ Association. A dedicated website and a downloadable application provides guidelines on improving the farm’s soil fertility, managing grassland, selecting machinery, using water, energy and feed efficiently, managing time and inputs and waste. Farm level ‘cost-saving studies’ are conducted by members of the Smart Farming team and identify cost-savings measures which double as environmentally positive. In 2014, the Network conducted 30 farm cost-saving studies. The aim is that farmers will in turn inform other farmers of their findings. All the measures are voluntary and based on collaboration. The approach focuses on presenting the economic benefits of changing practices to farmers.

Finally, financial support for Irish farmers is available through national existing schemes such as the Targeted Agricultural Modernisation Scheme (TAMS)\(^{211}\); Agri-environmental scheme (AEOS) and the Leader scheme which forms part of the Rural Development Program for 2014-2020. The latter totals € 4 billion out of which € 1.9 billion is from Ireland and the remaining is from the EU\(^{212}\). Under this program, funding totalling € 1,450 million will be available under the GLAS (Green Low-Carbon Agri-Environment Scheme) to fund measures such as riparian margins, fencing of watercourses to prevent cattle access, low emission slurry spreading and green cover for arable land. The maximum payment is € 5,000 and will be available for up to 50,000 farmers throughout the 2014-2020 period. It is anticipated that 25,000-30,000 farmers will participate in the Scheme in 2015. The adoption of this scheme entails compulsory soil sampling and compulsory involvement of an advisor from FAS. In addition, the GLAS+

\(^{205}\) http://www.agriculture.gov.ie/farmingsectors/dairy/dairyefficiencyprogramme/


\(^{207}\) http://www.teagasc.ie/publications/2012/1637/Paul_Crossan_Teagasc.pdf

\(^{208}\) http://www.bordbia.ie/industry/farmers/quality/pages/beefqualityassurancescheme.aspx

\(^{209}\) http://www.teagasc.ie/advisory/better_farms/

\(^{210}\) http://smartfarming.ie/

\(^{211}\) http://agriculture.gov.ie/farmerschemespayments/farmbuildings

\(^{212}\) Note that the Rural Development Programme has been adopted by the European Commission on 26 May 2015. See http://ec.europa.eu/news/2015/05/20150526_en.htm.
scheme will provide to farmers who take on particularly challenging environmental actions the opportunity to qualify for an additional payment of up to € 2,000 per annum.

5.6.4.2 Good solutions to reduce nutrient losses in livestock production

In the Southern and Eastern Ireland region, livestock are kept outdoors for an important share of the time and the following measures have been identified as potential ways to reduce nutrient losses from these livestock.

Optimising grazing intensity and rotational grazing

The Southern and Eastern region relies on outdoor grazing for its dairy and cattle livestock. On average, dairy cows are kept outdoors for approximately 66 % of the time, cattle (beef) for >60 % of the time while about one third of laying hens are reared in free range conditions (Mounsey, 2012).

Grass continues to be the dominant crop in Ireland and mild winters and cool summers ensure grass growth through almost the entire year (Environmental Protection Agency, 2012). While the grazing season is already long, it has been estimated that the Southern part of the region could extend its grazing season further by 20-30 days (representing a 10 % increase)\textsuperscript{213}.

Alongside outdoor grazing, the establishment of rotational grazing (i.e. moving cattle regularly to new paddocks) was found to be successful in the region\textsuperscript{214}. Rotational grazing ensures a more even spread of the nutrients contained in the manure and reduces soil compaction and de-vegetation due to regular movement of the livestock.

Prefer organic farming

Conversion from conventional to organic farming practices could reduce nitrogen leaching due to reduced application of organic and chemical fertilisers and stock numbers. The former Irish Organic Farming Action Plan 2008-2012 reported a target of 5 % of the Utilisable Agriculture Area (UAA) being under organic production by 2012. This represented an increase in actual land area of circa 520 % in comparison to the 2008 level of 0.9 %. Beef production is believed to hold the greatest potential for increasing the volume (and land area) reared organically (Irish Department of Agriculture, Fisheries and Food, 2008). However, the target was not reached since the organic farms represented 1.2% of the UAA in 2013 (Irish Department of Agriculture, Food and the Marine, 2013).

While it is unclear whether organic farming will provide sufficient income for farmers, it is being practised successfully in Ireland at stocking rates of up to 2 livestock units per hectare (Teagasc, 2014). An Irish study on organic farming suggests that it can result in a relatively higher net margin in comparison to conventional farming systems when accounting for reduced output, additional costs on one hand and for price premiums and payments and subsidies (Teagasc, 2009). However, a range of barriers to such growth have been identified by farmers, including higher food prices potentially affecting consumers’ willingness to pay premium prices and increases in (conventional) beef and milk prices making the conversion more challenging (Irish Department of Agriculture, Fisheries and Food, 2008). In addition, food production levels and the premium prices for organic produce in Ireland would likely decrease if a greater number of farmers in this region adopted organic farming, possibly resulting in lower food exports (reduced revenue) and reduced income for farm families.

\textsuperscript{213} Expert review

\textsuperscript{214} Regional conference feedback from farmers having adopted the measure
The Irish organic market has been described as static in the past few years. An increase of the organic production could help stimulate this sector. A 2010 survey revealed that 57% of Irish consumers would buy more organic products if those were being produced in Ireland (Teagasc, 2011a). As a result the development of the organic farming sector could contribute to both the expansion of the organic food market and the reduction of nutrients used in agriculture.

**Transfer manure**

Where there is a surplus of nutrients, manure can be exported to other farmland with spare nitrogen capacity. The transfer of manure is governed by the requirements of the Nitrates Regulations. There are limits to the content of N from manure that can be applied in Ireland. This measure is likely to be appealing to farms exceeding a production of 170 kg N/ha or 250 kg N/ha if under derogation and potentially some receiving farms within the limits set by the legislation.

Odour nuisance could be increased by manure transport. It should be noted, however, that in the Southern and Eastern region dairy cows are kept outdoors for about 66% of the time and manure excreted cannot be collected and managed as easily as manure from housing. Biosecurity and animal health due to diseases that can be transferred are issues that need to be taken into account when transferring manure.

Based on a national farm survey in Ireland, between 26-43% of farmers reported a willingness to import pig or poultry manures either on a payment or free of charge basis. Demand is strongest among arable farmers, younger farmers and those of larger farm size with greater expenditure on chemical fertilisers per hectare and those not benefiting from a nitrates derogation (Environmental Protection Agency, 2012). In the Southern and Eastern region, the majority of the farms are livestock farms with excess manure produced, so the demand for manure transfer may be lower than in the rest of the country. Variation can be observed within the NUTS 2 region. For example, for County Kilkenny, the average stocking rate is of 111 kg/N/ha for dairy and beef farmers, so for this county in general, excess manure is not produced.

A survey of the farming practices conducted in 2011 found that in Ireland 4% of all farmers’ imported slurry and/or farmyard manure and 1% exported slurry and/or farmyard manure. The importations represent 652 000 t of slurry and 25 500 t of farmyard manure while the exportations represent 260 t of slurry and 10 000 of farmyard manure (data for pig and poultry farms are missing) (Hennessy, et al., 2011). It is unclear whether there are non surplus areas close to the surplus areas and the extent to which this measure is economically feasible. However, it should be noted that tillage farmers can avail grant aid to put in place storage to allow them to import livestock manure (Irish Department of Agriculture, Food and the Marine, 2015).

**Cover slurry tanks**

In the Southern and Eastern region, cattle slurry is mostly kept under slatted tanks, with some in open tanks. The European Union (Good Agricultural Practice for protection of waters) Regulations 2014 do not specify a need for a cover on slurry tanks. The Nitrate Regulations do, however, require the allowance of an increased capacity to accommodate rainfall in the case of uncovered tanks. Feedback from the regional conference and the stakeholder consultation of this project indicate that most farmers comply with the manure storage capacity (ranging from 16 to 22 weeks) required by the action programme implementing the Nitrate Directives. According to 2013 Eurostat data on coverage of manure storage units, 80% and 90% of the holdings with liquid manure and slurry storage vessels respectively have covered slurry tanks in Southern and Eastern Ireland (Eurostat, 2014h). As a result the potential for further improvement by implementing this measure is limited.
From 2001 to 2010, a grant programme under the Farm Waste Management Scheme has led to up to €2.11.2 billion to be spent during this period on, inter alia, improving manure storage capacities. Currently, farmers can avail of grant aid under the Targeted Agricultural Modernisation Schemes (TAMS) to cover existing slurry tanks (Irish Department of Agriculture, Food and the Marine, 2015).

**Improve collection of manure from housing**

Data available suggest that when cattle (beef) is kept indoors, slatted floors or loose housing system is used (Mounsey, 2012). Eurostat data for Southern and Eastern Ireland suggests that the main housing system is loose with solid dung and liquid manure (25,230 out of 58,740 holdings), closely followed by loose housing system with slurry (20,490 holdings) (Eurostat, 2013). Partly slatted floor is more suited for larger units due to cost of converting existing buildings (i.e. installation of slatted flooring and building collection pits) and thus is appropriate for the region given the large number of intensive farms located there.

**5.6.4.3 Good solutions to reduce the nutrient losses in crop production**

Inappropriate spreading of manure in Southern and Eastern Ireland contributes to nutrient losses and raising the levels of nutrients (primarily nitrates) in water bodies. The following measures have been identified as potential ways to reduce the excessive fertilisation of soils in the Southern and Eastern Ireland.

**Improve the use of fertilisers through use of nutrient management plans**

The use of fertilisers, and in particular inorganic fertilisers, is an important contributor to emissions of ammonia, and surplus of nitrogen and phosphorus. The large variability in field-level nutrient application rates relative to recommended rates and high spatial variability in soil P status indicates significant potential to improve nutrient management at the field level (Environmental Protection Agency, 2012).

Recent research conducted by Teagasc has found that at farm level there can be variation on the level of fertilisers that are applied on field, and some dry-fields were systematically over-fertilised. There also appears to be scope for use of this measure as although many farmers performed soil (66 %) and manure (47 %) analysis only few of them elaborate a nutrient management plan (27 %), according a survey performed by Teagasc in 2010 (Wall, 2014).

In order to improve the efficient use of fertilisers (including inorganic fertilisers), farmers in the Southern and Eastern region of Ireland have been encouraged to use simplified nutrient management plans developed by Teagasc in the Agricultural Catchments Programme. These plans are based on a soil fertility map which identify the needs of each parcel of fields. From the results, a colour-coded map is produced (Figure 97). In the example below, the map indicates which fields are in need of additional phosphorus (blue or yellow) and those which already have a high level of phosphorus (red on the figure).
Feedback from farmers at the regional conference highlighted the potential success of this measure, being simple and clear to use.

**Use digested manure from biogas production**

Anaerobic Digestion (AD) is set to play a significant role in the future energy market in Ireland (SEAI, 2011). It is a well-developed and commercially available technology contributing to energy independence/self-sufficiency as it has the potential to supply constant electricity, heat, gas and transport fuel. The technology has also been associated with negative effects such as land use change, however in the context of Ireland AD could contribute to the achievement of the set Renewable Energy targets. In addition to the co-materials, on average 6 cows or 110 tonnes of slurry can satisfy one household’s annual electricity demand of 4 000 kWh. The price of an AD plant in Ireland ranges between € 4 000 to € 7 000 per kW, i.e. installation of a 250 kW plant would cost € 1.2 million to € 1.7 million (SEAI, 2011). In 2014, only 6 anaerobic digestion plants existed in Ireland, mostly in the south and south-east of the country.

A case study in the Limerick County highlights a number of essential items that need to be addressed for the installation of an AD plant, including planning consent, application for waste permit/licence, Animal By-Product (ABP) approval from Department of Agriculture, Food and the Marine (DAMF), grid...
In Ireland, slurry is applied by tanker and splash plate on 97% of all Irish farms in 2009 (Hennessy, et al., 2011), a method which does not distribute slurry evenly over land and often results in over application (Ryan, 2005). For farmyard manure, side discharge is the most common method and is used by 57% of farms. Rear discharge is the next most common method and is used on 27% of farms. This application technique leads to volatilisation and nutrient losses due to uneven spreading.

In Ireland, the use of upward facing splash plates or a sludge irrigator to broadcast liquid manure is banned by the European Union (Good Agricultural Practice for protection of waters) Regulations 2014 (S.I. No 31 of 2014). The Regulations specify that organic fertilisers must be applied in as accurate and uniform a manner as is practically possible. No requirements are set in relation to the use of specific application techniques.

In Southern and Eastern Ireland grasslands are the most dominating land use so the trailing shoe technique would be the best suited application technique for slurry as it minimises grass contamination and would not run into the same issues that injection would encounter on some of the region’s soils. It was noted that the implementation of this technology in the region, where the majority of the slurry is broadcasted by splash plate method, has been limited by the high purchase and running costs of the machinery required. In Ireland, it is estimated that the switch to trailing shoe could reduce ammonia emissions to air by 28% which in turn would mean an increase in nitrogen for the crops and a reduced run-off of nitrogen to water bodies (Lalor, 2010b).

Costs identified for Ireland are presented in Table 51 distinguishing costs for splashplate application and trailing shoe application techniques. For each, costs are split between costs from the slurry tanker and the costs from the spreading equipment (i.e. the splashplate and the trailing shoe).
In Ireland, a dedicated support programme aiming to promote the uptake of low ammonia emission application of organic fertiliser was introduced in 2005. The programme provided for a 40% subsidy for low emission ‘trailing shoe’ slurry spreading technology. The response to the ‘trailing shoe’ grant has been slow and it was kept as the Rural Environment Protection Scheme (REPS) under the 2007-2013 Rural Development Programme with a target of 6,000 farmers using trailing shoe or injection technology by 2013\textsuperscript{215}. The Rural Development Programme 2014-2020 also includes support for low-emission slurry spreading measures.

\textit{Incorporate manure into the soil}

Incorporation of manure once spread is an efficient way to limit nitrogen losses to air through ammonia volatilisation. In the context of Southern and Eastern Ireland, such a technique would be limited to tillage land crops and reseeded grassland, which represent 30% of the UAA in the region. According to Eurostat data, in 2010, out of 4.9 million of hectares of UAA for the whole of Ireland, solid manure was applied to 1.1 million hectares and incorporated immediately only for 44,000 hectares of UAA, this seems to suggest that there is a scope for this measure to be applied more widely (Eurostat, 2013e).

\textit{Use adequate tillage techniques}

Favouring no-till and reduced tillage also holds great potential for reducing nutrient losses. Southern and Eastern Ireland has some of the best drained soils in Ireland, which are more likely to have lower levels of soil organic matter relative to the wetter midlands and western regions. In the South-Western part of the country, for instance, soils are often waterlogged, which allows the soil to filter and store nutrients. As a result, tillage can have a limited but positive effect on some soils by aerating the soil, which can decrease nitrous oxide emissions and run-off (Rochette, 2011). While implementation of reduced tillage and no tillage may result in some labour cost-savings, both may require significant investments for the purchase of specialist machinery and/or result in an increased use of herbicides and pesticides. Yield loss may also potentially result due to weeds, leading to further costs. Further potential measures include restrictions on cultivation of slopes such as contour ploughing or reduction of the area under autumn ploughing in regions susceptible to soil erosion.

\textit{Spreading at the right time for stage of crop growth}

In addition to the restrictions on time and place, like prohibited spreading periods and the general limitation on N and P application to the crops’ needs, manure application should be planned based on the crop development stage and environmental conditions (temperature, wind, solar radiation, precipitation, and soil conditions). For this, application timing management systems (ATMS) could be

\textsuperscript{215} Ireland CAP Rural Development Programme 2007-2013
used. ATMS are mainly computer-based models that calculate the amount of nutrients lost during and following application for the average regional environmental conditions. ATMS encourages farmers to spread manure in cool, windless and humid conditions, on flat land and away from surface waterways and ideally in the evening, when wind speed and air temperature are lower. When spreading on tilled land, application on freshly cultivated soils allows for more rapid infiltration. This technique was highlighted during the workshop as being in early stages and quite costly. As a result, early adopters are more likely to be intensive farmers.

**Consider climatic and geographical conditions**

Nutrient transfers in the region are affected by a range of environmental factors including a high number of wet days, combined with the seasonality of rain and abundance of soils rich in clay, resulting in soils being often waterlogged, increased erosion and sedimentation on slopes and near watercourses by tillage and livestock access to stream and rivers banks (Met Eireann, 2014), (Environmental Protection Agency, 2013a), (Joint Research Centre, 2013b), (SWAN, 2013). These climatic conditions must be taken into account when making decision of applying manure or other fertiliser to the soil. Feedback from the regional conference was that a simple tool similar to the soil mapping could be developed in order to help farmers with deciding the best timing to apply the manure (with links to meteorological data) and the quantity to apply.

It is noted that the predominance of grassland farming in the Southern and Eastern region, with both dairy and beef cattle spending a large proportion of their life outdoors, may contribute to increased sedimentation from river banks. Fencing off surface water to limit cattle access to water has been considered during the review of Ireland’s Nitrates Action Programme (NAP) in 2013 but not included in the 3rd NAP pending collation of further evidence on the risks to water quality (Department of the Environment, Community and Local Government and the Department of Agriculture, Food and the Marine, 2013). The measure, however, is available as a voluntary option to cattle farmers under the Agri-Environment Options Scheme (AEOS) (Department of the Environment, Community and Local Government and the Department of Agriculture, Food and the Marine, 2013) and under the Green, Low carbon, Agri-environment Scheme (GLAS) in the Rural Development Programme 2014-2020. It is interesting to note that France has attempted to implement this measure but that the uptake was limited. Fencing off surface water was found to be inconvenient and posed a financial burden on farmers in charge of their maintenance for very little benefit in return.

Even simpler measures could also be encouraged in order to reduce nutrient losses, for example a farmer participating in the regional conference indicated that starting to plough vertically (i.e. in comparison to the slope) had made a difference in the fertilisation and yields of his fields.
5.7 Weser-Ems (DE)<sup>216</sup>

**Box 10 – Weser-Ems case study - In brief**

The Weser-Ems region is characterised by a high nitrogen surplus. The elevated nitrate levels (two-thirds of the groundwater sample points exceed 50 mg/l) in the groundwater bodies pose a threat to drinking water quality and possibly to human and ecosystem health. The nutrient losses through \( \text{NH}_3 \) and \( \text{NO}_x \) emissions results in increased N deposition and causing acidification of water (groundwater and marine water) and soil. The latter also enhances the nutrient load to groundwater and surface water, eventually affecting marine waters. Eutrophic marine waters present algae blooms often leading to toxic algae occurrence. They replace sea grass, which is a valuable habitat for birds and fish.

The impacts of nutrient losses are caused by the intensive livestock production that leads to high amount of manure available in the region that can exceed crop needs in counties such as Vechta and Cloppenburg. The intensive energy crop production, the excessive application of fertilisers that is partly due to a lack of awareness on the agronomic needs and the environmental impact of agricultural practices and the lack of appropriate storage and application equipment for manure also causes nutrient surplus and losses. Furthermore, the drainage of former bog areas and the current cultivation of these organic soils for agricultural production presents another cause for the mentioned impacts. Lastly, the losses of nutrients are enhanced by natural factors such as sandy soil, the presence of organic soils that are drained to produce crops, high groundwater table and abundant rainfalls that increase leaching risks.

Costs for the reduction of nitrate in drinking water include the cost of preventive measures and the cost of nutrients removal. The cost of preventive measures varies according the measures. For instance, among the voluntary activities of farmers, contracts are in place between farmers and water utilities who agreed to limit the amount of nitrogen containing fertilisers for those areas. The water users pay a compensation to the farmers which is part of their water bill to mitigate the high nitrate levels in water bodies. The cost of this measure is € 4.3 million. Another example is precision farming whose cost varies from € 895 to € 723/kg N. The cost of N removal from water varies from € 5 to € 125/m³ water according the technology. The passive use value damages include the effects on the surrounding natural environment (e.g. damages of ecosystems neighbouring arable land through nitrogen deposition) from the farming industry as well as eutrophication of aquatic ecosystems. Thus, the \( \text{NH}_3 \), \( \text{NO}_x \) and the formation of particulate matter cause costs of € 32/kg N-\( \text{NO}_x \) and € 22/kg N-\( \text{NH}_3 \) emitted including medical treatment, wage and productivity losses as well as the willingness to pay for avoiding environmental and health effects in Germany. The cost of acidification is evaluated to € 1 800/t \( \text{NO}_x \) based on the willingness to pay for avoiding the damages by EU citizens. Policy action costs with reference to nutrient surpluses that are budgeted for Weser-Ems include training for farmers on optimised nutrient management and amounted € 1.8 million in 2009.

The good practices proposed for Weser-Ems mainly aim at reducing nitrate leaching to groundwater and ammonia emissions. They especially apply for the counties Vechta and Cloppenburg where several intensive indoor livestock rearing are located and where the amount of manure produced exceeds the crops’ needs. The first set of measures presented in this section aims at limiting the amount of nutrient produced or applied on field through the adjustment of the feed quantities, by raising awareness among farmers through support of fertiliser management, using the most suited application techniques and converting arable land to unfertilised grassland in areas at risk. The second set of measures aims to limit or reduce the nutrient losses by covering manure during storage and spreading at the right timing. The third set of measures concerns manure recycling by encouraging slurry separation that would help using manure as fertiliser and valorise the solid fraction if it is used to produce energy though anaerobic digestion. Rewetting of organic soils and the establishment of paludicultures are suggested to avoid the further mineralisation of organic matter and leaching of nutrients to the groundwater.

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The Weser-Ems region (total area of 14 970 km²) is situated in north-west Germany in the federal state of Lower-Saxony\textsuperscript{217}, and represents 31 % of the Lower-Saxony territory. As the name suggests, the region is divided by a watershed; the western part belongs to the Ems catchment whereas the eastern part belongs to the Weser catchment. In the north-west, together with parts of Bavaria, the Weser-Ems region shows the highest nutrient surplus in Germany.

In Lower-Saxony, 2.6 million ha were dedicated to agriculture in 2014 (Federal Statistics Office of Lower-Saxony, 2014). In 2012, the Weser-Ems region represented 32 % of the Lower-Saxon agriculturally used area (Eurostat, 2015a). In Lower Saxony, 15 % of area was grassland, 40 % was arable land and 25 % was forest in 2014 (Federal Statistics Office of Lower-Saxony, 2014). This division is only indicative for the study region, as most of the Lower Saxon forest is situated in the south, in the Harz Mountains; the north of the study region has a higher share of grassland. In 2013, the Weser-Ems region accounted for 52.1 % of the Lower Saxon agricultural production\textsuperscript{218} and 11 % of the total German agricultural production (Eurostat, 2015h).

Livestock production is intensive in the Weser-Ems region. More than half of the chicken for fattening, around one-third of the pigs and laying hens, and one-fifth of the cattle of Germany are produced in Lower-Saxony with the focus of production in the Weser-Ems region (Flessa, et al., 2012). The region

\textsuperscript{217} Lower Saxony has a size of 4 761 800 ha

\textsuperscript{218} Agricultural output, production value at basic price
shows a high livestock density of 1.9 LU/ha in 2010 (Jansen-Minßen, 2012) (the value exceeds 3 LU/ha in 13 communities of the region), which is relatively high compared to Lower-Saxony and to Germany more broadly (Bäurle & Tamásy, 2010; Niedersächsisches Ministerium für Ernährung, Landwirtschaft, Verbraucherschutz und Landesentwicklung, 2011).

In Lower-Saxony, arable land accounted for 1.9 million ha in 2014 (Federal Statistics Office of Lower-Saxony, 2014). Maize, mainly for bioenergy production, is produced on 20% of the agriculturally used area and 16% is used for root crop and wheat production. Other grains accounted for 15% of the agriculturally used area (Federal Statistics Office of Lower-Saxony, 2014).

Adapted from (Marketinggesellschaft der Land- und Ernährungswirtschaft e.V., 2010)

Figure 99 – Pattern of agricultural production in the study area

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219 Ministry of Food, Agriculture, Consumer protection and Regional Development of Lower-Saxony

220 Marketing association of the agro and food industry

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5.7.1 Notable impacts of nutrient surplus

This section focuses on the impacts of nutrient excess that are specifically related to agricultural practices in the Weser-Ems region. In this region, the main impacts are caused by nitrogen and phosphorus surpluses that mostly affect water and air. Potassium excess is not a major issue in the region. Furthermore, Germany has adopted the "whole territory approach" under the Nitrates Directive, with obligatory measures in place in its whole territory.

Source: Own maps with data from (Landwirtschaftskammer Niedersachsen, 2013)

Positive numbers indicate a surplus whereas negative numbers indicate a lack of nutrients.

Figure 100 – Nutrient balances for P and N in the Weser-Ems region regarding the nutrient provision of organic fertilisers (production & import)

Impacts of nitrogen losses

Nitrate load in freshwater

Surface water - The surface water bodies present elevated nitrate levels and are thus characterised as significantly polluted (Keppner, et al., 2012). The elevated levels have an impact on the biological status\(^{221}\) of surface water bodies. In the Ems river basin the majority of the rivers (98 %) and lakes (90 %) did not reach a good ecological status or potential as aimed for in the WFD in 2014 (Hanusch, et al., 2014).

Nitrate leaching is high throughout the Ems catchment and the county of Oldenburg (Fuchs, et al., 2010). The dissolved N in precipitation water measured on open land has a level of 10 mg NO\(_3^-\)/L (Bellack, et al., 2013) and is therefore elevated in comparison to pristine areas (< 1 mg NO\(_3^-\)/L (Panno, et al., 2006)) but similar to other concentrations measured in Europe (Preziosi, et al., 2010). Nitrate is deposited on the soil (precipitation water is only one source for deposition) from where the leaching takes place. In forests, nitrate leaching is especially enhanced. Here, trees function like a comb. The ammonia

\(^{221}\) As defined by the WFD
compounds are adsorbed on the high surface of the trees and are washed off with the precipitation water. For all Lower-Saxon monitoring sites in forests, thresholds for tolerable nitrate input were exceeded (Fier, et al., 2013).

In the Weser-Ems region, many drinking water cooperatives have been set up to cope with the high nitrate levels. For the implementation of measures\(^{222}\), farmers receive a subsidy which is covered by the Lower Saxon water and nature conservation board (NLWKN) and the European Agricultural Fund for Rural Development (EAFRD).

**Groundwater** In lower-Saxony, 85% of the drinking water is produced from groundwater (Schültken, 2015). In two-thirds of the surface near groundwater stations, the nitrate levels surpass the threshold of 50 mg NO\(_3\)-/L (Bellack, et al., 2013). Reaching the objective of the EU Water Framework Directive of good chemical water quality is thus unlikely for many of the aquifers in the Weser-Ems region.

**Marine eutrophication**

For coastal waters, nitrate is the limiting factor for algae growth (Conley, 2000). For the delta regions of the Ems and Weser, the ecological status of coastal and transition waters is categorised as moderate or poor according to the WFD (Keppner, et al., 2012). In the Wadden Sea on the Lower Saxon coast, algae blooms occur frequently. An impact of the reduced oxygen content and light influence is the decreased coverage of seaweed in Lower Saxon marine areas (Umweltbundesamt, 2013). Seaweed beds are a natural habitat, nesting ground and food source for migratory birds (Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz – NLWKN, 2011). For many fish species, the ecosystem is needed as feed and spawning area. A reduction of sea grass beds also results in a threat to the bird and fish fauna.

Since 1985, a decrease in the phosphate load to the North Sea has been observed. This could not be observed for nitrates, which can partially be explained by the high residence time of nitrate in groundwater bodies in the region. To re-establish a natural nutrient level in the North Sea, the concentrations of the tributaries would need a drastic decline. For the Ems in order to reach the natural conditions, the load of N should decrease from 31,500 t/a (load in 2009) (Geschäftsstelle Ems; Ministerie van Verkeer en Waterstaat; Bezirksregierung Münster, 2009) to 1,400 t/a\(^{223}\) (Claussen, et al., 2007). For the Weser a reduction to 4,000 t/a needs to be achieved to reach the required good ecological status in the long term compared to the load in 2000 (Claussen, et al., 2007). It is estimated that 55.5% of the N load to the North Sea from the river Ems stems from organic and mineral fertilisers (Geschäftsstelle Ems; Ministerie van Verkeer en Waterstaat; Bezirksregierung Münster, 2009)\(^{224}\). Naturally, the N/P ratio is 7:1. The decreased concentration of phosphate with a stable concentration of nitrate causes the N/P ratio to change. This affects the occasional occurrence of algae species with a possible shift to toxin-producing algae like *Alexandrium* (Claussen, et al., 2007).

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\(^{222}\) E.g. counselling about fertiliser management, testing of organic manure and sampling of soil for the residual nutrient content in autumn, the time restricted application of organic manure, application of organic fertilisers with regards to the protection of freshwaters

\(^{223}\) The nutrient load to reach a good ecological status of the rivers was calculated with the model MONERIS (Behrendt, et al., 2003)

\(^{224}\) Office Ems; Ministry of Transport, Public Works and Water Management; District administration of Munster
Ammonia (NH₃) pollution in the air and soil acidification

In 2009, the total ammonia emissions for Lower Saxony were 135 000 t. 15% (20 250 t) of these were emitted from uncovered manure (Flessa, et al., 2012). As the Weser-Ems region is a territory of intensive livestock rearing, the ammonia concentrations are even elevated when measured during the manure application ban, when the stored manure emits ammonia (Köster, et al., 2012). The emission area shows evident patterns according to the location of emitters as portrayed in Figure 101. The area with the highest ammonia concentration in the map is characterised by a high density of poultry mast systems located around Vechta and Cloppenburg. The livestock density in these counties exceeds 2 LU/ha (Bäurle & Tamásy, 2010). The development of NH₃ emissions throughout the year shows elevated values for February and March, the time when the application ban for manure in accordance to the Nitrates Directive expires (Köster, et al., 2012). Generally, the deposition ranges between 23 and 37 kg N/ha a (Mohr & Dämmgen, 2013).

When ammonia is deposited on soils, it leads to a decreased pH in the soil. Acidification occurs when the soil does not have the capacity to neutralise the acids. In the Weser-Ems region, acidification occurs in forest regions, while liming is used on agricultural soils, thus preventing acidification. For the majority of the Lower Saxon forests, the input of acids through atmospheric deposition has exceeded the critical loads of the ecosystems (Meesenburg, et al., 2013), resulting in a decreased ecosystem health with the effect of higher sensitivity to disturbances like pests or extreme weather events.

Measurements in the region portrayed forest damages in forest sites close to confined animal feeding operation for poultry (Mohr & Dämmgen, 2013). They show damages on the vitality of the tree population in the forest marginal area. Furthermore, the study assessed a low C/N ratio in the humus layer, ranging
between 17 and 25, indicating enhanced nitrate leaching\textsuperscript{225}. Nitrate leaching was observed not only in proximity but also in areas adjacent to the mast systems. As the region is naturally nutrient-poor due to mainly sandy soils, nutrient inputs have a strong effect on the ecological systems. Species that are adapted to nutrient poor sites do not occur anymore in the saturated area under observation (Mohr, et al., 2011).

\textbf{Contribution to climate change}

The contribution of agricultural activities to climate change in Lower Saxony accounted to 28\% of the total GHG emissions in 2009. In particular, mineralisation processes of peat that take place in bog lands that are drained for agricultural activities account for the largest share of the GHG emissions in the regional agriculture (Flessa, et al., 2012). Emission of N\textsubscript{2}O from fertilised soils is the second largest contributor. For Lower Saxony in 2007, 90\% of N\textsubscript{2}O emissions stemmed from agriculture (Flessa, et al., 2012).

\textbf{Impacts of phosphorus losses}

\textit{Eutrophication of freshwater}

Phosphates reach the surface water via run-off. The input is highest where bog and marsh lands are agriculturally used which is the case in the Weser-Em\textsuperscript{s} region (Fier, et al., 2013). Phosphates develop in these soils mainly from the mineralisation of organic matter that under agricultural usage is exposed to oxygen in the air and can therefore be decomposed. Different soils in the area release different amounts of phosphorus: marsh lands release 40 – 50 kg P/km\textsuperscript{2}\text{*a}, whereas bog lands and other agriculturally used soils stay below 20 kg P/km\textsuperscript{2}\text{*a} (Bezirksregierung Weser-Em\textsuperscript{s}, Dezernat 502 Aurich, 2005).

The total amount of phosphate that is lost in the Em\textsuperscript{s} catchments amounts to 1 200 t per year, with 67\% coming from agriculture (Geschäftsstelle Em\textsuperscript{s}; Ministerie van Verkeer en Waterstaat; Bezirksregierung Münster, 2009).

Phosphate is the limiting growth factor for algae in freshwater ecosystems (Conley, 2000). Elevated levels can cause eutrophication. This has been observed for the shallow lakes in the region (e.g. Alfsee, Zwischenahner Meer). Their ecological functioning is especially sensitive to nutrient input. Algae blooms occur occasionally in the region, especially in long and warm summers. However, around 90\% of the surface waters in the Weser & Em\textsuperscript{s} catchments had a good chemical status\textsuperscript{226} according to the evaluation for the reporting period 2009 – 2014. In addition, 2\% of the surface water bodies in the Em

\textsuperscript{225} The lower the C/N ratio is the higher is the N content in the soil in general. Therefore, a bigger share of N is leached to the groundwater.

\textsuperscript{226} 88.6\% of the surface waters in the Em\textsuperscript{s} catchment as well as 90\% of the already assessed surface waters in the Weser catchment have good chemical water quality (Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten- und Naturschutz; Ministerie van Verkeer en Waterstaat; Bezirksregierung Münster, 2009; Flussgebietsgemeinschaft Weser, 2009).

\textsuperscript{227} With the Directive on Environmental Quality Standards becoming effective, none of the surface water bodies in the Em\textsuperscript{s} river basin portrays a good chemical status anymore, as most of them show pollution of mercury that is above the limit (Hanusch, et al., 2014). A similar exacerbation can be expected for the Weser river basin as well.
catchment and around 12% in the Weser catchment had a good ecological status according to the WFD (Hanusch, et al., 2014; Flussgebietsgemeinschaft Weser, 2009).

5.7.2 Causes of nutrient losses

Farming system and agricultural practices

The main causes for nutrient loss to the environment in the Weser-Ems region is the intensive fertilisation with organic fertilisers, the cultivation of marsh and bog lands and the ploughing of grasslands for maize production.

Intensive livestock production locally leading to an excess of manure produced compared to crop needs – As said above, the Weser Ems is a region with intensive livestock production. In the state of Lower Saxony, 253 000 t of nitrogen from livestock manure is produced annually. Especially in the Cloppenburg and Vechta counties, the allowable threshold level for N per ha is exceeded (Landwirtschaftskammer Niedersachsen, 2013) as a result of the high livestock density with more than 2 LU/ha (Bäurle & Tamásy, 2010). In counties such as Vechta and Cloppenburg, the amount of the produced organic fertilisers (manure and biogas digestate) and with that the supply of nitrogen surpasses the plant demand for nutrients (Landwirtschaftskammer Niedersachsen, 2013). As regards phosphorus, for eight out of 17 counties in the region, the supply exceeds the demand for the nutrient (Landwirtschaftskammer Niedersachsen, 2013). About 231 000 ha (which corresponds to 5% of the total area of Lower Saxony) would be additionally needed for spreading organic fertilisers in order to keep the supply and demand balanced.

High bioenergy production leading to increasing intensive energy crop production – In 2013, 580 biogas plants were installed in the region. In comparison to Lower Saxony, the installed electrical power is high (0.4 - 0.6 kW/ha (Murek, 2013)). The agricultural area that is needed for the biogas production is in Cloppenburg. With 23% of the agricultural area, it is the highest in the region (Murek, 2013). Nutrient inputs connected with maize for biogas production stem from the mineralisation of organic matter following ploughing of the grassland. Additionally, maize is intensively fertilised. Maize cultures are rarely accompanied by catch crops due to their high and dense growth. The excessive nutrients can therefore leach easily into the groundwater. A catch crop planted after the harvest in autumn is not sufficient in taking up the nutrients. Ploughing as a preparation for the catch crop also promotes the mineralisation process and thus supports nutrient leaching. With the establishment of subsidies for biogas production, the installed electrical power has almost tripled from 2005 to 2011 (Murek, 2013), the area of ploughed grassland has increased accordingly. It is mostly used for the production of maize that is used in biogas reactors. Furthermore, the production of biogas from maize yields higher revenues for farmers than the management of grassland. This resulted in rising rental prices for agricultural areas that in turn work against farmers who manage the land as grasslands and who are thus not able to pay higher rents. Furthermore, livestock manure is also used together with energy crops as input material to the biogas plants. The digestate resulting from the biogas plants has not been subject, to date, to a clear regulatory framework, thus being utilised as an organic fertiliser beyond the limits set by the Nitrates Directive for processed manure.

Overfertilisation due to lack of accurate knowledge and fear to lose production – When the amount of fertiliser is assessed, the amount of nutrients applied on the field is higher than the amount

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Community of the Weser river basin

Chamber of Agriculture of Lower-Saxony
that would be needed for plant growth, because losses are taken into account. Fertilisation beyond the
assessments is perceived by many farmers as a practice preventing possible losses in production.
However, fertilising beyond the crop requirements, beyond being unnecessary in terms of yield
production, also results in run-off and leaching into the environment.

**Lack of appropriate storage equipment** – Manure needs to be stored in proper storage vessels in
those periods in which it cannot be applied on land. This is a requirement of the Nitrates Directive.
Covering the manure reduces the evaporation of NH\(_3\). In Lower Saxony only storage systems that are
newly constructed and that exceed a capacity of 6 500 m\(^3\) fall under the regulations to cover the manure.
Existing and smaller storages do not require coverage. The amount of NH\(_3\) that could be prevented from
evaporation was estimated in 2012 as 0.8 kt NH\(_3\) from cattle manure and 2.3 kt NH\(_3\) from pig manure
per year for Lower Saxony (Flessa, et al., 2012).

**Cultivation of organic soils** – In Lower Saxony, the cultivation of organic soils has a long history that
dates back to the Middle Ages. Today, turf is exploited and sold as garden soil. For cultivating marsh
and bog lands, those have to be drained to keep the water table low. When the naturally water-saturated
soils are exposed to the oxygen in air, the mineralisation process starts and N\(_2\)O amongst other
greenhouse gases is emitted. Approximately 67 % of the Lower Saxon bog land, 235 000 ha, is used
for agricultural purposes. Due to the heavy use of these ecosystems, only 5 % remains in its natural
conditions (Flessa, et al., 2012).

**Ploughing of grassland and conversion to arable land** – Within the last two decades, the grassland
area has continuously decreased. Between 1990 and 2009, 30 % of the grassland area in Lower Saxony
was changed to another use, in particular arable land (Schramek, et al., 2012). One, but by far not the
only reason, is the establishment of subsidies for biogas production explained above. Converted
grassland in the region is mostly used for the production of maize that is used as feed or as substrate
in biogas reactors (Schramek, et al., 2012). Ploughing up grassland results in increased leaching of
nutrients and run-off conditions.

**Environmental conditions**

Natural factors influence the effects of farming practices and especially fertiliser inputs.

**Leaching enhanced by soil texture** – The soil type is important for the correct calculation of the nutrient
demand. In the southern part of the region (in the Ems catchment south of Hamm), mainly Cambisols
and Albeluvisols occur. These soils are a fairly good basis for arable activities. However, the use of
fertiliser is essential on these soils. The soil texture is sandy with shares of silt and loam. Due to the
high sand content, they have a high permeability which makes them susceptible to nutrient leaching and
dry periods. In the Central Ems catchment and the Western Weser catchment, Podsols occur which are
sandy and nutrient poor. Nevertheless, with an input of fertiliser, they present a good basis for
agricultural activities. They have similar susceptibilities as described before.

**Organic and groundwater-influenced soils** – In the north of the region, groundwater tables are high
which makes the soils difficult to use for agricultural purposes. These marshlands need to be drained to
make the land arable. It is fertile and in use for agricultural purposes. Bog-soils occur in the south of
Oldenburg and Bremen which have often been drained in the past, resulting in high nutrient leaching in
the present. These locations are very vulnerable to nitrate leaching after ploughing. Nitrogen release
after ploughing has to be considered at the fertiliser planning for years.

**Leaching through abundant rainfall** – The study region belongs to the maritime climate of Western
Europe with an Atlantic-influenced climate. This is expressed by relatively low temperature variations
throughout the year and a positive water balance. In times of abundant rainfall, leaching is enhanced.
In terms of climate change, no effect on the water balance is expected whereas the temperatures will rise (Niedersächsisches Ministerium für Umwelt, Energie und Klimaschutz, 2014). Thus, the nutrient loss through leaching is not likely to increase. However, higher temperatures could increase the mineralisation rate, which might have an effect on available nutrients and thus losses from leaching.

5.7.3 Costs of the environmental and health effects

5.7.3.1 Social economic description of the study area

The study region Weser-Ems had a total population of 2,446,345 inhabitants in 2014. It represented 31.4% of the population of Lower Saxony and 3% of the total German population (Eurostat, 2015d). With a GDP of €253.6 billion in 2014, Lower Saxony is the fifth strongest economic region in Germany and creates 8.7% of the German GDP (Arbeitskreis Volkswirtschaftliche Gesamtrechnung der Länder, 2011). In 2012, the GDP of the Weser-Ems region was €74,860 million, representing 31% of the GDP of the Lower-Saxon territory (Eurostat, 2015f).

The biggest share of the Lower Saxon and the Weser-Ems gross value added is provided by the manufacturing industry (32% in 2014) (Data Portal of Germany (GovData), 2015). Volkswagen and Salzgitter Steel are the most important enterprises that are included in this category. After automobile production, the food industry is the second most important sub-sector, with €25.6 billion overall turnover in 2009 (out of that, €8.1 billion from the meat industry alone) and 67,000 employees (Flessa, et al., 2012). Following are the service sector with financing, renting and company service providers (24%), public and private service providers (24%), trade, traffic, catering and hotel industry (18%) and agriculture (1.5%). The tourism sector, in which 340,000 people work, is affected by the nutrient surplus due to algae blooms resulting from eutrophication in the North Sea. Tourism is an important economic sector for the region creating a value of €15 billion per year (Niedersächsisches Ministerium für Wirtschaft, Arbeit und Verkehr, 2014). In 2012, agriculture represented 2.3% of the gross added value in Weser-Ems (Eurostat, 2015c).

5.7.3.2 Review of economic damages

In order to better understand the economic damages associated with nutrient surplus in the region, a wide range of literature was reviewed. Based on the classification of economic damages caused by environmental impacts presented in Annex 14, costs found in the literature for each damage category are presented below along with explanations on how these costs have been estimated.

It should be noted that all cost data reported in this section are taken from the primary studies found in the literature, and expressed in the value of the year when the study was conducted. To date, no specific cost assessments for the impacts of nutrient surplus have been carried out in the region. As such the following section focuses on general costs assessment, but also relevant initiatives in Lower Saxony, which allow for statements on costs.

Clean up and restoration costs (CRC)

In the Weser-Ems region, the elevated nitrate levels in the groundwater bodies pose a threat to drinking water quality and to human and ecosystem health. For the reduction of nitrate in drinking water, rather than
than end-of-pipe technologies to remove nitrate from waters, prevention has been often identified by water utilities as the best way to address the problem.

**Preventive measures to reduce nutrients concentration in water**

Regarding the management of nitrate in the groundwater, it is worth mentioning two main policy tools. In Germany, the Nitrates Action Programme is mainly implemented by the “Fertiliser Ordinance”. The second tool relates to the Rural Development Programmes (RDP) that all Member States are obliged to establish. Within this instrument, “groundwater cooperation schemes” were established. These schemes consist of a contract between farmers and water users. The farmers receive compensation payments if they implement certain measures that aim to reduce the nitrate levels in the groundwater

Those payments are financed by the water users who pay an addition to their water bill. Currently, the system consists of two pillars. The first pillar contains extension services for farmers, which is treated under the Policy Action Costs section below. The second pillar contains voluntary activities that are carried out by the farmers. These activities include e.g. precision agriculture or fertilisation according to residual N in autumn (see Table 52). Those are reimbursed for the losses and additional expenditures and work within the cooperation (financed in the RDP period 2014 – 2020). Special measures in the study region comprise arrangements for the time-restricted application of organic manure, application of organic fertilisers with regards to the protection of freshwaters, as well as the field nutrient balances (Hartung, et al., 2011). As these measures were established due to the high nitrate level in surface near groundwater bodies, the costs for the implementation of voluntary activities will be classified in this study as clean-up and restoration costs.

**N removal**

The end-of-pipe technologies comprise physical-technical methods such as ion exchange, nanofiltration, reverse osmosis and electrodialysis and denitrification as a biological technique (Holländer, et al., 2008). Information about the extent of implementation in the study region could not be retrieved. Exclusively denitrification removes the nitrogen from the water and releases nitrogen gas to the atmosphere. All other techniques concentrate the nitrate either in a fraction of the water or bound to electrodes. Those require proper waste treatment.

When comparing the expenditures for cleaning per m³ of drinking water, it appears costly to apply end-of-pipe technologies (see Figure 102). Regarding environmental effects, preventive measures are preferred over end-of-pipe technologies as it affects not only the pumped raw water but the water resources as a whole.

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232 The farmers agree to implement measures such as demand adapted fertilisation, extended application bans for nitrate containing fertilisers, or relinquishment of ploughing after the harvest.
Regarding the costs involved with the mitigation of gaseous emissions, there are different pathways for which mitigation measures can be considered. Firstly, NH$_3$ degasses from uncovered storage of manure. Therefore, among the options for ammonia emissions reduction, storage capacity for organic fertiliser can be covered. Secondly, NH$_3$ degasses from fields when organic fertiliser is applied but have not been incorporated. Thirdly, intensively fertilised soils emit NO$_x$ and N$_2$O. Fourthly, N$_2$O is emitted when soils rich in organic matter (e.g. peat and marsh land) is drained for agricultural use. For all these pathways, several measures presented in Table 52 can be implemented in order to mitigate the effect. Due to the input of atmospheric acids (ammonia that reacts with water to ammonium), ecosystems suffer acidification. On agricultural land and commercial forests, liming is applied to counter the effect.
Table 52 – Clean up and restoration costs found in literature

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Costs (€)</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure for clean-up and restoration purposes</td>
<td>Voluntary activities</td>
<td>€ 4 275 670</td>
<td>Unconsolidated rock soils west of the Weser Weser-Ems without the area surrounding Osnabrück, with the county of Diepholz</td>
<td>2009</td>
</tr>
<tr>
<td>GHG mitigation costs for rewetting of marsh and bog lands and abandonment</td>
<td></td>
<td>20 – 73 €/t CO₂-eq</td>
<td>Lower Saxony</td>
<td>2011</td>
</tr>
<tr>
<td>Adapted application of manure and direct incorporation</td>
<td></td>
<td>38 - 747 €/t CO₂-eq</td>
<td>Lower Saxony</td>
<td>2011</td>
</tr>
<tr>
<td>Increasing covered storage capacity for organic fertilisers (beyond legislation requirements)</td>
<td></td>
<td>63 €/t CO₂-eq</td>
<td>Lower Saxony</td>
<td>2011</td>
</tr>
<tr>
<td>Fertilisation according to residual N in autumn</td>
<td></td>
<td>73 €/t CO₂-eq (1 278 €/t N)</td>
<td>Lower Saxony</td>
<td>2011</td>
</tr>
<tr>
<td>Liming on 1 132 005 ha grassland (2 t/ha every 3 years)*)</td>
<td></td>
<td>18 866 750 €/a 16.7 €/ha per yr.</td>
<td>Lower Saxony</td>
<td>2014</td>
</tr>
<tr>
<td>Liming on 1 726 594 ha cultivated land* (3 t/ha every 3 years)</td>
<td></td>
<td>43 164 850 €/a 25 €/ha per yr.</td>
<td>Lower Saxony</td>
<td>2014</td>
</tr>
<tr>
<td>Liming of 1 042 100 ha forest** (3 t/ha every 7 years)</td>
<td></td>
<td>11 165 357 €/a 10.7 €/ha per yr.</td>
<td>Lower Saxony</td>
<td>2014</td>
</tr>
<tr>
<td>N removal</td>
<td></td>
<td>5 – 125 €/m³ of drinking water depending on the technology</td>
<td>Germany</td>
<td>2008</td>
</tr>
</tbody>
</table>

Use value damages (UVD)

Elevated nitrate levels in the environment have far-reaching consequences for the environment, which eventually threaten the economic value added due to effects on human health, the natural resources, tourism and biodiversity. For this value damage category, no quantitative data was found in the literature.

Several reports in the media could be found that highlight the blue algae growth in the oligothrophic lake Dümmen in the region that is in focus of local and regional tourism. The occurrence of blue algae in the summer months threatens the revenues of the local tourism services (Schrammar, 2011). Due to the occurring lack of oxygen in the inland and marine waters, organisms living close to the ground as well as the fish fauna are affected (Umweltbundesamt, 2010). The latter also affects fisheries and may induce losses of income.

Passive use value damages (PUVD)

Ozone that is created from NOₓ is a risk for human health as it irritates the respiratory organs. The creation of Particulate Matter from NOₓ and NH₃ has similar health effects. These effects cause costs of
€32/kg N-NOₓ and €22/kg N-NH₃ including medical treatment, wage and productivity losses as well as the willingness to pay for avoiding pain in Germany (Brink & van Grinsven, 2011). The pollution with NH₃ and NOₓ has several consequences on the environment. An additional nutrient supply in natural ecosystems can lead to the shift of habitat conditions. Species that are adapted to nutrient poor habitats will be replaced by nutrient rich adapted species. The critical load of atmospheric acids on forests is exceeded for many forests located in Lower Saxony. The composition of forest species is already altered (Mohr, et al., 2011). The costs of the acidification effect is accounted to €1,800/t NOₓ based on the willingness to pay for the damages by EU citizens (Schwermer, 2007). Furthermore, NOₓ is the predecessor for tropospheric ozone, which can be harmful to plants.

Nitrous oxide emissions possibly leading to climate change also pose various threats on the natural environment. With elevated temperatures and a changing hydrological regime, living conditions for animal and plant species change. Ecosystems can cope to a certain degree with these changes, but are more susceptible to disturbances like extreme weather events (heavy rain, storm, heat waves and dry spells). Alien species can easily invade such frail ecosystems. The costs due to biodiversity loss are difficult to estimate, as neither the effect nor the extent of the effect is simple to forecast.

**Policy action costs (PAC)**

As mentioned under the Clean-Up Costs section, water cooperation schemes are in place under the Lower Saxony RDP. The cooperation’s first pillar contains extension services for farmers, counselling about fertiliser management, testing of organic manure and sampling of soil for the residual nutrient content in autumn. The service is connected to payments by the Lower Saxony Water Management, Coastal Defence and Nature Conservation Agency (NLWKN) with cofinancing of the European Agricultural Fund for Rural Development (EAFRD).

The subsidies for integrated RDP finances training and information campaigns amongst others.

**Table 53 – Policy actions costs found in literature**

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (€)</th>
<th>Costs in ct/m³ of pumped raw water</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Awareness raising &amp; counselling</td>
<td>Extension services for farmers</td>
<td>1,846,062 €</td>
<td>2.1</td>
<td>Unconsolidated rock soils west of the Weser Weser-Em's without the area surrounding Osnabrück, with the county of Diepholz</td>
<td>2009</td>
</tr>
</tbody>
</table>
5.7.4 Good practices to reduce nutrient losses at farm level

The good practices proposed for Weser-Ems mainly aim at reducing nitrate leaching to groundwater and ammonia emissions. They especially apply for the counties Vechta and Cloppenburg where several intensive indoor livestock rearing are located and where the amount of manure produced exceed the crops’ needs. The first set of measures presented in this section aims at limiting the amount of nutrient produced or applied on field through the adjustment of the feed quantities, by raising awareness among farmers through support of fertiliser management, using the most suited application technique and converting arable land to unfertilised grassland in areas at risk. The second set of measure aims to limit or reduce nutrient losses by covering manure during storage and spreading at the right timing. The third set of measures concerns manure recycling by encouraging slurry separation would help at using manure as fertiliser and valorise the solid fraction if it is used to produce energy though anaerobic digestion, the rewetting of organic soils and the establishment of paludicultures.

5.7.4.1 What has already been done in the region?

The Fertilizer Ordinance is the main piece of legislation implementing the Nitrates Directive and includes several requirements relating to N and P fertilisation, quantities, timings and modalities. In the region, several measures (as part of the voluntary measures outlined in the Rural Development Programme 2014-2020) are already implemented, including e.g. measures on crop land as the use of cover crops and the reduction of bare fallow as well as avoiding legumes as cover crops. Furthermore, measures related to feeding practices such as the adjustment of the quantity of feed and the addition of amino-acids as well as phytase are already well implemented in the region (Flessa, et al., 2012). In Lower Saxony, the majority of animals are already fed with fodder with reduced raw protein and phosphate content. This additional measure is organised between the Chamber of Agriculture Lower Saxony, the administrative district and the farmer (Landwirtschaftskammer Niedersachsen, 2014).

Farm advisory services address inter alia site specific fertilisation with regards to crop needs and environmental conditions is applied for farms covering around 300 000 ha in Lower Saxony. Further advisory services are focusing on the so-called water cooperations, which are established to manage drinking water quality. Those are voluntary actions established between water utilities and farmers to lower the nitrate levels in the aquifers. The advice focuses on voluntary actions like catch-crops, undersown grass, reduced tillage, reduced nitrogen fertilisation, changing crop rotation as well as result oriented compensation. The contracts are established with a run time of one year. Cooperation schemes between water and farming community were developed in Lower Saxony already from 2005.

A payment-by-result approach aiming to reduce N surpluses was tested within the EU Life project WAgricO in 2006-2007 which fed into the agri-environmental measures starting in 2010. One objective of the project was to select and develop measures with good ecologic and economic efficiency to achieve the goals of the Water Framework Directive (WFD). Measures with positive impacts of the mineral N content in autumn and N balances and good cost-efficiency were selected in the project. The implementation at 50 model farms in three pilot areas in Lower Saxony was accompanied with a participatory approach. Advisory services and trainings were provided to the participants. Discussion groups and workshops at local level, of the project members and of experts were conducted.

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233 Farm advisory systems have been introduced together with the cross-compliance mechanism in 2003. The Rural Development Programme supports farmers to make use of advisory services and supports Member States in setting up new farm advisory services where needed, but also supports the implementation of the WFD.

234 http://www.wagrico.org
innovative aspect was a result oriented measure to reward for improved N-efficiency. However, during the project it was shown that this measure is not yet suitable for practical application due to difficulties with data collection and management (high data demand e.g. soil samples of different soil horizons, water samples or nutrient balanced on farm or plot level). Further activities were considered to be necessary to develop reliable impact indicators and control mechanisms for the result-oriented approach in Lower Saxony (Frelih-Larsen, et al., 2014).

Lower Saxony is the federal state with the highest installed electrical power of biogas plants and thus biogas is in general widely established. The average installed electrical power of the biogas plants in Lower Saxony is with 396 kW high (Nds ML & Nds MU, 2012). However, the German Renewable Energy Act mainly supports for larger plants as those installed in Lower Saxony energy production from renewable sources, i.e. maize (Dağaşan, et al., 2014). Due to the high production of manure, Lower Saxony has the potential to generate more energy from manure and less from other feedstocks, thus increainge the sustainability of energy production from biogas. Moreover, the digestate resulting from the anaerobic digestion can be applied as a fertiliser and has generally a higher content of mineralised nitrogen than raw manure (Børgeesen & al, 2013). Under the current circumstances, anaerobic digestion in biogas plants is not particularly attractive to farmers, as the high electrical power of most of the installed plants receives subsidies only when using renewable sources. Despite the big potential for further emission reduction during the digestion process and the storage of the fermentation substrate, the solution would not yield sufficient acceptability among farmers and is therefore not presented as good practice.

In 2012, several agricultural institutions in Lower Saxony established a manure transfer system. From the period 07-2012 until 06-2013, 1.8 million tonnes of organic fertiliser were traded from the Weser-Em region to other regions in Lower Saxony (Jansen-Minßen, 2014a). The Chamber of Agriculture in Lower Saxony summarises that this measure still has potential to further reduce the surplus. To exploi this potential, regulations would need to more strictly treat nutrient surpluses. Due to the advanced establishment of the manure stock exchange in Lower Saxony, we will not describe in detail the measure “transferring manure from farms with a surplus to farms with a shortage for substitution of chemical fertilisers”. Another measure that seemed suitable and is a requirement for transferring manure is the keeping of indoor livestock. As the region already hosts intensive indoor livestock rearing, we consider the potential already exhausted.

### 5.7.4.2 Good practices to reduce the nutrient losses in livestock production

#### Use feeding practices to reduce the amount of nutrients produced

The adjustment of the quantity of animal feed to the needs of the animals and the use of feeding practices to increase the assimilation of nutrients contained in feed by animals are effective measures to reduce the nutrient content in manure. An assessment of reduction potential for greenhouse gas emission of the agricultural sector in Lower Saxony found that measures aiming at adjusting sugar content and the addition of amino-acids as well as phytase to increase the digestibility of nutrients provide good results in terms of NH$_3$ emission reduction and P content reduction.

For the most effective application of this measure, the farmers need to group the animals according to sex, age and production stage as feed requirements differ (Carter, et al., 2012b; Spiehs, 2005). When keeping animals separated, feed can better be adjusted. The farmers will need to regularly adjust feeders, and provide regular maintenance of the feeders, bunks and drinking troughs (Spiehs, 2005).

This measure requires a moderate level of know-how by the farmers. Firstly, an appropriate feeder design for age and type of animal has to be set up (Spiehs, 2005). This also requires knowledge on the availability of nutrients in feed ingredients on the base of which diets have to be formulated (Sutton &
In Lower Saxony, the demand-based feeding is quite advanced. The Agricultural Chamber Lower Saxony gives, for example, recommendations for fodder reduced in raw protein and phosphate (RAM). The success of this approach can be enhanced with the recommendations for ‘highly (nutrient) reduced fodder’ by the German Agricultural Association (DLG, 2014).

Furthermore, sufficient knowledge is needed for the application of the measure in order to ensure that livestock production will remain constant (Knowlton, et al., 2004). For example due to variable compositions of feed, routine laboratory feed analyses and a ration formulation program are necessary to adjust diets and maintain minimum nutrient excretions (Sutton & Lander, 2003). For Weser-Ems, this is particularly relevant as the livestock farmers retrieve their fodder in large shares from the regional enterprises producing compound feed. This type of feed amongst grains also contains residues from the food industry. Therefore, the fodder that the farmer purchases changes in composition regularly (Landwirtschaftskammer Niedersachsen, 2014). For the farmers, the measure will potentially lead to an increase in revenues due to the achieved savings for fodder. The costs for the regular analyses of fodder will partly decrease the gain.

While measure such as the adjustment of the quantity of feed or the use of additives are already well implemented in the region (Flessa, et al., 2012), the stakeholder consultation of this project revealed a low acceptance for the measures related to multiphase feeding and the adjustment of protein content. This is because the implementation costs for the farmers are high and the implementation is not incentivised through revenue gains or funding programmes. In the future, this might change due to changing incentives through amendments to the Fertilisation Ordinance. The amendments might include the implementation of farm specific reporting of nutrients and the extension of ban-periods for fertilisation, as the draft of the Federal Ministry for Nutrition and Agriculture suggests (Bundesministerium für Ernährung und Landwirtschaft, 2014). It was estimated that the implementation potential for the measure in the region is high (Flessa, et al., 2012; Jansen-Minßen, 2014). Generally, more information should be dispersed amongst farmers to exploit the margin between excessive and sufficient fodder provision. The measure is not and was not promoted in the RDP in the period 2014 - 2020 and the last funding period 2007 - 2013.

Cover manure during storage

Manure covering decreases the surface area where emissions can take place. By reducing the manure/air interface, ammonia emissions can be prevented. The regulations in Germany for storage facilities for slurry and solid manure only require coverage for new facilities exceeding 6 500 m³ storage capacity. In the study region, many smaller or existing facilities do not require coverage and are sources of ammonia emissions, as well as NOx, particular matter and odour emissions.

For Lower Saxony, the amount of abated N in the environment was calculated to 3.1 kt NH₃ for pig and cattle slurry and 36 t N₂O (Flessa, et al., 2012). The potential for the region is estimated as very high as only 41% of the farms have storage units for manure (Eurostat, 2010b). Organic fertiliser production amounts to 25 million tons in the region (Landwirtschaftskammer Niedersachsen, 2013).

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235 Federal Ministry of Nutrition and Agriculture of Lower-Saxony

236 The numbers are calculated on basis of the current emissions from open pit outdoor manure storage. The numbers present the case that all manure storage facilities that are currently uncovered are covered with foil.

237 Calculated from “Manure storage and treatment facilities: number of farms and areas by economic size (SO in euros), livestock units (LSU) of farm and NUTS 2 regions”.
Therefore, the upgrade of existing manure storage facilities with coverage is proposed as good practice for the region.

**Improving manure processing: slurry separation**

Treatment of manure with separators results in a solid portion that is rich in phosphorus as well as a liquid portion with a high amount of ammonium. The solid portion has a high potential for energy production in biogas plants. In order to reduce the nutrient surplus in high livestock density areas, the solid substrate could be traded in the regional manure stock exchange and be used as digestion substrate in the northern part of the region. Treatment with manure separators has been tested in Lower-Saxony, aimed at producing N-P-K fertiliser from the manure. The Agricultural Chamber Lower-Saxony estimates this measure as relevant for the region (Jansen-Minßen, 2014b).

**5.7.4.3 Good practices to reduce the nutrient losses in crop production**

In Weser-Ems, the high amount of manure available in the region can exceed crop needs in counties such as Vechta and Cloppenburg. Given this point, two types of measures should be implemented to reduce the amount of fertilisers (inorganic and organic) applied: adjusting the amount of fertiliser applied to crops needs and finding application methods on field to apply manure in a more targeted way.

**Improve fertilisation management plans for all farms**

It is common practice that the amount of fertilisers applied to the crops is overestimated by the farmers as the calculations accounts also for nutrient losses. In order to not compromise the amount of harvest, farmers are used to apply an amount of nutrient higher than the nutrient amount demanded by the crops. The measures focusing on addressing excessive fertilisation therefore aim at raising awareness of farmers through support of fertiliser management.

Advisory services are provided to improve fertilisation based on site specifics. Water cooperatives are established in 6% of the Lower Saxony. In total 30% of the region is advised by the Agricultural Chamber of Lower Saxony in the frame of the WFD and the potential to further promote the improvement of site specific fertilisation management planning is apparent and emphasised by the Agricultural Chamber of Lower Saxony (Jansen-Minßen, 2014a).

Extending the preparation of fertiliser management plans to the whole territory allows for optimising the amount of nutrients applied to the conditions of the land (soil type, crop demand and remaining nutrients). The measure includes the calculation of N, P and K leftovers and the resulting fertiliser amount for the next growth period. The following options should be taken into account:

- Crop nutrient requirements for the expected yield under the given environmental circumstances;
- Additional nutrients available in the soil due to mineralisation of crop residues and organic fertilisers;
- Lime content (or pH) and soil organic matter content;
- Nutrient supply through irrigation and further site management (in addition to fertilisation);
- Conditions that influence nutrient availability;
- Measurements of soil samples to assess the nutrient residue.

Furthermore, a timed application of fertiliser with regards to the crop needs and ability to uptake needs to be considered. Here, it is important that no fertiliser is applied if the soil is saturated with water, flooded, frozen or covered with snow. Subsequent and interim crops may only be supplied with the amount of nitrogen that responds to the current nitrogen demand of the crop. Here the limits stated by the Fertiliser Ordinance should be taken into account (limits of 40 kg ammonium-N and 80 kg total-N per hectare). This takes account of the mineralisation processes after the harvest of the main crop. After
harvesting maize, rape, potatoes, sugar beets, vegetables or legumes, the nutrient demand of the subsequent crop can be covered with the mineralised N from residues (Jongebloed, 2013).

The calculations are based on soil samples in autumn (Nmin), which also serve as a monitoring element of this measure. The application is fine-tuned according to the crop type and the local conditions and can go beyond the mandatory requirement set out in legislation.

The potential for Lower Saxony and thus the region is estimated as high.

**Spread manure at the right timing considering the stage of crop development (adequate timing), use of Application Timing Management Systems (ATMS)**

The measure is included in the compliance with water cooperation in Lower Saxony which is partly paid by the European Agricultural Fund for Rural Development (EAFDR). As the water cooperatives and advisory services by the Agricultural Chamber Lower Saxony in the frame of the WFD are not established for the whole region, we estimate a high nutrient saving potential for the regions outside of water cooperation agreements. The measure partly resembles suggestions according to the Fertilisation Act, which contains many not options that are not mandatory. In combination with the manure stock exchange as established in Lower Saxony, the manure could be sold and would bring additional revenue.

**Use the most suited application techniques**

The volatilisation of nitrogen contained in the manure leads to NH₃ emissions during field application. Therefore it is important to use application techniques such as band application or injection that decrease the surface area of manure in contact with air and thus decrease potential for volatilisation and subsequent NH₃ emissions. Band spreaders drag perforated hoses behind them from which slurry is applied close to the ground. Injection systems slit the soil open and inject the fertiliser in different depths. For all these possible techniques, specific equipment is necessary and needs to be rented or purchased. Band spreading costs 1.3 €/m³ slurry more than regular broadcast spreading including the renting of the equipment. For the injection method, the price premium amounts to 2 €/m³ slurry. Some of the techniques have moderate uptake due to costs of new machinery, but high fertiliser N prices are encouraging the uptake particularly using contractors. Also the operation costs (costs of operating the farm machinery and the time required for field operations) are more than the broadcast spreading.

A similar measure is promoted within the agro-environmental measures for the new funding period 2014-2020: the CULTAN technique. The CULTAN technique also applies fertiliser by injection into the soil or bandsanspreiding. The difference is that CULTAN, which stands for *Controlled Uptake Long Term Ammonia Nutrition*, only uses ammonium fertiliser (mineral fertiliser). When applying the fertiliser so called depots are established in the soil or on the surface of which the plant can retrieve ammonium for a long time. As ammonium is not being leached, it is feasible to store it on-site. Slurry can also be used

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238 please see a description of ATMS on page 95

239 Investment costs (Huijsmans, et al., 2004): Slurry tanker: € 11 000 to € 16 000 (depend on size), trailing hose: € 10 500 + tractor costs: € 43 500 to € 56 500; Trailing foot: € 6 800 to € 9 000 + tractor costs: € 43 500 to € 56 500; Shallow injector: € 7 700 to € 11 500 + tractor costs: € 56 500 to € 66 250; Arable land injector € 5 500 to € 9 500 +tractor costs: € 56 500 to € 66 250.

240 Operating costs in €/m³ slurry (Huijsmans, et al., 2004): Broadcast spreading: 2.05 to 10.13, Trailing hose: 2.98 to 15.71, Trailing foot: 2.73 to 15.72, Shallow injection: 3.12 to 17.93, Arable land injection: 2.93 to 16.87
for the CULTAN technique, when it is processed with the addition of ammonium sulphate. This is needed to increase the ammonium content in the slurry. Otherwise, the nutrient content is too low and fertilisation needs to be repeated. It shows the great potential for this measure as well as for its application for slurry. A high share of farms in the region contract companies to spread the manure on fields. These enterprises have adopted advanced application techniques already (Jansen-Minßen, 2014b).

5.7.4.4 Good practices related to land conversion

Lower Saxony is a region in which organic soils have been used for agricultural purposes for centuries. Organic soils are found in the coastal area, the delta region, and in Leer county. In order to use these soils, they have to be drained. Drainage and the resulting exposure to oxygen in the air sets off mineralisation causing greenhouse gas emissions and nutrient leaching to the water bodies. The objective of this set of measures is therefore to reduce the emission of GHGs and nutrient leaching from peat lands.

In total 234,800 ha of peat lands are located in Lower Saxony. Due to the heavy use of these ecosystems, only 5% have been left in their natural condition. Therefore, it is of utmost importance to avoid the further drainage of wetlands. The restoration of the natural condition in the Lower Saxon bogs and raised bogs would reduce the amount of greenhouse gas emissions of 74% and 78% respectively (Flessa, et al., 2012).

Rewet organic soils

Regarding the large number of degraded wetlands in Lower Saxony, the mitigation potential of this measure is large. Depending on the degree of restoration, the costs for rewetting may involve high initial investments for the use of machinery and labour, dismantling of drainage or excavation, or may be low cost (Underwood, et al., 2013). Constructed wetlands require maintenance and are highly site-dependent and thus vary greatly.

Establish paludicultures on rewetted peat lands

The conventional rewetting of peat lands often is established together with nature conservation attempts. This requires the termination of agricultural activities on the land. The measure of establishing paludicultures on these rewetted soils promises a new form of incentive for the farmer as the land can still produce revenues.

The measure reaches similar environmental objectives as rewetting organic soils. It has to be combined with this measure and presents additional revenue for the farmer. The measure has co-benefits for nature conservation through the restoration of habitat for rare bird and plant species that occur in the threatened ecosystem. So far, the measure is implemented in pilot projects to demonstrate its feasibility and economical viability e.g. in the Hankhausen bog area in Lower Saxony, Brandenburg and Mecklenburg-Western Pomerania, the area close to Lake Constance or international examples like Belarus and Poland (Wichtmann, et al., 2010). The sites where those pilot projects are implemented are former turf mining sites or border areas of existing bog lands that can be recultivated. To reach the full potential, the measure needs stronger political support e.g. through the promotion in the RDP or the German Renewable Energy Act. The European Parliament and Council stated in a decision to support paludiculture as it provides a possibility to mitigate GHG emissions (European Parliament & European Council, 21 May 2013), which shows that the measure has reached political acceptance.

Convert arable to grassland in areas of high soil risk

Organic soils are prone to nutrient leaching and erosion. Land use change from arable production to grassland converts the land to a less intensive use and reduces the potential for nutrient losses.
Although this is less favourable than rewetting peat lands, it is a first step to allow for elevated water tables, and delivers still considerable improvement for the nutrient loss situation.

To reduce the leaching and erosion risk, the grassland sward needs to be seeded at an appropriate density rate (20 kg per ha generally recommended (RSBP, 2008) and rolled afterwards in order to ensure productivity, prevent weed establishment and encourage consolidation of seeds (Hybu Cig Cymru, 2008).

The costs for the farmer are high, as the concerned area is no longer used for production. Therefore, an acceptance without compensation is unlikely. The measure is financed under the agri-environmental-climate measures within the RDP in the period 2007 - 2013 as well as in the current financing period and is part of the voluntary water cooperatives. The latter financing is more attractive to farmers as they receive higher equalisation payments. The potential for the study region is high due to the large area of degraded wetlands as outlined above. The full potential of this measure could probably be achieved if the payments in the frame of the RDP would be increased.
5.8 Wielkopolskie (PL)\textsuperscript{241}

Box 11 – Wielkopolskie case study - In brief

The Wielkopolskie region faces a number of pressures from nutrient surpluses, largely due to agricultural production as well as industrial and municipal waste. Wielkopolskie is affected by a high nitrogen and phosphorus load in freshwater. P load leads to the eutrophication of lakes with visible algae blooms which affect biodiversity as well as recreational and tourism activities. N and P loads from Wielkopolskie are discharged into the Szczecin lagoon and the Baltic Sea, leading to eutrophication of the lagoon (P limiting in spring and N limiting in summer) and of the sea. It is estimated that 26% and 39% of the Polish monitoring sites for transitional and coastal waters were eutrophic or hypertrophic respectively in 2008-2011. Wielkopolskie is also affected by high ammonia emissions that may lead to acidification, in particular in forests.

Restructuring and intensification of farms, large and concentrated livestock production with poor farm infrastructure, lack of implementation and accuracy of fertilisation management plans, very few examples of collective action between farmers and dense drainage networks have contributed to nutrient losses in the region. Wielkopolskie’s light, acidic, and highly erodible soil and the hydrological system of the lagoon also contributes to nutrient losses, causing regional as well as transboundary effects.

The unit abatement costs for nutrient (N and P) reduction in Poland (corresponding to agricultural measures such as the reduction in the use of fertilisers, the reduction of livestock and the spreading time of manure) have been estimated between € 12.5 to € 223.6 per kg of phosphorus reduced and between € 1.32 to € 11.1 per kg of nitrogen reduced by farmers.

In Wielkopolskie, the identified good practices include measures aiming at decreasing the local source of pollution by extending and improving fertilisation management planning for all agricultural sites, improving application techniques or processing manure to ease its transfer when the quantity of manure produced locally exceeds the carrying capacity of the farm. Another type of good practice relates to the reduction of the amount of nutrients lost during storage by putting in place an impermeable floor, covering the solid manure stored in heaps or increasing the height of manure heaps and by cooling slurry. Composting manure would also help at reducing the volume of manure stored and thus avoid exceeding the manure storage capacity of the farm that may lead to the application of manure at an inappropriate timing. Good practices can also help at avoiding and reducing nutrient losses when manure is applied through measures such as using the most suitable application technique and the construction of sedimentation ponds. Lastly, measures aiming at increasing the soil quality such as the use of conservation tillage, the incorporation of straw and soil coverage (in particular with nitrogen fixing crops) would help at decreasing the amount of fertiliser that farmers need to buy.

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Wielkopolskie is a region located in western central Poland (see Figure 103) where oceanic and continental influences meet, resulting in a mixed climate. The region is part of the Oder river basin. The Warta, one of the largest tributary of the Oder River, passes through the Wielkopolskie region. The Oder river drains into the Szczecin lagoon and thus into the coastal waters of the Baltic Sea (Korth, et al., 2013).

Figure 103 – Geographical location of Wielkopolskie

Source: Own compilation

Wielkopolskie has one of the most extensive agricultural production systems among the regions under study. However, it is one of the regions with the most intensive production systems\(^{242}\) in Poland (Mizgajski, 2007; Central Statistical Office of Poland, 2015): Wielkopolskie is among the Polish regions with the highest average farm size, the highest mineral fertiliser consumption (N, P, K, Ca), the highest livestock density and the highest cereal yield in 2013 (Central Statistical Office of Poland, 2015). Wielkopolskie is the leading agricultural region in Poland. In 2013, agriculture production in the region represented 16.9 % of the value of the national production (Central Statistical Office of Poland, 2015)\(^{243}\) and about 1.7 million ha were dedicated to agriculture, representing 57 % of the total area of the region (Statistical Office in Poznan, 2014; Eurostat, 2014a). Arable land accounted for 81 % of the total agricultural land in Wielkopolskie with 14.2 million ha in 2013 (Statistical Office in Poznan, 2014). Cereals are the prevailing products. 73 % of the arable land was dedicated to cereal production in 2013, mainly wheat (967 kt in 2013), maize for grain (904 kt in 2013) and triticale (864 kt in 2013) (Central Statistical Office of Poland, 2015; Statistical Office in Poznan, 2014). It represented 16 % of the national production (in tonnes) in 2013 (Central Statistical Office of Poland, 2015). Wielkopolskie is also the leading region of Poland for sugar beet production (2 450 kt in 2013) and vegetable production (Marshal Office of the Wielkopolska region in Poznan, 2015; Statistical Office in Poznan, 2014) that represented

\(^{242}\) According to Eurostat, an intensive farming system is characterised by the significant use of capital and inputs such as fertilisers, pesticides and feedstuff, leading to higher yields per unit of land than in extensive farming (Eurostat, 2014).

\(^{243}\) Agricultural output at basic price (OBP)
21.8 % and 11.4 % of the national production respectively in 2013 (Central Statistical Office of Poland, 2015). The region also produces a large quantity of fodder maize with 4,154 kt produced in 2013 (Statistical Office in Poznan, 2014).

In the region, livestock was raised in 57.4 % of the holdings in 2013 (Eurostat, 2015i). Among the holdings with livestock, 38.9 % had less than 5 LSU and 71.1 % had less than 20 LSU in 2013 (Eurostat, 2015j). Livestock production is mainly pigs, poultry and cattle. The region presents the highest number of pigs and the third highest number of cows in the country (PURPLE, 2013; Central Statistical Office of Poland, 2015). In 2013, 860,000 cows were recorded, which corresponds to 15.4 % of the national herd. Also, the region accounted for 36 % of the number of pigs in Poland with 3.9 million live swines in 2013 (Statistical Office in Poznan, 2014; Central Statistical Office of Poland, 2015; Eurostat, 2014a). In Poland, in 2010, 34 % of farms kept pigs for their own needs and 21 % of the milk produce was used in farms (own consumption and animal feed) (Cebulska, et al., 2012; Szajner, 2009).

5.8.1 Notable impacts of nutrient surplus

This section focuses on the impacts of nutrient losses that are specifically related to agricultural practices in the region of Wielkopolskie. Nutrient losses to water, soil and air do not only result from agricultural production but also from industrial activities and urban settlements. There is no identified issue related to potassium excess in this region.

Wielkopolskie is the Polish region with highest phosphorus surplus with 7.4 kg P/ha. It is the second most saturated region in Poland for nitrogen with an average of 84 kg N/ha (Ziółkowski, et al., 2013) in 2005-2008. According to Fotyma et al. (2012), the nitrogen budget amounted 90.4 kg N/ha UAA in 2008-2010 (Fotyma, et al., 2012), showing a slight increase compared to the previous period. The distribution of the nitrogen budget in 2005-2008 is shown in Figure 104 and the distribution of phosphorus is shown in Figure 105.

![Figure 104 – Nitrogen budget in Wielkopolskie (outlined in red) and other regions in Poland in 2005 – 2008 [kg P/ha of agricultural land (average value from 3 years)]](image)

Source: (Gaj & Bellaloui, 2012)

244 About 17 saws (1 saw = 0.3 LSU) according to the Polish calculation method
Since 2012, 527,000 ha of the total Wielkopolskie area are designated as a Nitrate Vulnerable Zone (NVZ), which represents 17.3% of the area of the region. In Poland, NVZs represent 4.5% of the national area and 7.4% of the UAA (Polish Water Management Authority (KZGW), 2014).

**Impacts of nitrogen losses**

**Nitrogen load in freshwater**

**Surface water** – The average total nitrogen concentration in the Oder River (including nitrate, nitrite and ammonia) is low with 3.0 mg N/L in 2013\(^{245}\). In particular, the average concentration of nitrate in the Oder River is very low, amounting to 1.7 mg N/L in 2013 (5.2 mg NO\(_3^-\)/L)\(^{246}\). As a comparison, the European average nitrate concentration in rivers was 9.7 mg NO\(_3^-\)/L in 2010 (EEA, 2012c). In Poland, less than 1% of the monitoring stations for surface water quality exceeded 50 mg NO\(_3^-\)/L for the period 2008-2011 and 4.2% of the stations exceeded 25 mg NO\(_3^-\)/L (European Commission, 2013b).

It is interesting to note that although the nitrate concentration in surface water is low, the average nitrate concentration in drainage water in Wielkopolskie is relatively high, amounting to 10.1 mg N/L (44.7 mg NO\(_3^-\)/L) (European national average: 4.3 mg N/L (18.9 mg NO\(_3^-\)/L) (Fotyma, et al., 2012). The difference of concentration between the drainage water and the surface water can be partly explained by the high

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\(^{245}\) Unpublished data from the Inspectorate of Environment Protection (Szczecin) communicated by Marianna Pastuszak (November 2014)

\(^{246}\) Unpublished data from the Inspectorate of Environment Protection (Szczecin) communicated by Marianna Pastuszak (November 2014)
nutrient retention capacity of the riverine system (39.5 % in Poland) (Patuszak, et al., 2012; HELCOM, 2011).

**Groundwater** – Regarding groundwater, the figure below shows that more than 35 % of the tested shallow groundwater samples were higher than 50 mg NO$_3$-L in certain areas.

![Map of Wielkopolskie region](image)

**Figure 106 – Percentage of shallow groundwater samples with excess of nitrate according to Nitrates Directive criteria of 50 mg/L nitrates**

Nitrates from farms are the first source of pollution of deeper groundwater (Igras, 2004). It should be noted that the nitrates concentration in shallow groundwater does not prevail on the nitrates concentration in deeper groundwater. Indeed, a study led in the region showed that the nitrates concentration were still high in unconfined part of the flow system while an absence of nitrate was observed under confined conditions due to denitrification processes (Dragon, 2012).

**Eutrophication in the Szczecin lagoon**

Surface water such as the Oder River flows to lakes, lagoons (Szczecin Lagoon) and the Baltic Sea (Figure 107). This contributes to the accumulation of nutrients in water reservoirs and enhances the risk of eutrophication.

The Szczecin lagoon covers 687 km$^2$ but is quite shallow (average depth 3.8m) with brackish water that receives the water loaded in nutrients from the Oder River basin. Eutrophication is due to both nitrogen and phosphorus, depending on the season. Phosphorus is limiting in the spring while nitrogen is limiting in summer periods due to a lower freshwater inflow (Wielgat & Schernewski, 2002; Bangel, et al., 2001; Pastuszak, 2012a).
The average annual total nitrogen load\textsuperscript{247} discharged by the Oder River into its estuary in 2013 was 48,000 tons while the average nitrate load by the Oder River was 29,000 tons (Pastuszak, 2012b; Pastuszak, 2014). During the period 2003-2008, the average total nitrogen load in the Oder River was due to drainage from tilled land (48%), groundwater (24%) and Waste Water Treatment Plants (WWTP) (12%) (Kowalkowski, et al., 2012).

According to Stybel et al. (2012), a good chlorophyll status for lagoon water is less than 12.7 μg/L. In 2007, an average of 68.6 μg/L of chlorophyll A was reported in the lagoon (Stybel, et al., 2012). One of the most notable consequences of the trophic state of the water in the lagoon is abundant growth of phytoplankton leading to occasional blooms. There are two phytoplankton blooms. The first is in spring, predominated by diatoms, and the second occurs in summer with a high proliferation of blue green algae (e.g. Microcystis Aeruginosa) proliferation (Schernewski & Radziejewska, 2010) leading to bathing restrictions. It is estimated that nitrogen is the limiting nutrient in summer while phosphorus is the limiting nutrient of eutrophication in spring (Wielgat & Schernewski, 2002; Bangel, et al., 2001).

**Eutrophication marine water: the Baltic Sea**

It is estimated that 75% of the total nitrogen load into the Baltic Sea are discharged from the riverine outflow. The other source of nutrients is atmospheric deposition (Pastuszak, 2012a).

The largest nitrogen discharges are by far from Poland, representing 154,750 t N on average for the period 2005-2009 (26% of the total N discharged into the Baltic Sea) (NORDEN, 2011; HELCOM, 2013). In 2010, the total nitrogen load rose to 269,600 t N (NORDEN, 2011; HELCOM, 2013) due to unusual high outflows (Pastuszak & Witek, 2012a; Pastuszak, 2014). Poland is also the second greatest supplier of riverine water to the Baltic Sea, which partly explain the high nutrient discharge from the country (Pastuszak, 2012a). In Poland, agriculture was responsible for about 49% of the nitrogen load in 2006. Indeed, 65% of the nitrogen inputs in the riverine system are due to diffuse source in 2006, including 75% related to agriculture (HELCOM, 2011).

\textsuperscript{247} The total nutrient and phosphorus load mentioned below are flow normalised total load.
Inorganic nitrogen (nitrate, nitrite and ammonia) concentrations in the river flowing from the lagoon to the Baltic Sea (Świna Straits) were much lower than in the Oder (Pastuszak & Witek, 2012a; Pastuszak & Witek, 2012b). It is estimated that only 55 % of the total nitrogen load and 63 % of the total phosphorus load from the estuary discharge into the Baltic Sea.

The Baltic Sea is highly affected by eutrophication. The most notable area affected by marine eutrophication in the Baltics is the Kattegat Bay between Sweden and Denmark. Few visible indicators of the eutrophication such as surface masses of cyanobacteria or a decrease of water clarity due to the increase in surface masses of phytoplankton and macroalgae have been reported on the Polish coast. It is estimated that 26 % and 39 % of the Polish monitoring sites for transitional and coastal waters were eutrophic or hypertrophic respectively in 2008-2011 (European Commission, 2013b). The increase of phytoplankton biomass favours the development of opportunistic species over the perennial species, impacting the community structure and the habitat quality. In the Baltic Sea, the geographic distribution of the submerged aquatic plant bladderwrack has changed. So has the distribution of the Eelgrass that is considered as an appropriate eutrophication indicator (HELCOM, 2013) since the variation of nutrients explain up to 75 % of its distribution (Nielsen, et al., 2002). In addition, in the Bothnian Bay, where the background concentrations of nutrients are relatively low, positive relationships between increased nutrients in the water column and benthic communities are observed (HELCOM, 2013).

Ammonia (NH₃) emissions to air

In Poland, an annual average of 262 000 tons of ammonia was emitted in 2012, of which 98 % was from agricultural activities (Eurostat, 2014g). Wielkopolskie is the region with the highest ammonia emissions in Poland. The NH₃ emissions in Wielkopolskie are over 2.2 t/km²/year (Bienkowski, 2010). The high level of emission of ammonia caused by livestock in Wielkopolskie is due to cattle breeding (55 %), pig breeding (30 %) and poultry (15 %) (Bienkowski, 2010).

One major consequence of nitrogen emissions to air is the direct deposition or precipitation (rain and snow) of nitrogen on soils and water bodies (acid rains) thus increasing soil acidification, in particular in forest areas. (Bienkowski, 2010).

Impact of phosphorus losses

Phosphorus load in freshwater and eutrophication

The average total concentration of phosphorus (bounded and dissolved phosphorus) in the Oder River amounts to 0.2 P/L in 2013. The average dissolved phosphorus (phosphate) concentration in the Oder River is 0.04 mg P/L (0.1 mg PO₄³⁻/L) over the last 7 years, which is lower than the European average orthophosphate concentration in rivers (0.21 mg PO₄³⁻/L in 2010 (EEA, 2012c). According to the US EPA water quality criteria, phosphate concentration should not exceed 0.1 mg/L if streams

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248 Unpublished data from the Inspectorate of Environment Protection (Szczecin) communicated by Marianna Pastuszak (November 2014)
249 Unpublished data from the Inspectorate of Environment Protection (Szczecin) communicated by Marianna Pastuszak (November 2014)
discharge into lakes or reservoirs (in this case the Szczecin lagoon for instance) to control algal growth (US.EPA, 1986).

It is interesting to note that although the phosphate concentration in surface water is low, the average phosphate concentration in drainage water in Wielkopolskie is relatively high at 0.14 mg P/L (0.43 mg PO$_4^{3-}$/L) (Fotyma, et al., 2012). The difference of concentration between the drainage water and the surface water can be partly explained by the high nutrient retention capacity of the riverine system (46.5 % in Poland) (Patuszak, et al., 2012; HELCOM, 2011).

Phosphorus is commonly considered as the main limiting nutrient causing eutrophication in fresh water. Despite the average good quality of water, 43 % of fresh water sites were eutrophic or hypertrophic. In particular, 86 % of lakes were eutrophic or hypertrophic (European Commission, 2013b).

The Niepruszewskie Lake is an example of lake with freshwater eutrophication that is mainly due to agricultural activities in Wielkopolskie. It is fed by the Samica Steszewska River which lies in the Wielkopolska lowland. The land in the Samica River basin is primarily used for agriculture: 80 % of the land is arable. The lake showed high total phosphorus concentrations and algal blooms (Ławniczak, et al., 2008): between 2005 and 2007, the phosphorus concentration and the chlorophyll concentration in the lake were above the threshold for hypereutrophic water (25 μg/L for chlorophyll concentration and 0.1 μg/L for total phosphorus). In 2010, the nitrate concentration of the lake was 11 mg NO$_3^{-}$/L (Regional Inspectorate of Environmental Protection in Poznań, 2010). The pollution in lakes appear to be a major threat for ecosystems as the balance of species may be affected (Oleksyn & Reich, 1994). It should also be noted that, according to the Polish Ministry of Agriculture, there are other examples of regional lakes affected by eutrophication phenomena that are not mainly due to agriculture.

**Phosphorus load in the Szczecin lagoon and the Baltic Sea**

The average annual total phosphorus load$^{250}$ discharged by the Oder River into its estuary in 2013 was 3 200 tons while the average phosphate load by the Oder River was 500 tons (Pastuszak, 2012b; Pastuszak, 2014). During the period 2003-2008, the average total phosphorus load in the Oder River was due to erosion (28 %), WWTP (24 %), urban systems (17 %) and overland flow (15 %) (Kowalkowski, et al., 2012).

Dissolved phosphorus concentration is higher in the Świna Straits than in the Oder (Pastuszak & Witek, 2012a; Pastuszak & Witek, 2012b). Such concentrations in river waters have important impacts when reaching the downstream semi-closed lagoon of Szczecin. As mentioned above, phosphorus is the limiting factor of phytoplankton growth during spring (Wielgat & Schernewski, 2002; Bangel, et al., 2001). The excess of phosphorus bonds with bottom sediments which constitutes a natural reservoir of phosphorus. Depending on water characteristics, phosphorus trapped in the sediments is eventually released in considerable amounts which contribute to algae and phytoplankton population growth in the long-term (UNECE, 2011).

It is estimated that 95 % of the total phosphorus load in to the Baltic Sea is discharged from the riverine outflow (Pastuszak, 2012a). The largest phosphorus discharges are by far from Poland, representing 9 440 t P on average for the period 2005-2009 (36 % of the total P discharge to the Baltic Sea) (NORDEN, 2011; HELCOM, 2013). In 2010, the total phosphorus load rose to 14 910 t P due to unusually high outflows (NORDEN, 2011; HELCOM, 2013). As mentioned for nitrogen, the high nutrient discharge is partly explained by the fact that Poland is also the second greatest supplier of riverine water to the Baltic Sea (Pastuszak, 2012a). In Poland, agriculture was responsible for about 53 % of the

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$^{250}$ The total nutrient and phosphorus load mentioned below are flow normalised total load.
phosphorus load in 2006. Indeed, 65% of the nitrogen inputs in the riverine system were due to diffuse source in 2006, including 81% related to agriculture (HELCOM, 2011).

5.8.2 Causes of nutrient losses

Farming system and agricultural practices

Restructuration of farms and intensification leading to a decrease of SOM content and an increase of large and concentrated livestock farms – In 1989, in north and west Poland, the State Farms closed and gave their land and assets to the Treasury Agricultural Property Agency giving the opportunity to local farms to increase their land area (Banski, 2011). Since then, north and west Poland (the Oder basin), including the Wielkopolskie region, have seen their farms grow. Wielkopolskie was in 2013 the Polish region with the second largest number of farms over 100 ha (Central Statistical Office of Poland, 2015). In 2013, farms with more than 100 ha represented 2.2% of the UAA (Eurostat, 2015k) of the region for only 1.1% of the holdings (Central Statistical Office of Poland, 2015). On the other hand, 39.3% of the holdings covered less than 5 ha in 2013 while at national level, this was the case of 53.7% of the farms (Central Statistical Office of Poland, 2015). The regional average holding size was 14 ha/holding in 2010 (Eurostat, 2013a).

Decrease of SOM content – Following the restructuring and privatisation of the State-owned farms, the share of cereals increased while fodder crops were reduced. The amount of mineral fertilisers decreased while the yields increased overall. Livestock production drastically decreased, inducing a decline in the production of manure and in its use as an organic fertiliser. These changes resulted in a decrease of the SOM content, especially in Wielkopolskie where soils are light with a high risk of nutrient leaching (Jankowiak, et al., 2003). Due to the reduction of livestock production, the amount of manure produced is not always sufficient to locally compensate the low SOM content.

Increase of large and concentrated livestock farms – The restructuring of farms also led to a specialisation and concentration of livestock production. This is due to, amongst others, the Polish accession to the EU, and related increasing competition. In Wielkopolskie, the average livestock density is relatively low compared to the other regions under study with 0.7 LSU251 ha UAA in 2009-2011 (national average: 0.45 LSU/ha UAA) (Fotyma, et al., 2012). In general, this suggests that livestock production does not exert strong pressure on the environment (Pietrzak, 2012a). However, the impact of large scale livestock production on water quality can be significant (Pietrzak, 2012a). In Wielkopolskie, the highest nutrient surplus is found in livestock farms, in particular in farms solely or mainly dedicated to pig production (Pietrzak, 2012a). The region produced 16% of the slurry in Poland, 18% of the manure and 17% of the liquid manure in 2008-2011 (Fotyma, et al., 2012). The local high risk of environment pollution results from improper storage and handling of manure, in particular for slurry and when manure is stored directly on the ground in fields (Kalinowska, 2014; Skorupski, 2012).

Poor livestock farming infrastructure – It is estimated that nitrogen leaching from the livestock buildings and manure heaps represents about 40% of the total nutrient losses from farming sources in Poland (Pietrzak, 2012a). This is partly explained by the poor technical conditions of some livestock buildings and facilities or the absence of proper storage facilities. Indeed, almost 50% of these buildings

251 Note that the definition of LSU in Poland is different from that in Western Europe. Following Polish Ministerial regulations of 9 November 2004 (Dz. U. Nr 257, poz. 2573, 2004) Polish LU=cow weighing 500 kg. Hence it can not be compared to the average EU-28n livestock density (0.77 LSU/ha UAA in 2010) for which 1LSU = cow weighting 600 kg (Eurostat, 2012a). With the Eurostat definition, the livestock density amounted 1.1 LSU/ha UAA in 2013 (national average: 0.6 LSU/ha UAA) (Central Statistical Office of Poland, 2014).
were built before 1960 and now they do not all meet modern legal zootechnical requirements (Krasowicz, et al., 2012). In 2010, only 23 %, 38 % and 12 % of the holdings in Wielkopolskie had storage facilities for solid manure, liquid manure and slurry, respectively (Eurostat, 2013c). A significant share of the solid manure is stored in heaps (Dach, et al., 2012; Sobolewski, 2011), which means possible high risks of nutrient transport by leaching or run-off and increased atmospheric emissions.

The minimum storage capacity for manure must allow to store six months of manure produced in the Nitrate Vulnerable Zones (NVZs), according to the regional regulation implementing the action program aiming at reducing nitrogen surplus from agricultural sources in the NVZ.

Outside the NVZs, according to the Polish Fertiliser and Fertilisation Act, the storage capacity must be four months for farms breeding more than 40 000 fowls or 2 000 pigs over 30 kg or 750 sows. In addition, the construction of a concrete floor for solid manure storage is mandatory for these farms and the slurry tanks must be impermeable. On one hand, among the holdings with slurry tanks, only 6 % do not comply with the volume storage requirements (four months storage) while 73 % have a higher storage capacity. As regards storage of solid manure on concrete floors, in 11 % of holdings the surface of the concrete floor is too small and 41 % of holdings have higher storage capacity than required. (Polish Ministry of Agriculture and Rural Development, 2012). On the other hand, these requirements only concerned a small share of the holdings in Wielkopolskie: holdings with more than 100 LSU (eq. 333 sows) represented less than 1.7 % of the total holdings in the region in 2013 (Eurostat, 2015j). It should be noted that cattle farms are exempted from these requirements. Also, a storage volume of four months may not be sufficient to allow a proper application of manure at the right time. If the amount of manure produced exceeds the existing storage capacity, farmers generally apply the excess manure in the fields. Outside the appropriate period for manure spreading, the crops may not be able to assimilate applied manure, leading to an excess of nutrients in soil that could be easily leached.

The livestock manure storage facilities are not built or modernised partly because of a lack of financial resources (Alterra, 2007). Manure storage infrastructure has been improved over the last 10 years through the construction of concrete floor for solid manure storage and slurry tanks thanks to aid programmes252 and the requirements of national regulation. However, the subsidies provided by the aid programmes have been too low to improve the storage capacity of a large number of farms: by 2010 they had increased the storage capacity by only 0.25 % for solid manure and 0.14 % for liquid manure compared to the available storage capacity in 2002 (high estimation (Pastuszczak, 2010), and only benefited less than 5 % of the Polish holdings (Polish Ministry of the Environment, 2010). Hence, the situation is still not satisfactory (Kalinowska, 2014).

Lack of implementation and accuracy of the fertilisers management plan – Figure 108 below shows that the Oder River basin (west of Poland) has the highest nitrogen fertiliser consumption in Poland in 2009-2011 (Fotyma, et al., 2012). In Wielkopolskie, an average of 91 kg of nitrogen per hectare of agricultural land was applied between 2009 and 2011 (Fotyma, et al., 2012) and an average of 20.5 kg of phosphorus is applied per hectare each year (Ławniczak, et al., 2008; Igras & Fotyma, 2012).

252 See section 1.1.4.1

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According to the regional regulation implementing the action program aiming at reducing nitrogen surplus from agricultural sources in the NVZ, only holdings of more than 100 ha which are situated in a NVZ, must prepare a fertilisation management plan. Outside the NVZ, the drafting of a fertilisation management plan is mandatory for large pig and poultry farms (more than 100 LSU) according to the Polish Fertiliser and Fertilisation Act. This concerns a limited share of the holdings – in 2013, 1.1 % of the holdings had more than 100 ha and 2.7 % of the holdings had more than 100 LSU -, that represented one fourth of the regional UAA and 46.4 % of the regional LSU in 2013 (Central Statistical Office of Poland, 2015).

In addition, soil analysis is conducted only every three years, what the participants of the regional conference considered insufficient. A strong lack of information has been identified as a barrier by the farmers that participated in the regional conference. In particular, it concerns the method to elaborate a fertilisation management plan, the annual nutrient content of soils and the crop needs. This suggests that the fertilisation management plans may not be as accurate as they could be.

Very few farmer collective actions – In Poland, many farmer cooperatives flourished from the 19th century until the communism era. At the end of communism, in 1990, due to the change of the political and economic system, most of the largest cooperatives collapsed (Matczak, 2012). Since then, very few new farmer collective actions have been undertaken in Poland. Feedback from the regional conference indicate that the main collective actions mostly concern the sales of farm products. None or very few actions related to agricultural practices are implemented in the region. Wielkopolskie is one of the regions that counts the highest numbers of production cooperatives, relative to this low baseline. For

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253 As part of the dissemination part of the project regional conferences were held in selected regions. One was organised in Poznan, Poland in November 2014.
instance, Wielkopolskie registered 55 pig breeder groups in 2009, comprised of 20 farms on average, gathering 51% of the pig breeder national cooperatives (Matczak, 2012). Participants to the regional conference indicated that although farmers are receptive to what their neighbours do, the lack of trust is a barrier to the implementation of collective actions. Thus, as individuals, in particular the small Polish farmers, farmers cannot afford to buy expensive equipment or implement different techniques that is necessary to the modernisation and improvements of agricultural systems. For instance, only few farmers apply slurry with trailing hose or shoe, mainly due to the cost of the machines (Dach, 2015; Borkowski, 2014).

**Dense drainage network** – The dense drainage network with large tile drainage area may reduce the risk of water overload but increases the quantity of water drained and transported towards watercourses (Pastuszak, 2012a).

**Environmental conditions**

**Podzolics soils leading to high nutrient leaching and low soil fertility** – Wielkopolskie, similarly to the whole of Poland, has mainly podzolic soils (The Board of the Wielkopolskie Voivodship, 2011) which are soils with a coarse texture and a very acidic pH, developed in cold and humid territory.

Coarse-textured soils containing more sand particles have large pores and are highly permeable, allowing water to move through rapidly. The nutrients are transported with the water towards water bodies. By contrast, the emissions of N\textsubscript{2}O is low in sandy soil, which explains why this is not an issue in Wielkopolskie.

In addition, the soil is very acidic in Poland (Igras, 2004). In Wielkopolskie, 42.1% of the soils are considered as acidic and very acidic (Fotyma, et al., 2012). Soil acidity tends to enhance the dissolution of nutrients, in particular phosphorus. As a consequence, if nutrients are in the soil solution, they are more easily leached or transported by run-off. This results in soils with low fertility that are not favourable to agriculture (Chief Inspectorate for Environmental Protection, 2010). Pietrzak et al. (2012b) studied the pH of soils in Polish grassland. He showed that in Wielkopolskie, almost 26% of mineral soils and nearly 53% of organic soils have low available phosphorus content and 60% of mineral soils and 93% of organic soils have low potassium content (Pietrzak, 2012b).

**Important erosion risk** – In Poland, in the period from 2009 to 2012 28% and 21% of soils presented a high risk of water and wind erosion respectively (Baltic Compass, 2010). In Wielkopolskie, water erosion is not a major issue but soils face a higher risk of wind erosion (Borreli, et al., 2014). Some nutrients such as phosphorus are tightly bound to soil particles and may be transported with them during periods of high winds. In the region, erosion is responsible for 21% of loss of phosphorus (Kowalkowski, et al., 2012). In addition, soil erosion as a result of harvest activities may affect the region. Indeed, the region has a large production of sugar beet whose harvesting is responsible for soil loss of 9 t/ha/yr on average (Jones, et al., 2012).

**Impact of the climate on nutrient run-off and leaching** – Wielkopolskie is characterised by low precipitation (Joint Research Centre, 2012b) with an annual average of 540 mm (The Board of the Wielkopolskie Voivodship, 2011). In 2011, 39% of the districts threatened by drought in Poland were located in Wielkopolskie (Agricultural Drought Monitoring System (ADMS), 2011). The low rainfall decreases the risk of nutrient losses by run-off or leaching considering the low risk of water logging and heavy rainfall.

The number of days with snow can reach up to 57 days in Wielkopolskie. Cold and snowy winters (frozen soils) enhance nitrogen run-off which contributes to the nutrient load in water (Ławniczak, et al., 2008). Indeed, snow and ice constitute reservoirs of nutrients. During the melting period, important loads of
nutrients are freed in a short period of time. Plant uptakes are inferior to the available nutrients. Nutrients are then more likely to be leached or transported by run-off. In wet years, the phosphorus and nitrogen load in the Szczecin lagoon coming from Oder River can be up to twice as high compared to dry years (Schernewski & Wielgat, 2001).

**Lagoon structure that catches nutrients** – The Szczecin lagoon is characterised by intensive water dynamics and matter turnover (Christiansen, et al., 2002; Emeis, et al., 2002). Thus, the lagoon functions as a natural biological and dynamic treatment system (Jakuczun & Nowacki, 1994; Wielgat & Witek, 2004; Minning, 2004; Pastuszak, et al., 2005). A significant share of the nutrients are retained and degraded in the estuarine system. Prior to the dilution effect and the N uptake by phytoplankton, denitrification is the major process responsible for removing N in most estuaries (Pastuszak & Witek, 2012b). The amount of N removed from the Oder estuaries by denitrification is 30 % of the total nitrogen inputs from the rivers and the atmosphere (Pastuszak & Witek, 2012b), leading to high N₂O emissions.

**Horizontal salinity stratification of the water masses in the lagoon** – This feature increases the vulnerability of the Baltic Sea. It is mainly a result of the large inflow of freshwater from the catchment areas surrounding the Baltic Sea, including the many rivers. This salinity stratification does not allow ventilation and oxygenation of the bottom waters and sediments by vertical mixing of the water, which often leads to oxygen depletion. In the absence of oxygen, chemically reduced sediments also release significant quantities of phosphorus to the overlying water. When phosphorus is released, it contributes to eutrophication and algae and phytoplankton blooms (Pastuszak & Igras, 2012).

### 5.8.3 Costs of the environmental and health effects

**5.8.3.1 Social economic description of the study area**

Wielkopolskie accounted for 9 % of the Polish population in 2014. With a GDP of € 37 billion in 2012, Wielkopolskie was the region with the third highest gross value added in Poland and contributed 9.5 % of the national income in 2012. The south and south-east of the region are dominated by vast fields, where agriculture and related processing industries prevail. The eastern part of Wielkopolskie’s economy is mainly based on energy production, mining and the extractive industries of brown coal, natural gas and salt. Wielkopolskie is the leading agricultural region of the country and accounted for 16.9 % of the value of the national agricultural production in 2013 (Central Statistical Office of Poland, 2015) ²⁵⁴ for 12% of the national agricultural area in 2013. The region is also notable for its numerous and vast protected areas for fauna and flora. In Wielkopolskie, there are several lakes with high environmental value. These lakes are also important for local recreational activities and contribute to the dynamism of the region.

From a broader perspective, most of the coastal areas of the Oder River basin are protected natural area. Tourism, agriculture, fishing (3 000 t/a in the Szczecin lagoon), and shipping are important economic activities in the coast. Along the coastline, tourism is the exclusive activity and it is likely that altogether more than 10 million tourists visit the estuary region each year (Schernewski, et al., 2012).

In Wielkopolskie and the larger impacted area, the notable amount of nutrient surplus due to farming practices over the past years has resulted in significant environmental damages and economic losses (see below).

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²⁵⁴ Agricultural output at basic price (OBP)
Note that some data provided in this section may cover a larger area than the Wielkopolskie region. Indeed, the watercourses in the selected region of Wielkopolskie contribute to the nutrient load in downstream watercourses, lakes, lagoons and the Baltic Sea. Thus, it is interesting to look at these water bodies outside the selected region to measure the economic impact of local pollution at a larger scale. Very few documents have been found about the economic valuation of nutrient surplus damages on the specific region of Wielkopolskie.

5.8.3.2 **Review of economic damages**

In order to better understand the economic damages associated with nutrient surplus in Wielkopolskie, or more broadly in the Oder River basin, a range of literature was reviewed. Costs were identified and are presented below, based on the classification of economic damages caused by environmental impacts presented in Annex 14.

**Clean up and restoration costs (CRC)**

Nutrient overload in Wielkopolskie and throughout the Oder River basin has considerable impacts on water bodies and most notably on the Baltic Sea.

**Preventive measures to reduce nutrients concentration in water**

At the scale of the Baltic region (Baltic Sea and surrounding countries), according to the HELCOM report of 2009, measures with a unit abatement cost of less than € 150 000 per ton of reduced phosphorus were considered as cost-efficient. Projects on the eastern and southern side of the Baltic Sea were potentially ten times more cost-effective than projects carried out in the Nordic countries (HELCOM, 2009). Tyszewski & Puslowska-Tyszewska (2012) reported that for the Oder River basin, several actions to reduce agricultural nitrogen pollution in water have been planned and would cost around € 73 244 211. This expenditure represented 30 % of the total budget for agricultural measures in the Oder River basin management.

The study by Turner et al. (1999) evaluates the unit abatement costs for nutrients (N and P) reduction for all the Baltic countries, including Poland. These costs correspond to agricultural measures such as the reduction in the use of fertilisers, the reduction of livestock and the spreading time of manure (to limit leaching) for example. The study estimates that in Poland, the unit abatement cost for phosphorus reduction with agricultural measures is € 12.5 to € 223.6 per kg of phosphorus reduced and € 1.32 to € 11.1 per kg of nitrogen reduced. The average scale for the whole Baltic region is € 20 to € 1 900 per kg of phosphorus reduced.

In the same 1999 study, annual willingness to pay (per person and for the whole country) for a global nutrient reduction strategy in the Baltic Sea was evaluated. In Poland, the annual WTP per capita is estimated at € 92 and the national WTP per year is € 1 279 410. It is important to compare these estimations to other transition economies around the Baltic Sea such as Lithuania for example with a national WTP per year of € 101 530. Moreover, it is essential to bear in mind that these estimations were made in 1999 and that a lot of economic indicators used to make these estimations have changed.

Moreover, in the Turner study, it is estimated that for Poland, the reduction by 63 % of the nutrient load in the Baltic Sea would cost 1 056 000 €/year but considering that the benefits presented above reach 1 279 410 €/year, the net benefit of such a measure would be 193 710 €/year. These estimations show that the benefits of clean up measures were globally positive at the time of the study. No other studies have been found at this date to confirm this tendency with more recent estimations; however it shows that such measures are very costly.
**Nutrient removal**

Regarding drinking water treatments, in Poland, 19% of the population was supplied with drinking water not meeting quality standards in 2004 (GHK, 2006). This was due to the poor water quality regarding nitrate and ammonia concentration but also other pollutants such as Fe and Mn in excess, turbidity or pH for which agriculture may not be responsible for. Throughout Poland, another reason for poor quality of drinking water is the insufficient treatment installation (in terms of technology applied as well as maintenance condition). The total investment expenditure for drinking water treatment to meet legal standards was estimated at € 52 200 000 each year in Poland (GHK, 2006).

**Table 54 – Clean up and restoration costs found in literature**

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Cost</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Expenditure for clean-up and restoration purposes</td>
<td>Unit abatement costs for phosphorus reduction (average for the Baltic sea countries)</td>
<td>€ 20 to € 1 900 per kg of phosphorus reduced</td>
<td>Baltic Sea countries</td>
<td>2007</td>
</tr>
<tr>
<td></td>
<td>Unit abatement costs for phosphorus reduction with agricultural measures</td>
<td>€ 12.5 to € 223.6 per kg of phosphorus reduced</td>
<td>Poland</td>
<td>1999</td>
</tr>
<tr>
<td></td>
<td>Unit abatement costs for nitrogen reduction with agricultural measures</td>
<td>€ 1.32 to € 11.1 per kg of nitrogen reduced</td>
<td>Poland</td>
<td>1999</td>
</tr>
<tr>
<td></td>
<td>Annual national WTP for a global nutrient reduction strategy in the Baltic Sea</td>
<td>€ 1 279 410</td>
<td>Poland</td>
<td>1999</td>
</tr>
<tr>
<td></td>
<td>Drinking water treatment to meet legal standards (surface and groundwater) related to nitrate or ammonia but also other pollutants such as Fe, Mn or turbidity.</td>
<td>€ 52 200 000/year</td>
<td>Poland</td>
<td>2005</td>
</tr>
<tr>
<td></td>
<td>Agricultural nitrogen, water pollution reduction</td>
<td>€ 73 244 211</td>
<td>Oder River Basin</td>
<td>2012</td>
</tr>
</tbody>
</table>

**Use value damages (UVD)**

Nutrient load in the water bodies in Wielkopolskie contributes to the pollution of downstream waters including the Baltic Sea. In the sea, blue green algae caused by eutrophication and litter are the most important threats to tourism sector of the region. According to an interview study (Hasselström, 2008) there are no dramatic health or other effects from the algae, but reduced aesthetic values might have caused a slowdown in tourism industry growth. More importantly, beach tourism shows high sensitivity to future potential blue green algae blooms which can be toxic in high quantities.

Although the previously mentioned study shows that nutrient overload has little impact on the Baltic Sea near the Polish coast, it is less the case in the specific region of Wielkopolskie where lakes, for instance, are numerous and constitute important ecosystems. A study in the region (Fiszer, et al., 2012) showed that the four studied lakes presented a very low oxygen rate in summer as a consequence of their trophic state. Moreover, these lakes shelter an economically valuable fish species, *Coregonus albula* which is highly threatened by the lack of oxygen. However, no estimates of the associated economic loss have been found in the literature.
Table 55 – Use value damages of eutrophication found in literature

<table>
<thead>
<tr>
<th>Type of cost estimated</th>
<th>Description</th>
<th>Damage estimation</th>
<th>Sensitivity to future potential changes</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduced revenues in tourism, fishery sector etc.</td>
<td>Beach tourism</td>
<td>Slowdown the tourism activity on the coastal areas.</td>
<td>High, if the blooms occur more severely</td>
<td>Poland</td>
<td>2008</td>
</tr>
<tr>
<td></td>
<td>Recreational fishing</td>
<td>Medium (some species are affected by the consequences of trophic state in some lakes for example)</td>
<td>Medium (eutrophication becomes a nuisance for other marine species)</td>
<td>Poland</td>
<td>2008</td>
</tr>
<tr>
<td></td>
<td>Boating</td>
<td>Not affected</td>
<td>Medium (local effect may occur due to eutrophication)</td>
<td>Poland</td>
<td>2008</td>
</tr>
<tr>
<td></td>
<td>Cruises</td>
<td>Not affected</td>
<td>Low</td>
<td>Poland</td>
<td>2008</td>
</tr>
<tr>
<td></td>
<td>Real estate</td>
<td>Not affected</td>
<td>Medium</td>
<td>Poland</td>
<td>2008</td>
</tr>
</tbody>
</table>

**Passive use value damages (PUVD)**

The Baltic Sea is among the most productive ecosystems, with much of the area providing services with an annual worth in the range of € 1 460 to € 2 190 per hectare (Costanza, et al., 1997). Moreover, there are numerous surface water bodies in Wielkopolskie which constitute important ecosystems. Pollution may damage the ecosystems thus potentially affecting biodiversity by changing the balance between species. It is the case of the lakes presented in the study (Fiszer, et al., 2012) in which some fish species suffer from the trophic state of the water and may eventually disappear. At this date, no studies have been found to evaluate the externalities regarding biodiversity, ecosystems, etc.

**Policy action costs (PAC)**

In Wielkopolskie, different measures in the scope of the Action Programme implementing the Nitrate Directive have been put into place in order to reduce surplus of nutrient at farm level. Environmental monitoring of surface water and groundwater is carried out by the Regional Inspectorate for Environmental Protection in Poznań, soil monitoring is carried out by the Regional Chemical and Agricultural Station in Poznań (Soil monitoring in the framework of the State Environmental Monitoring Programme is under responsibility of the Institute of Soil Science and Plant Cultivation in Pulawy) (Regional Water Management Authority in Poznan, 2012).

Development and dissemination of information materials, organisation of workshops and meetings for farmers, and other measures such as development of a tool to calculate production of nitrogen at farm level, are carried out by the Agriculture Advisory Centre in Poznań. Cost valuations have been made for only some of the measures so far and are presented in the Table 56.

In addition, monitoring activities at national level related to water and soil pollution have been identified, although they are not specifically intended to reduce surplus of nutrient.

The most relevant recent examples of surface water, groundwater and soil monitoring programmes in the framework of the State Environmental Monitoring Programme (Polish Inspectorate for Environmental Protection, 2014), that can be related to agriculture activities, are:
• Monitoring of arable soil chemistry, funded to the amount of € 190 337 for the period 2010-2012\textsuperscript{255};
• Monitoring of rivers and lakes sediments, funded to the amount of € 476 900 for the period 2010-2012;
• Assessment of river basins based on results of surface water bodies monitoring (rivers), funded to the amount of € 723 386 for the period 2008-2010.

Similarly, at regional level, recently implemented programmes co-financed by the National Fund for Environmental Protection and Water Management in the framework of the State Environmental Monitoring programme (Regional Inspectorate for Environmental Protection in Poznań, 2011), are:

• Purchase measuring and control instruments to carry out tests of water in the framework of SEM, funded to the amount of € 139 112 in 2007;
• Purchase of supplies for air monitoring network for the region and the mobile stations to support air monitoring station network, funded to the amount of € 340 535 in 2010;
• Modernisation of facilities and equipment in laboratories in Poznań, Kalisz and Konin (local delegations) in order to ensure the appropriate conditions for of analytical instrument, funded to the amount of € 318 704 in 2011.

Table 56 – Policy actions costs found in literature or information obtained from the regional authorities

<table>
<thead>
<tr>
<th>Type of cost</th>
<th>Description</th>
<th>Cost (euros)/year</th>
<th>Scale</th>
<th>Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Assessment of the eutrophication degree of inland surface waters, marine</td>
<td>Evaluation according to the methodology developed by the Chief Inspector of Environmental Protection. It takes into</td>
<td>N/A</td>
<td>National</td>
<td>Since 2010 every 3 years</td>
</tr>
<tr>
<td>internal waters and coastal waters (Jagusiewicz, 2009).</td>
<td>consideration the indicators characterizing oxygenation conditions, nutrient conditions as well as a selected</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>biological element.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Surface water, groundwater and soil monitoring programmes in the framework</td>
<td>Monitoring of arable soil chemistry</td>
<td>€ 190 337</td>
<td>National</td>
<td>2010-2012</td>
</tr>
<tr>
<td>of the State Environmental Monitoring Programme</td>
<td></td>
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<td></td>
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<tr>
<td></td>
<td>Monitoring of rivers and lakes sediments</td>
<td>€ 476 900</td>
<td>National</td>
<td>2010-2012</td>
</tr>
<tr>
<td></td>
<td>Assessment of river basins based on results of surface water bodies monitoring (rivers)</td>
<td>€ 723 386</td>
<td>National</td>
<td>2010-2012</td>
</tr>
<tr>
<td>Implementation of the Action Programme aiming at reducing nitrogen surplus</td>
<td>Environmental monitoring of soil</td>
<td>N/A</td>
<td>Regional</td>
<td></td>
</tr>
<tr>
<td>from agriculture sources in NVZ</td>
<td>Education activities: development of information materials for farmers</td>
<td>N/A</td>
<td>Regional</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Counselling activities: organisation of trainings and counselling sessions for farmers in NVZ</td>
<td>N/A</td>
<td>Regional</td>
<td></td>
</tr>
</tbody>
</table>

\textsuperscript{255} Amounts in EUR in this paragraph are calculated based on the average bank exchange rates at the end of the year of adoption of each measure.
5.8.4 Good practices to reduce nutrient losses at farm level

In Wielkopolskie, the identified good practices include measures aimed at decreasing the local source of pollution by extending and improving the nutrients fertilisation management plan for all agricultural sites, improving the application technique or processing manure to ease its transfer when the quantity of manure produced locally exceeds the carrying capacity of the farm. Another type of good practices relates to the reduction of the amount of nutrients lost during storage by putting in place an impermeable floor, covering the solid manure stored in heaps or increasing the height of manure heaps and cooling slurry. Composting manure would also help at reducing the volume of manure stored and thus avoiding exceeding the manure storage capacity of the farm that may lead to the application of manure at an inappropriate time. Good practices can also help at avoiding and reducing nutrient losses when manure is applied through measures such as using the most suitable application technique and the construction of sedimentation ponds. Lastly, measures aiming at increasing the soil quality such as the use of conservation tillage, the incorporation of straw and soil coverage (in particular with nitrogen fixing crops) would help at decreasing the amount of fertiliser that the farmers needed to buy.

5.8.4.1 What has already been done in the region

Since 1996, the Polish government has issued policy guidelines to integrate environmental concerns into agricultural processes. The Polish Code of Agricultural Good Practices was developed in 2004, prioritising the “sustainable balance of mineral nutrients” and creating a monitoring programme to track the mineral nitrogen content in soil. The programme aimed to determine the correct fertiliser rates for crops and assess how to prevent nitrogen leaching from soil to water. Research is also being conducted on how to streamline fertiliser uptake in vegetable production through integrated and organic production methods.

In 2002-2006, the EU pre-accession aid programme called SAPARD, the programme “Restructuring and modernisation of food sector and rural development 2004-2006” and the Rural Development Programme 2004-2006 supported the construction of a significant quantity of manure pits and slurry tanks. A total of 30,057 renovation projects took place in Wielkopolskie for an amount of € 1.7 million (Lipiński, 2012).

The Wielkopolska Agriculture Advisory Centre in Poznań proposes trainings and workshops for farmers and advisors. It has also implemented a network of demonstration farms in the context of the Baltic Deal project. In 2012, the region accounted five demonstration farms (Baltic Deal, 2012b). In addition, the three river basins in the Wielkopolskie Vovoidship have water management plans to improve the surface water quality as well as reduce the nutrient load transported to the Baltic Sea. Poland is also the co-coordinator (along with Finland) of the Priority Area “Nutri” of the EU Strategy for the Baltic Sea Region, which is aimed at reducing nutrient inputs to the Baltic Sea. Lastly, a project founded by Baltic Sea 2020 on “Self-evaluation concerning nutrients by farmers in Poland” has been launched in 2013. The objectives of the project are to train farm advisors and engage with farmers to propose measures for nutrient run-off reduction based on their experiences. In 2013, the project led to the publication of a guide to improve nutrient management for both advisors and farmers (Ulen, et al., 2013).

5.8.4.2 Good practices to reduce the nutrient losses in livestock production

It is interesting to note that manure is mostly spread in autumn but that crop assimilation is higher in spring: 45% of solid manure and 40% of liquid manure is applied in autumn and 30% of both solid and liquid manure is applied in spring. The remaining 10% of the solid manure and 15% of the liquid manure is applied during summer (Baltic deal, 2013). Between the application period, the manure that cannot be stored in vessels is often stored directly on the ground.
**Install an impermeable floor**

As for the measure related to the improvement of storage equipment conditions, the high investment required by the renovation or the building of adequate storage tanks results in a low probability of implementation (Skorupski, 2012; Kalinowska, 2014). Programmes have been implemented to help farmers build impermeable (e.g. concrete) floors for the temporary storage of solid manure and tanks for liquid manure and slurry. For example, the European SAPARD programme\textsuperscript{256} provided grants between 2004-2006 for the building of storage equipment in an area of 51 664 m\textsuperscript{3} for liquid manure and slurry and 55 281 m\textsuperscript{3} for solid manure, including in Wielkopolskie (Polish Ministry of the Environment, 2010). This program does not exist anymore. The Rural Development Program (RDP) 2009-2013 offered subsidies for the investments in the modernisation of animal production farms, contributing up to 75 % of the costs of investments undertaken for the implementation of the Nitrates Directive (Korczyńska, et al., 2013). The new Rural Development Programme 2014-2020 is much weaker in terms of water protection measures. This is caused by the transfer of 25 % of funds from pillar II of the Common Agricultural Policy into pillar I (Kalinowska, 2014; Institute for European Environmental Policy (IEEP), 2015). Still, the public support for farm investments amounts about € 2.5 billion (European Commission, 2014b), although it represents a decrease of 22 % of the budget compared to the previous RDP (European Commission, 2014c). Additional financial programmes at national or regional level may encourage farmers to increase their storage capacities. While the size of holdings has increased in the past years in the region, a particular attention should be paid to not disproportionately favour bigger farms when providing financial aid (Nitsch, 2003). According to the participants of the regional conference, farmers may also need more information on the availability of financial aids.

**Cover the solid manure stored in heaps**

The majority of the storage equipment for liquid manure and slurry was already covered in Wielkopolskie in 2010 (98 % and 95 % of the tanks, respectively) while this is not the case for solid manure (Eurostat, 2013c). Thus, the possibility for improvement is high for solid manure (Nitsch, 2003) that is mostly stored on the ground in heaps (Dach, et al., 2012; Sobolewski, 2011; Nitsch, 2003). Covering solid manure heaps results in a significant decrease of ammonia emissions and losses of nutrients by run-off in the case of rain. However, this measure may be costly, depending on the type of cover and the frequency of manure removal since the removal of the cover induces additional labour time. Therefore, this measure is mainly suitable for large farms.

**Increase the height of manure heaps**

The stakeholder consultation of this project suggested another way to decrease ammonia emissions through increasing the height of the manure heaps. Thus, the compaction of the manure under its own weight decreases the amount of manure in contact with air which reduces volatilisation. Increasing the height of the heaps (4-6 m) requires proper equipment, such as a telescopic loader (Krysztoforoski, 2014). This method is a less effective alternative to manure heaps coverage. However, it is easier to implement and relevant if farmers cannot afford to implement manure coverage.

**Compost manure**

Composting would allow the volume of stored manure to be decreased. Manure could also be easily transferred and sold to others farms. In the case of pig manure, straw needs to be added to obtain the

\textsuperscript{256} Special Accession Programme for Agriculture and Rural Development, managed by the European Commission under the Agenda 2000 (European Commission, 2001).
C/N ratio required for optimal microorganism activity. In addition, the application of compost would increase the carbon content of soil, which would generally be beneficial in the region.

In Poland, composting is commonly done through the use of an aerator on the farm or as an external service. In Poland, solid manure can be easily composted by storing it on the ground and providing regular aeration. This last step may require specific machinery. The cost of an aerator for usage in domestic production is estimated at € 0.15/t of processed manure in Poland (instead of € 0.8-1.2/t of manure processed for Western Europe). With costs of around € 7 200 for an aerator (instead of € 15 000 in Western Europe), the use of such form of composting might be cost-effective for many farmers (Baltic deal, 2013).

This measure has been identified by the participants of the regional conference as a key good practice that should be further subsidised.

**Cool slurry**

Cooling slurry highly decreases ammonia emissions, implying an increase of the amount of nutrients that can be applied on field and thus possible savings due to less purchase of fertilisers. This measure is particularly appropriate for the region considering the important pig production that produces a high quantity of slurry (84 % of total manure produced (Agro Business Park, 2011)). This measure requires slurry to be stored in a tank, in order to be able to control the temperature (through the installation of an appropriate cooling system).

The high cost of devices has been identified by the participants of the regional conference as a barrier for the implementation of this measure. However, the costs can be reduced. Indeed, farmers can do a lot of work by themselves (Krysztoforoski, 2014) without external workforce. They can adapt the existing storage facilities by isolating the facilities and building the cooling system, which would be less costly than purchasing an entire new cooling system. In addition, the installation of heat exchangers, which is relatively simple, can allow recuperating energy to heat home and livestock buildings, heat water or dry crops (Krysztoforoski, 2014). The system is the most economically viable for farms with a high slurry production. A high quantity of slurry to be cooled would produce a high quantity of energy and consequently sufficient savings to compensate the purchase of materials for the installation. No example of implementation of this measure in the region has been found.

### 5.8.4.3 Good practices to reduce nutrient losses in crop production

**Extend and improve the N and P fertilisation management plans for all farms**

In addition to the requirements of the action programme implementing the Nitrates Directive in NVZs, the Polish Code of Good Agricultural Practices contains further provisions to protect water. It provides advice on elaborating a fertilisation plan, in particular on rates and timing of application of fertilisers and the most appropriate application techniques. Currently, only large farms (> 100 ha UAA) within the NVZ and pig and poultry farms with more than 100 LSU outside the NVZ must prepare a fertilisation management plan. The requirement should be extended to all farms of the NVZs and cattle farms outside the NVZ. Extending the requirement to the entire territory would further reduce nutrient losses and on the other hand, increase the farmers’ income by reducing the need for purchased fertilisers.

Farmers that participated in the regional conference indicated that they draft fertilisation plans every year. However, they also mention lack of information on the preparation of a fertilisation plan. Local associations and farm advisory systems from public bodies are the key to the success for the development of such fertilisation strategies at farm level trough their role in the diffusion of the information and the development of training. Capacity building would probably improve the uptake of this measure. Demonstration farms and highlighting the economic savings are also a good way to
improve farmers’ knowledge as proposed by the Wielkopolska Agriculture Advisory Centre in Poznań and in the context of the Baltic Deal project.

Soil content was monitored within the framework of the State Environmental Monitoring 2010-2012 which included N, P and K content (Jagusiewicz, 2009). However, soil analyses are only conducted every three years, which does not allow an accurate calculation of the nutrient budget. Taking soil samples each year in early spring before fertilisation or in autumn after harvest would allow to accurately prepare the fertilisation plans. Although the participants of the regional conference considered the frequency as insufficient, individual soil analysis in laboratories are not considered as an option by farmers, considering their costs, especially for small farms. Testing plant nutrient requirements is also needed to balance the nutrient budget. Since the tests are costly, feedback from the regional conference suggest that encouraging the rental of analysis equipment or collective purchase of testing devices could be a solution, although even this can be too costly for small farms.

Lastly, while elaborating a fertilisation management plan, farmers should also consider the soil pH. If the soil is very acidic, which is the case of many areas in the region, farmers should consider increasing the soil pH to increase nutrient assimilation. The effect of liming on crop assimilation and yield has been proved in many European regions (Decoopman, 2011; Fabre & Kockmann, 2002). However, balancing the lime input is required to avoid any reverse effect due to a too high pH and nutrient losses in water that may affect human health.

**Use the most suitable application technique**

In Poland, most farmers appear to spread slurry with vacuum tanker and splash plate (Dach, 2015; Borkowski, 2014). No information on the application techniques for solid manure were found. Considering the high share of arable area in Wielkopolskie, improving the application technique could have a strong impact on ammonia emissions. In addition, all of the application techniques can be used in the region. Thus, the use of rear discharge spreader for solid manure application and band spreading or injection of slurry should be promoted, considering the significant ammonia emissions reduction. However, considering the type of soils in Wielkopolskie, very careful use is required of these machines to avoid the leaching of a high amount of nutrients. Therefore, these techniques can only be used by farmers that already have a very good knowledge in the development of fertilisation management plan. Since some of the application techniques may require significant investment to purchase the specific equipment, this practice may only be possible for the largest farms. Encouraging collective action could also be a good way to allow such investment to smaller farms.

**Favour conservation tillage to reduce nutrient leaching and erosion**

Wielkopolskie is affected by tillage erosion and wind erosion. Tillage erosion is responsible for soil loss resulting in the loss of nutrients, in particular P. Note that soil losses are also particularly high during sugar beet harvesting. Favouring conservation tillage or no tillage helps to reduce the loss of nutrients due to tillage. Conservation tillage also improves soil structure, increasing the soil water retention and thus increasing nutrient retention. It also decreases the decomposition of residues and thus increases the available nitrogen for crops. In addition, due to the type of soil which is mostly sandy in Wielkopolskie, it is expected that tillage will have only a limited effect for reducing the nutrient losses by leaching or emissions to air (Rochette, 2011). Hence reducing tillage will not increase the N₂O emissions from soils.

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257 The Baltic Deal project ended in 2015. It has been renewed and the new project is called the Baltic Deal BRIDGE.
This measure may only concern large holdings with few or no livestock. Indeed, this measure cannot be implemented in holdings with livestock production since farmers that use manure as a fertiliser need to plough down the manure in soil. Moreover, reduced tillage may require specific equipment that only the largest farms (i.e. over a thousand hectares) can afford. For farms with a small acreage the purchase of this type of equipment is too expensive (Karaczun, et al., 2008). The largest farms represent less than 1 % of the total holdings in Wielkopolskie while accounting for 26 % of the UAA (Eurostat, 2014a).

The national action programme implementing the Nitrates Directive requires that manure is incorporated within 24 hours (Nitsch, 2003). No information on the compliance of this measure by the regional farmers is available. Considering the high potential of improvement of this measure, further investigations and survey campaigns can be relevant to help the authorities identify whether this measure is worth encouraging.

Use digested manure from biogas production to increase the assimilation of nutrients by crops compared to undigested manure

Anaerobic digestion could provide additional income to farmers and a stable digestate that can easily be transported and sold. Currently, anaerobic digestion of manure is rarely used in Poland. Energy use from biogas amounted only to 0.08 % of the Polish energy pool in 2010 (Luostarinen, 2013a). In 2012, the energy from biogas was produced in 178 biogas plants, 21 of them processing manure in 2010 but not one was in the Wielkopolskie region (Grzelak, et al., 2012). The Baltic Sea Action Plan sets a target that 25 % of the manure should be transformed into biogas by 2025. In this context, Poland is launching one of the world’s largest programmes, constructing nearly 2 500 large biogas plants (in the range of 1-3 MWe), supported by several thousand smaller biogas plants (50-200 kW) (Dach & Pilarski, 2012). The Polish Ministry of Economy has stated the objective of building 2 000 agricultural biogas plants by 2020 (Karaczun, 2010; Luostarinen, 2013a).

Wielkopolskie shows the highest energy potential in Poland for the development of production of biogas from manure and from energy crops with an estimated to 95 MWe, i.e. 25 % of the biogas energy from manure that could be produced in Poland (Luostarinen, 2013a). Hence the development of new biogas plants is favourable in the region. Although the potential of energy production in Wielkopolskie is high, not all the manure produced is available for the production of biogas since the manure is also used as a fertiliser. Moreover, the available manure is mostly in a solid form. Solid manure can only be digested as a co-substrate in the co-digestion of slurry. Hence, only 7.4 % of the total manure can be used for biogas production at the national level (Luostarinen, 2013b).

Biogas production from manure is technologically relatively simple as the processes are the most developed for such materials. On the other hand, it requires transportation of a lot of water and other substrates such as crop residue in the slurry in case of farm co-operative biogas plants, which makes the profitability of manure-based biogas production a challenge. The use of slurry without other substrates may not be viable considering that it requires large digester volumes while the biogas production per digester volume is low (Luostarinen, 2013a). In Wielkopolskie, most of the biogas produced would originate from solid pig manure and poultry manure, emphasising the need for suitable technological solutions for solid manure and high nitrogen content (Luostarinen, 2013b). In addition, since crop residues have a higher methanogenic power than manure, farmers may give priority to the production of crops for energy production. However, it may lead to an increase in the production of intensive crops such as corn for silage or land use change from grassland to highly intensive crop land, using more fertilisers and risking over-fertilisation to increase the yield. In addition, it should be highlighted that the removal of crop residues for the production of bioenergy may result in a deficiency in nutrients, inducing the need for the application of fertiliser.
An issue is the fact that Poland, and Wielkopolskie in particular, is massively covered by Natural Protected Areas (such as National Parks or Natura 2000 areas) and urban areas, which results in a difficulty to find suitable locations for plants. Moreover, the support of the administration is necessary to obtain the expected results (Dach & Pilarski, 2012).

In Poland, the average cost of 1 MW is € 3.6 million. Considering that the co-financing represents 50% of the investment costs in Poland, the overall investment costs amount to € 9 billion. In Wielkopolskie, only private companies may be able to invest such a high amount. They may then consider renting the use of the digestion plant to farmers. The Bank for Environmental Protection in co-operation with the Wielkopolskie Fund for Environment Protection and Water Management proposes preferential credits for the investments in biogas production plants up to 2MW (Korczyńska, et al., 2013).

**Cover with catch crops and legumes to reduce nutrient leaching**

This measure is particularly adapted to the region because the effect of catch crops is larger in sandy soils than in clay soils, although the effect of catch crops is higher when catch crops are grown on land that has more than 0.8 LU / ha which is not the case of Wielkopolskie (Børgesen & al, 2013). Covering and catch crop application appears to be a popular measure in Poland that is already implemented on grasslands. The farmers believe that cultivation of groundcover crops for green fertiliser, utilisation of harvest residues and laying land fallow are the easiest techniques to implement (Karaczun, et al., 2008).

The use of legumes can be relevant as an additional source of nutrients, replacing mineral fertilisers for soils with poor nutrient content or subject to high nutrient losses, which is the case in the region considering the podzolic soil. However, it is important to ensure that the ground is not left fallow following the incorporation of legume residues since they are a high source of nitrogen that can result in rapid losses in case of excess of N. The use of legumes is already widespread in Poland where legume growing in mixtures with cereals is characteristic of the national agriculture. In Poland, the purpose of mixing legumes and cereals is to optimise the use of spatial, temporal and physical resources both above- and below ground.

Hence, while this measure has a highly positive effect on nutrient leaching, it is already well implemented in the region (Kus & Matyka, 2007). The potential further development probably only concerns a small share of the regional farms. Also note that from 2015, areas with nitrogen-fixing crops were chosen by Poland as Ecological Focus Areas (EFA) in the new Common Agricultural Practice (CAP (Polish Ministry of Agriculture and Rural Development, 2015). Note that the requirements related to the EFA do not concern farms with an area below 15 ha of arable land.

**Incorporating straw**

Considering the regional soil type, incorporating crop residue is of great interest (Krysztoforoski, 2014) since it enriches soils in organic matter and has a positive impact on soil structure and soil water retention capacity, which in turn reduce the risk of nutrient leaching. Residues such as straw have high C/N ratio and help to immobilise nitrogen that is slowly released. By improving the nutrient content of the soil, the incorporation of residues also avoids or reduces the use of additional fertiliser and thus results in savings.

In Wielkopolskie, it was estimated that no farm incorporated straw in 2005. However, this practice were well developed in some other Polish regions (Kus & Matyka, 2007). This measure can have a high effect on the global regional nutrient loss since 84% of the region’s UAA is arable land. In Poland, ploughing under is the most common solution for utilisation of harvest residues. Farmers apply this type of practice in order to enrich their soils and due to a lack of alternative uses of the straw considering the size of the cattle and the breeding systems (Karacznun, et al., 2008). This is particularly the case in Wielkopolskie that produce a high quantity of pigs that do not use straw for bedding.
Construct sedimentation ponds to retain nutrients

This measure could help to reduce the nutrient concentration in groundwater and surface water. Small sedimentation basins retain soil particles and nutrients from run-off water. The nutrients are retained by the soil particles, absorbed by the plants or emitted to air in case of nitrogen. The sedimentation ponds are particularly suited to capture phosphorus bonded with soil particles. The effect on dissolved phosphorus or nitrogen is lower, except if they are assimilated by plants.

The pond can be constructed by widening a section of a ditch (Baltic Compass, 2012). It can be small: if it is placed close to the phosphorus source, the amount of water passing is smaller and the area needed for the wetland decreases (Kynkäänniemi, 2013). The flow of run-off may require to be slowed down by increasing soil level along ditches to block the pathway of the water. If the area is highly sloped, a second wetland filter can be added to increase the sedimentation efficiency. The measure does not require specific investments or technical knowledge (Juszczak, et al., 2007). Additional costs for labour to increase the soil level or widen a section of a ditch should be considered. Moreover, depending on the land price for the area taken out of production and the resulting income loss from not harvesting crops there, costs may vary. Once established, the sedimentation ponds require little maintenance. The accumulated sediments need to be regularly removed. They can be applied to field as a phosphorus-rich fertiliser. This measure can be combined with conservation tillage and vegetative filter strips at the end of the field to reduce the nutrient losses (Baltic Compass, 2012).

Recommended by the project Baltic Deal that aims at proposing best agricultural practices in the Baltic Sea region (Baltic Deal, 2015), sedimentation ponds are currently rare in the country. From 2002 to 2012, around 300 ponds have been constructed in Poland. The construction of small and shallow wetlands in forests was supported by the Small scale water retention” project 2007-2014 (Heeb, et al., 2012; Worzyl & Wyra, 2014). No information was available on a possible renewal of this project. Currently, ponds with a surface area of less than 100 m² and ponds with vegetation are considered as an Ecological Focus Area in the new CAP in Poland (Polish Ministry of Agriculture and Rural Development, 2015).
6. Conclusion

Closing mineral cycles is a real challenge for farmers. Significant efforts have been made to reduce the impacts of nutrient losses on the environment and health over the past few years, but practices can still be improved.

A large panel of good practices exist and there is no ideal solution, i.e. a practice that is low-cost and easy to implement, with a high potential to reduce the losses for all nutrients. Good agricultural practices can be more or less effective according to the regions and also within a region. The effects of a practice and a combination of practices on the environment and human health depends on the regional farming production systems, the choice of agricultural practices, and the intensity of production. It also depends on local environmental conditions such as soil and climate conditions. Thus, farmers have to choose a combination of practices by considering their possible effects on each nutrient cycle (N, P, K but also other nutrients), their technical feasibility and their cost for the farm in the shorter and longer terms.

Certain good agricultural practices were proposed for the majority of the studied regions. For instance, this is the case of the implementation and the improvement of fertilisation management plan, manure processing or the use of best suited application techniques. However, as shown in the case studies, a good practice does not necessarily fit all the EU regions, because it may not be relevant in terms of impacts on the nutrient losses, it may be too costly or have too many technical barriers or it may simply be already well implemented. For instance, drip irrigation was recommended for Italy but it is less relevant in Murcia since this practice is already widely implemented. Manure composting or the increase of the height of manure heaps were found to be good practices in Wielkopolskie while slurry tank coverage and denitrification/nitrification process were proposed in Brittany.

Some innovative practices such as the use of struvite or the reuse of waste water from agricultural drainage water appear to have a high potential to close the mineral cycles in some regions. However, these practices are not widely implemented at farm level within the EU, because they may be still at a research or pilot level. Thus, further research is necessary to propose new practices that would fit to farmers with a reasonable cost. In addition, knowledge and technological support to farmers as well as collective actions promotion are required to extend the use of promising technologies and initiatives. For instance, precision farming initiatives, such as the use of Geographic Information Systems (GIS) proposed by Teagasc in Ireland to simply manage fertilisation plans should be further developed.
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