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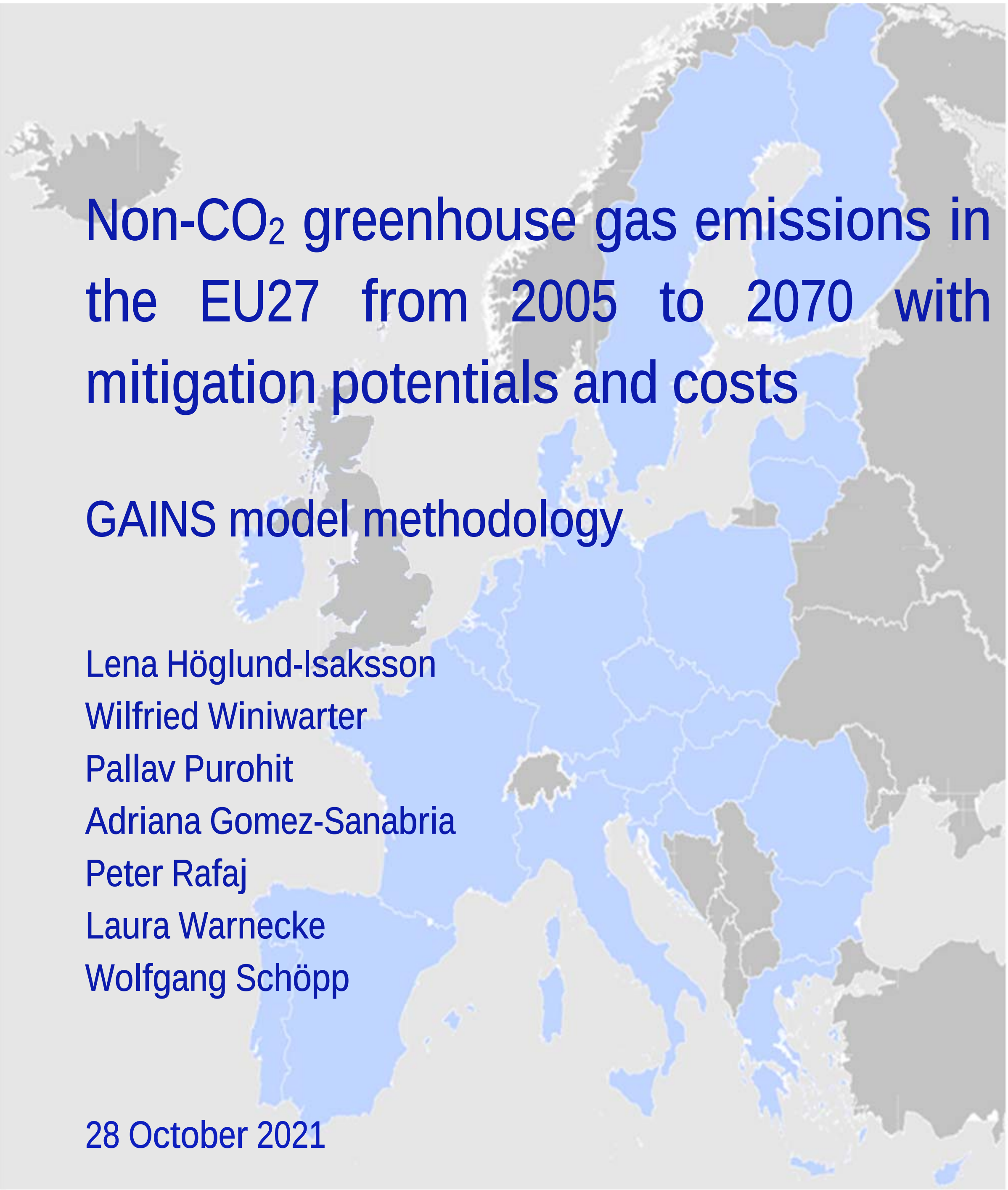

# Non-$CO_2$ greenhouse gas emissions in the EU27 from 2005 to 2070 with mitigation potentials and costs

## GAINS model methodology

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## List of Abbreviations

| | |
|---|---|
| API | American Petroleum Institute |
| CAPRI | Common Agricultural Policy Regionalized Impact (agro-economic model by University of Bonn) |
| CDD | Cooling degree days |
| CRF | Common Reporting Format (of official reportings of national greenhouse gas inventories to UNFCCC) |
| CWPB | Center-worked prebake (specific technology in aluminium production) |
| DG-ECFIN | Directorate General for Economic and Financial Affairs |
| DPI | Dry powder inhaler |
| EEA | European Environment Agency |
| EFMA | European Fertilizer Manufacturers Association |
| EHPA | European Heat Pump Association |
| EIA | Energy Information Administration |
| ESIA | European Semiconductor Industry Association |
| ETP | Energy Technology Perspective |
| ETS | Emissions Trading System |
| EU | European Union |
| EU-27 | 27 member countries of the European Union |
| FAO | United Nation's Food and Agriculture Organization |
| FAOSTAT | Food and Agriculture Organization's statistical database |
| FBC | Fluidized bed combustion |
| GAINS | Greenhouse Gas and Air Pollution Interactions and Synergies (integrated assessment model by IIASA) |
| GHG | Greenhouse Gas |
| GIS | Gas Insulated Switchgear |
| GSHP | ground source heat pumps |
| GWP | Global Warming Potential |
| HFO | Hyrdofluoroolefins |
| IIASA | International Institute for Applied Systems Analysis |
| IEA | International Energy Agency |
| IFA | International Fertilizer Association |
| IPCC | Intergovernmental Panel on Climate Change |
| LABORSTA | Labour Statistics database from International Labor Organization |
| MAC | Mobile air-conditioners |
| MDI | Metered dose inhaler |
| NTUA | National Technical University of Athens |
| OECD | Organization for Economic Co-operation and Development |
| PCAs | Pollution Control Agreements |
| PFPB | Point-feeder prebake (specific technology in aluminium production) |

| | |
|---|---|
| PRIMES | Modelling system for energy supply and demand in EU-28 (by NTUA) |
| SWPB | Side-worked prebake (specific technology in aluminium production) |
| TEAP | Technology and Economic Assessment Panel of UNEP |
| TIMER | Targets IMage Energy Regional (simulation model by Netherlands Environmental Assessment Agency) |
| UNEP | United Nations Environment Programme |
| UNFCCC | United Nations Framework Convention on Climate Change |
| USEPA | United States Environmental Protection Agency |
| USGS | United States Geological Survey |
| VAM | Ventilation air methane |
| VSS | Vertical stud Söderberg (specific technology in aluminium production) |
| XPS | Extruded polystyrene |

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# 1 Introduction

This report presents the methodology for constructing European Union (EU) non-$CO_2$ emission scenarios in the Greenhouse gas, air pollution interactions and synergies (GAINS) model developed by the International Institute for Applied Systems Analysis (IIASA). The described methodology refers in particular to the EU 2020 Reference scenario for non-$CO_2$ greenhouse gases (GHGs), their future mitigation potentials and associated costs projected for the EU-27 countries in five-year intervals to 2070. The work forms part of the EUCLIMIT5 project[1], which aims at producing long-term projections for all emissions of GHGs in the EU-27 consistent with the European Commission's latest available macroeconomic and population projections. The 2020 Reference scenario with underlying assumptions has been published in EC (2021). This report provides further detailed information on the GAINS model methodology for the projections of the non-$CO_2$ GHGs, i.e., methane ($CH_4$), nitrous oxide ($N_2O$) and three groups of fluorinated gases (F-gases) viz. hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride ($SF_6$).

Throughout the text in this report there are references made to a "baseline" scenario. Please note that in this context a "baseline" non-$CO_2$ scenario refers to any scenario that describes emissions under a continuation of the current legislation implemented to control non-$CO_2$ emissions. Hence, a non-$CO_2$ "baseline" takes the energy sector input data from the PRIMES model as given, also when the PRIMES energy input data describes a full decarbonization of the energy sector. The non-$CO_2$ "baseline" scenario describes the starting point for assessing the technical and behavioral mitigation potentials in non-$CO_2$ "mitigation" scenarios.

The report is structured as follows. Section 2 presents the general GAINS methodology for estimating non-$CO_2$ greenhouse gas emissions for EU-27. Sections 3, 4 and 5 describe in detail the methodology applied for estimation of emissions by source for $CH_4$, $N_2O$ and F-gases, respectively. Finally, Section 6 provides a comparison between emissions reported by member states to the UNFCCC (Dec, 2019) for year 2015 and the emissions estimated by the GAINS model for the same year.

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[1] The EUCLIMIT5 project is financed by the European Commission under Service Contract N° 340201/2019/813567/CLIMA.C.1 "Model based Evaluation of EU Climate Policies".

# 2 GAINS emission estimation methodology

## 2.1 Estimation of emissions

In the general GAINS methodology (Amann et al. 2011), emissions from source $s$ in region $i$ and year $t$ are calculated as the activity data $A_{its}$ times an emission factor $ef_{ism}$. If emissions are controlled through implementation of technology $m$, the fraction of the activity controlled is specified by $Appl_{itsm}$, i.e.,

$$E_{its} = \sum_m [A_{its} * ef_{ism} * Appl_{itsm}], \tag{1}$$

where $ef_{ism} = ef_{is}^{NOC} * (1 - remeff_{sm})$and$\sum_m Appl_{its} = 1$, (2)

and where

| | |
|---|---|
| $A_{its}$ | is the activity (e.g., number of animals, amounts of waste), |
| $ef_{ism}$ | is the emission factor for the fraction of the activity subject to control by technology $m$, |
| $Appl_{itsm}$ | is the application rate of technology $m$ to activity $s$, |
| $ef_{is}^{NOC}$ | is the no control emission factor for activity $s$, and |
| $remeff_{sm}$ | is the removal efficiency of technology $m$ when applied to activity $s$. |

Hence, for each emission source sector, country and year specific sets of application rates for all the possible technologies (including no control among the possible technologies) are defined such that application rates always sum to unity.

## 2.2 Activity data

In GAINS, activity drivers for emission projections enter calculations externally using projections from different internationally recognized sources. For EU projects, the GAINS model uses energy scenarios produced by the PRIMES model (E3Modelling, Athens) and agricultural scenarios from the CAPRI model (EuroCare, Bonn). These activity scenarios are produced in consistency with the macroeconomic scenarios published in the European Commission's 2021 Ageing report (EC, 2021a). These are also the basis for deriving activity data in GAINS, which are not part of the input data provided by the PRIMES or CAPRI models (i.e., activity data for the waste, wastewater and F-gas source sectors).

## 2.3 Emission factors

The choice of emission factors for estimation of non-$CO_2$ greenhouse gas emissions in GAINS follows the methodology recommended in IPCC reports and guidelines (IPCC, 2006, 2013 and 2019) as closely as available data allows. This includes conversion to $CO_2$ equivalents using Global Warming Potentials (GWP) without climate-carbon feedbacks from the Fifth Assessment Report (AR5) of 28 times that of $CO_2$ for methane, 265 times that of $CO_2$ for nitrous oxide, and various species specific GWPs for the various F-gases. With the ambition to produce as consistent estimates across countries as possible, an extensive survey and compilation of available country-specific information on parameters with significant effects on emissions was undertaken. For several emission sources, there is enough country-specific information available to derive emission factors that are closer to IPCC Tier 2 factors than Tier 1 default factors. Emission factors that are derived from underlying country-specific information improve consistency in estimates across countries and provide an opportunity to better understand inconsistencies and uncertainty in emission estimates reported by countries to the UNFCCC and other inventories.

## 2.4 Baseline assumptions on emission control

Within the EUCLIMIT5 project, the GAINS baseline emission scenarios reflect current and future impacts on emissions from legislation adopted as of Dec 2019.

## 2.5 GAINS model results and nationally reported emission data

The principal difference between GAINS model estimates and those reported by member states to the UNFCCC is that the GAINS model for given circumstances and technological settings applies a consistent methodology across all countries, whereas estimation methodologies applied by countries often differ in various respects across countries. Discrepancies are carefully investigated and adjustments made when appropriate, i.e. to the extent that the consistency in methodology across countries is preserved. For $CH_4$ and $N_2O$, any remaining deviations in total country emissions according to the Common Reporting Format (CRF) tables as of December 2019 and GAINS model results have been included as “calibration residual” for the year 2015, such that totals fully agree for this year. The calibration factors are projected into the future proportionally to the growth in total emissions by gas, as projected in GAINS before calibration. The calibration factors reflect minor emission sources specific for individual countries and not covered in the GAINS model structure, but can also be due to a use of different methodological approaches when deriving emission factors at the sector level. For F-Gases, such a final calibration to reported total emissions for 2015 has not been conducted due to a large unexplained variation in reported sectoral leakage rates across countries and due to incompleteness in national reporting for some sources and countries.

## 2.6 Non-$CO_2$ GHG mitigation and cost estimation in GAINS

### *2.6.1 Technical mitigation potentials*

The mitigation potential assessed in the marginal abatement cost curves of the GAINS model refers to feasible reductions in emissions through adoption of mitigation technologies defined as installations or applications of physical equipment or material, or modifications in physical parameters affecting emissions. It should be noted that the technical mitigation potential may differ from the politically feasible mitigation potential as the latter also takes into account costs and political barriers for implementation. In the GAINS baseline scenarios for non-$CO_2$ greenhouse gases, no effects on costs and removal efficiencies from technological development are accounted for. The reason is that incentives to adopt (and therefore further develop) technology that will reduce non-$CO_2$ greenhouse gases are to a large extent driven by the introduction of policies directed specifically at non-$CO_2$ mitigation. An exception would be technologies that simultaneously reduce methane emissions and recover or save gas that can be utilized for energy purposes. Demand for the latter technologies may arise spontaneously if the future price of gas is expected to be high enough to make gas recovery profitable in the long run. Hence, in contrast to most $CO_2$ mitigation opportunities, where demand for adoption and development of more energy-efficient technologies is primarily driven by a wish to save on energy costs, there is generally less reason to invest in development of improved mitigation technologies for non-$CO_2$ greenhouse gases (since most of them have only additional costs and do not generate revenues – with exceptions such as anaerobic digestion) unless directed policies are put in place and maintained for a foreseeable future. The development of non-$CO_2$ abatement technology therefore rely heavily on the stringency of the policies implemented and their effectiveness in stimulating continuous technological development. E.g., policies that merely require adoption of pre-defined best-available-technology are less likely to stimulate continuous technological development than policies that financially penalize every additional unit of emissions released (see e.g., Popp 2003; Pizer and Popp 2008; OECD 2011). The existence of incentives for continuous technological development of non-$CO_2$ abatement technology is therefore considerably more uncertain than for $CO_2$ mitigation technology.

A common way to include the effect of technological change in mitigation technology in assessment models is to make assumptions about the learning effect on costs from cumulative technology adoption (see e.g., Jamasb and Köhler 2007). In such experience curves, the learning effect is usually measured in terms of a percentage reduction in unit costs *C* for each doubling of the cumulative capacity installed *Cap*, i.e.,

$$C = \alpha \times Cap^{-\varepsilon} \tag{3}$$

$$LR = 1 - 2^{-\varepsilon} \tag{4}$$

where $\alpha$ is a constant, $\varepsilon$ is the learning elasticity and *LR* is the learning rate.

Jamasb and Köhler (2007) survey the literature on experience curves to sample empirical estimates of learning rates. They conclude that the variability is very large both between technologies and sectors and for different time periods. A majority of published learning rate estimates refer to development in electricity generation technologies. McDonald and Schrattenholzer (2001) survey these technologies and find empirical measurements of learning rates ranging from 1.4 to 35 percent reduction in unit costs for every doubling in the cumulative technology capacity. They conclude that a 17% learning rate would be appropriate for long-term energy models. However, Jamasb and Köhler (2007) point out that the evidence on declining learning rates over time suggests that a 20 percent learning rate might be on the high side when taken as a general assumption. This is particularly true when modeling further development of already existing technologies. For non-$CO_2$ mitigation technology we adopt effects on costs from a 15 percent learning rate, which in Equation 4 implies a learning elasticity of 23.4% and a doubling of the adopted capacity every 15 years. It is further assumed that the effect of technological development on costs lasts for 30 years after policy incentives have been introduced.

Depending on whether the estimates available from literature on costs of specific technologies are representative of the situation today or reflect costs expected in the future, the starting year for the learning effect on costs differs between technologies. Table 1 presents the adjustment factors applied to investment and operation and maintenance costs for two categories of technologies. The first category refers to existing technologies that can be implemented already from the year 2025 and for which current costs are fairly well known. For this category, an effect of technological development on costs is expected from 2025 onwards. The second category refers to technologies that are described in literature as still under development (i.e., reduction of $CH_4$ emissions from enteric fermentation through breeding or changes in animal feed) or to technologies for which cost estimates from literature are stated as expected future costs (i.e., options to control $N_2O$ emissions from agricultural soils). For these options an effect on costs from technological development is accounted for from 2035 onwards.

Table 1: Adjustment factors for investment costs, operation and maintenance (O&M) costs, and emission factors used in GAINS to account for effects of technological development of non-$CO_2$ mitigation technology from 2025 to 2070.

| | Category 1 technologies | | Category 2 technologies | |
|---|---|---|---|---|
| | Technologies with current costs and removal efficiencies provided in literature. Include all technologies except those specified as Category 2 | | Technologies with future expected costs and removal efficiencies provided in literature. Includes CH4 measures addressing enteric fermentation emissions (i.e., breeding and animal feed options) and N2O measures addressing emissions from agricultural soils (i.e., VRT, nitrification inhibitors and precision farming) | |
| Year | Investment and O&M costs | Emission factors | Investment and O&M costs | Emission factors |
| 2025 | 1 | 1 | 1 | 1 |
| 2030 | 0.88 | 0.95 | 1 | 1 |
| 2035 | 0.82 | 0.9044 | 1 | 1 |
| 2040 | 0.765 | 0.8601 | 0.88 | 0.95 |
| 2045 | 0.72 | 0.8179 | 0.82 | 0.9044 |
| 2050 | 0.68 | 0.7778 | 0.765 | 0.8601 |
| 2055 | 0.651 | 0.7397 | 0.72 | 0.8179 |
| 2060 | 0.651 | 0.7397 | 0.68 | 0.7778 |
| 2065 | 0.651 | 0.7397 | 0.651 | 0.7397 |
| 2070 | 0.651 | 0.7397 | 0.651 | 0.7397 |

Apart from lowering costs, technological development in abatement technology can also come in the form of improved removal efficiency. The effect on removal efficiency is closely linked to whether or not policies target emissions directly, e.g., through emission charges. Command-and-control regulations which require installations of best available technology are poor stimulators of improvement in removal efficiency, while market-based instruments that are directly linked to emission reductions are more likely to stimulate improvements in the removal efficiency (Popp, 2003). Empirical analyses of how environmental regulations affect technological development through effects on application rates for abatement technology patents usually show “peak” patterns, i.e., a sharp increase in technological development activities just after the introduction of a regulation, then a high level of development for a few years, and finally a leveling off reflecting that much of the readily available opportunities for technological development have been exhausted (OECD, 2011). Similar “peak” patterns have also been found for improvements in removal efficiencies following introductions of incentive-based environmental regulations (Popp, 2003; Höglund-Isaksson and Sterner, 2010). As no information is available in the literature for control of non-$CO_2$ emissions per se, we survey the literature for known effects on patent application rates and removal efficiencies of technologies in response to waste management regulations and air pollution control. Similar to non-$CO_2$ control, development of waste management and air pollution technologies relies to a high degree on implementation of directed policies to drive incentives for technological development.

OECD (2011, p.30) shows how the number of patent applications for solid waste management technologies (recycling, composting and incineration) closely responds to the timing of regulation introductions in different countries. Denmark was the first country to adopt an environmental protection law in 1973 and an important objective was to control the amount of waste deposited to landfills when land area was becoming scarce (Copenhagen Cleantech Cluster 2012; Civilstyrelsen 2012). Danish patent applications in this sector started to rise sharply in the beginning of the 1980s, peaked in the early 1990s and have since declined (OECD 2011). Similar increases in patent applications for waste management technology occurred in the US in the late 1980s and in Japan in the early 1990s following the amendment of the US Resource, Conservation and Recovery Act in 1984 and the Japanese Recycling Law in 1991, respectively (OECD 2011, p.131).

From the early 1990s, the US and Sweden introduced market-based instruments complementing or replacing command -and control regimes in air pollution control. As shown by Popp (2003), the command –and control regulations of $SO_2$ emissions from coal-fired utility plants in the US introduced under the Clean Air Acts of 1970 and 1977, are reflected in a considerable increase in US patent applications for scrubbers during this period. Notable is that this technological development was primarily focused on developing technology at lower costs and without significant improvements in removal efficiencies. Only after the 1990 Clean Air Act, when a cap- and trade scheme for $SO_2$ emissions was introduced, did the removal efficiency of the available technologies improve significantly. Popp (2003) interprets this as a result of a switch in policy target from targeting adoption of certain technology to targeting emission reductions directly. Table 2 summarizes the empirical findings of the short and long-term effects of air pollution regulations in the US (Popp, 2003), Sweden (Höglund-Isaksson and Sterner, 2010) and Japan (Matsuno et al., 2010). While Popp (2003) measures technological development directly when he measures the annual improvement in removal efficiency of a certain technology (scrubbers), the other studies use indirect measures, where the effect of technological development on emissions is summed up with the effect on emissions of adopting existing technology. The latter produces higher annual changes in emission rates, in particular in the short run as it then also reflects the plants' initial adoption of technology in responses to the regulations. In the long run, it can be expected that a larger fraction of the annual change is due exclusively to the effects of technological development.

On the basis of the long-term effects found in the literature survey presented in Table 2, we assume a long-term improvement in the effectiveness of non-$CO_2$ mitigation technology to remove emissions of one percent per year. The resulting adjustment factors applied on emission factors are presented in Table 1 for Category 1 and 2 technologies, respectively. Like for costs, the effect on removal efficiencies from technological development are assumed to last for 30 years after implementation of policy incentives.

Table 2: Some empirical evidence of the effects of air pollution regulations on targeted emissions in short and long runs.

| Source | Regulation | Year of first imple-mentation | Number and type of plants in analyzed sample | Emission parameter description | Total effect | | Short-term effect | | Long-term effect | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | Entire period | Average annual change in parameter | Period with rapid decline in parameter | Average annual change in parameter | Period with slower decline in parameter | Average annual change in parameter |
| Popp (2003) | US 1970, 1977, 1987 CAA (command -and control) | 1970 | 193 coal-fired plants with scrubber | $SO_2$ emission rate of new scrubbers | 1974-1990 | about -0.4% | 1974-1978 | about -1.5% | 1978-1990 | about 0 |
| | US 1990 CAA (cap -and trade) | 1990 | | | 1990-1997 | about -1.6% | not available | | not available | |
| Höglund-Isaksson and Sterner (2010) | Swedish NOx charge -plants not (yet) adopting abatement technology | 1992 | 180 to 280 power plants (> 50 MWh) targeted in 1992 to 2007 | $NO_X$ emissions per energy output of targeted plants | 1992-2007 | -3.5% | 1992-1997 | -5.8% | 1997-2007 | -2.4% |
| | Swedish NOx charge -plants having adopted abatement technology | 1992 | | | 1992-2007 | -5.7% | 1992-97 | -11.2% | 1997-2007 | -2.9% |
| Matsuno et al. (2010) | Japanese $SO_2$ compensation law | 1974 | >3000 sites targeted by pollution load regulation | $SO_2$ emissions of targeted plants | 1973-2003 | -7.1% | 1973-1987 | -11.5% | 1987-2003 | -3.0% |

### 2.6.2 Mitigation costs

Unit costs for mitigation of non-$CO_2$ GHGs per unit of activity are in GAINS calculated as the sum of investment costs, labour costs, non-labour operation and maintenance costs, cost-savings due to recovery or saving of electricity, heat or gas, and non-energy cost savings. The unit cost of technology *m* in country *i* and year *t* is defined as:

$$C_{itm} = I_{im}\left[\frac{(1+r)^{T_m}\times r}{(1+r)^{T_m}-1}\right] + M_{im} + (L_{im}\times W_{it}\times w_{is}) - S_{im} - \left(E_{im}\times p_{it}^{electr}\right) - \left(H_{im}\times p_{it}^{heat}\right) - \left(G_{im}\times p_{it}^{gas}\right) \tag{5}$$

| | | |
|---|---|---|
| where | $I_{im}$ | is the upfront investment cost for technology *m* in country *i*, |
| | $\left[\frac{(1+r)^{T_m}\times r}{(1+r)^{T_m}-1}\right]$ | is the annualization factor for the investment cost with interest rate *r* and technology lifetime of $T_m$ years, |
| | $M_{im}$ | is the annual operation and maintenance cost for technology *m*, |
| | $L_{im}$ | is the fraction of annual work hours for operating technology *m*, |
| | $W_{it}$ | is the annual average wage in country *i* in year *t*, |
| | $w_{is}$ | is a country-specific wage adjustment factor for type of sector *s* (agriculture or manufacturing industry), |
| | $S_{im}$ | is the sum of non-energy annual cost-savings, |
| | $E_{im}$ | is the amount of energy recovered and utilized as electricity, |

$p_{it}^{electr}$ is the industry electricity price in country *i* in year *t*,

$H_{im}$ is the amount of energy recovered and utilized as heat,

$p_{it}^{heat}$ is the industry heat price in country *i* in year *t*,

$G_{im}$ is the amount of gas recovered, and

$p_{it}^{gas}$ is the industry gas price in country *i* in year *t*.

In the EUCLIMIT5 scenarios, the unit costs are expressed in constant 2015 Euros per unit of activity and a market interest rate of 10 percent is adopted in order to be consistent with the relevant (market) interest rate used in PRIMES (E3modelling, 2020) mitigation costs for $CO_2$.

Country and sector specific annual average wages are taken from LABORSTA (ILO, 2010) for historical years and growth in future wages is proportional to the expected future development in sector value added as provided by PRIMES (E3modelling, 2020) and in consistency with the European Commission's 2021 Ageing Report (EC, 2021a).

In the GAINS estimation of $CH_4$ mitigation costs, energy recovery from biogas production or reduced leakage of natural gas during production, transmission and distribution is valued at the electricity, heat or gas consumer price in industry as estimated and projected by PRIMES (E3modelling, 2020).

Gas recovery refers to recovered gas of an upgraded quality of 97 percent $CH_4$. For some mitigation options, e.g., when biogas is recovered from large-scale anaerobic digestion (note not applicable to small farm anaerobic digesters), upgrading from 60 to 97 percent $CH_4$ is necessary for supplying the gas to the grid (Persson, 2003). Costs for upgrading gas have in these cases been included in investment costs.

Similar to how the country and year specific emission level $E_{its}$ is estimated for each sector in equation (1), the total mitigation cost in sector *s* in country *i* and year *t* is defined for sets of application combinations of the possible technologies applicable in the sector. For a given country, year and sector, a technology setting is defined such that the sum of all application rates $Appl_{itsm}$ of possible technologies *m* (including the no control option) is always unity. The total cost of each technology set is defined as:

$$TC_{its} = \sum_m [A_{its} * C_{itm} * Appl_{itsm}] \;, \qquad (6)$$

where $A_{its}$ is the activity level, $C_{itm}$ is the cost per unit of activity and $\sum_m Appl_{itsm} = 1$ .

The average cost per unit of reduced emissions is first calculated for each technology available to a sector by dividing the unit cost with the difference between the technology emission factor and the no control emission factor, such that:

$$AC_{itm} = \frac{C_{itm}}{ef_{it}^{No\_control} - ef_{itm}}. \qquad (7)$$

Within a sector, the technologies available are first sorted by their respective average cost. The technology with the lowest average cost is ranked the first-best technology and assumed adopted to its full extent in a given sector. The second-best technology is the technology with the second lowest average cost and is assumed available for adoption provided it can achieve an emission factor that is lower than the first-best technology. The marginal cost of the second-best technology when implemented in the cost curve is defined as:

$$MC_{it2} = \frac{C_{it2} - C_{it1}}{ef_{it1} - ef_{it2}} \tag{8}$$

In a similar manner, each additional technology available in a sector is added on top of the next best available technology. Note that a technology with both a higher average cost and a higher emission factor than another technology available to a sector will not be adopted at all as it is less effective in reducing emissions and come at a higher cost than other available technologies. In this way, a marginal mitigation cost curve with strictly convex properties is built up sector-wise by country and year.

# 3 $CH_4$ emission sources

## 3.1 Overview of $CH_4$ emission sources and control implemented in the baseline scenario

$CH_4$ emission sources in the GAINS model include anthropogenic $CH_4$ emissions from livestock, rice cultivation, biodegradable solid waste, wastewater, coal mining, oil and gas production, gas transmission pipelines, gas consumer distribution networks, and combustion emissions from fuel used for energy consumption and from open burning of agricultural waste residuals (Höglund-Isaksson et al., 2020). Other types of open burning of biomass for non-energy purposes, e.g., human-induced forest fires, have been excluded from the analysis due to lack of systematic information. Table 3 provides an overview of the structure of $CH_4$ sources in GAINS and their respective control options.

In addition to the structure presented in Table 3, it should be mentioned that the livestock categories dairy cows, non-dairy cattle, pigs, poultry and sheep have been further split into the following farm-size classes; less than 15 livestock units (LSU), 15 to 50 LSU, 50 to 100 LSU, 100 to 500 LSU, and above 500 LSU. This allows for considering the impact of scale on mitigation potentials and costs.

Table 4 presents a list of current legislation affecting $CH_4$ emissions in the EU member states. The EU common agricultural policy (CAP) affect emissions through changes in livestock numbers and fertilizer use, which are activity drivers estimated by the CAPRI model and imported as externally produced data into GAINS. Agricultural policies modelled specifically in GAINS are impacts on farm AD control from feed-in tariffs and other subsidies introduced in selected member states to stimulate co-digestion of manure and the EU-wide ban on burning of crop residues on fields. Other policies adopted EU-wide include the different Waste sector Directives. The 2018 amendment of the EU Landfill Directive specifies that a maximum 10% of total municipal solid waste (MSW) can be landfilled in 2035, with a five years grace period granted to Bulgaria, Croatia, Cyprus, Greece, Hungary, Latvia, Romania, and Slovakia, which in 2035 must meet a target of maximum 25% MSW landfilled in 2035. The EU Waste management framework Directive defines a waste treatment hierarchy in which recycling of waste is preferred to energy recovery, which in turn is preferred to landfill of untreated waste. In addition to the EU-wide Directives in the waste sector, six member states have introduced complete bans on landfill of biodegradable waste (Austria, Belgium, Denmark, Germany, Netherlands, Sweden).

Table 3: Overview of GAINS structure of $CH_4$ emission sources and mitigation options.

| Emission source | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| Coal mining | MINE_BC_PRE | Brown coal mining: pre-mining emissions | BC_DEGAS | Pre-mining degasification |
| | MINE_BC_VAM | Brown coal mining: emissions during mining | none | no option identified |
| | MINE_BC_POST | Brown coal mining: post-mining emissions | none | no option identified |
| | MINE_HC_PRE | Hard coal mining: pre-mining emissions | HC_DEGAS | Pre-mining degasification |
| | MINE_HC_VAM | Hard coal mining: emissions during mining | VAMOX | Oxidation of ventilation air methane |
| | | | VAMOX_VENT | VAMOX combined with improved ventilation system |
| | MINE_HC_POST | Hard coal mining: post-mining emissions | none | no option identified |
| | COAL_ABAND | Abandoned coal mines | FLOODING | Good practice -flooding of mines |
| Oil production | PROD_AGAS -CRU | Oil production: venting associated gas | REC_USE | Extended recovery and utilization of associated gas |
| | | | REC_USE2 | Monitoring of flaring and venting of associated gas not recovered |
| | PROD_LEAK -CRU | Oil production: unintended leakage | GP | Leakage control using LDAR technology |
| Oil refinery | PR_REF | Oil transportation, storage and refining | FLA_REF | Leakage control using LDAR technology |
| Gas production | PROD_AGAS -GAS | Gas production: venting associated gas | REC_USE | Extended recovery and utilization of associated gas |
| | PROD_LEAK -GAS | Gas production: unintended leakage | GP | Leakage control using LDAR technology |
| Long-distance gas transportation | TRANS -GAS | Gas transmission pipelines | CONT_TRANS | Set of measures incl. new controller & dry seals |
| | | | COMPRESS | Pipeline upgrade to minimum leakage rate |
| Consumer gas distribution networks | DOM -GAS | Domstic sector: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | IN_BO -GAS | Industrial boilers: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | IN_OC -GAS | Other industry combustion: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | PP_EX_OTH -GAS | Power plants existing: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | PP_NEW -GAS | Power plants new: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | TRA_RD_HDB/HDT / LD4C/LD4T -GAS | Road transport/bus/trucks/cars/vans: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | CON_COMB -GAS | Other combustion sources: gas fuel | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |
| | NONEN -GAS | Non-energy use: gas | CONT_NET | Leak detection and repair (LDAR) programs |
| | | | REPL_NET | Replacement of grey cast iron networks |

Continued Table 3: Overview of GAINS structure of $CH_4$ emission sources and mitigation options.

| Emission source | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| Livestock | AGR_BEEF -OL_F | Non-dairy cattle with liquid manure management: enteric fermentation emissions | FEED | Feed additives and/or changed feed management practices |
| | | | BREED | Breeding through selection on farms with > 50 LSU: enhance productivity, fertility and longevity to minimize kg CH4/kg milk |
| | AGR_BEEF -OL | Non-dairy cattle with liquid manure management: manure management emissions | FARM_AD | Farm-scale anaerobic digestion on farms with minimum size 100 LSU |
| | AGR_BEEF -OS | Non-dairy cattle with solid manure management: enteric fermentation and manure management emissions | FEED | Feed additives and/or changed feed management practices |
| | | | BREED | Breeding through selection on farms with > 50 LSU: enhance productivity, fertility and longevity to minimize kg CH4/kg milk |
| | AGR_COWS -DL_F | Dairy cows with liquid manure management: enteric fermentation emissions | FEED | Feed additives and/or changed feed management practices |
| | | | BREED | Breeding through selection on farms with > 50 LSU: enhance productivity, fertility and longevity to minimize kg CH4/kg milk |
| | AGR_COWS -DL | Dairy cows with liquid manure management: manure management emissions | FARM_AD | Farm-scale anaerobic digestion on farms with minimum size 100 LSU |
| | AGR_COWS -DS | Dairy cows with solid manure management: enteric fermentation and manure management emissions | FEED | Feed additives and/or changed feed management practices |
| | | | BREED | Breeding through selection on farms with > 50 LSU: enhance productivity, fertility and longevity to minimize kg CH4/kg milk |
| | AGR_PIG -PL | Pigs with liquid manure management | FARM_AD | Farm-scale anaerobic digestion on farms with minimum size 100 LSU |
| | AGR_PIG -PS | Pigs with solid manure management | none | no option identified |
| | AGR_POULT -LH | Laying hens | none | no option identified |
| | AGR_POULT -OP | Other poultry | none | no option identified |
| | AGR_OTANI -BS | Buffaloes | none | no option identified |
| | AGR_OTANI -HO | Horses | none | no option identified |
| | AGR_OTANI -SH | Sheep and goats | BREED | Breeding through selection on farms with > 50 LSU: enhance productivity, fertility and longevity to minimize kg CH4/kg milk |
| Rice cultivation | RICE_FLOOD | Rice cultivation | COMB_RICE1,<br>COMB_RICE2 | Water management (intermittent aeration), alternative hybrids and soil amendments |
| Agricultural waste burning | WASTE_AGR | Open burning of agricultural waste | BAN | Enforcement of ban to openly burn agricultural waste |

Continued Table 3: Overview of GAINS structure of $CH_4$ emission sources and mitigation options

| Emission source | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| Solid waste | MSW_FOOD | Municipal solid waste: food and garden | TREAT_AD | Source separation & anaerobic digestion with biogas recovery |
| | | | TREAT_HSC | Household composting |
| | | | TREAT_LSC | Source separation & large-scale composting |
| | | | TREAT_INC | Incineration of mixed waste with energy recovery |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| | MSW_PAP/ MSW_TEX/ MSW_WOOD | Municipal solid waste: paper/ textile | TREAT_REC | Source separation & recycling |
| | | | TREAT_INC | Incineration of mixed waste with energy recovery |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| | INW_FOOD | Industrial solid waste: food, beverages | FOOD_AD | Anaerobic digestion with biogas recovery |
| | | | FOOD_COM | Composting |
| | | | FOOD_INC | Incineration |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| | INW_PAP | Industrial solid waste: pulp and paper | PAP_INC | Recovery of black liqour for energy utilization |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| | INW_TEX | Industrial solid waste: Textile, footwear, leather | TEX_INC | Incineration with energy recovery |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| | INW_WOOD | Industrial solid waste: wood and wood products | WOOD_REC | Recycling for chip board production |
| | | | WOOD_INC | Incineration with energy recovery |
| | | | SWD_FLA | Landfill with recovery and flaring of landfill gas |
| | | | SWD_USE | Landfill with recovery and utilization of landfill gas |
| Wastewater | WW_DOM_CC_TRM | Domestic wastewater: centralized collection | DOM_CC_AER | Aerobic treatment |
| | | | DOM_CC_23 | Anaerobic treatment |
| | | | DOM_CC_23U | Anaerobic treatment (secondary/tertiary) with biogas recovery |
| | WW_DOM_DC_NOC | Domestic wastewater: decentralized collection, no treatment | none | no option identified |
| | WW_DOM_DC_TRM | Domestic wastewater: decentralized collection, with treatment | none | no option identified |
| | IND_FOOD_TRM | Industrial wastewater: food, beverages | IND_FOOD_AERO | Aerobic treatment |
| | | | IND_FOOD_ANAE_NON | Anaerobic treatment without biogas recovery |
| | | | IND_FOOD_ANAE_USE | 2-stage: Anaerobic with biogas recovery followed by aerobic treatment |
| | IND_PAP_TRM | Industrial wastewater: pulp and paper | IND_PAP_AERO | Aerobic treatment |
| | | | IND_PAP_ANAE_NON | Anaerobic treatment without biogas recovery |
| | | | IND_PAP_ANAE_USE | 2-stage: Anaerobic with biogas recovery followed by aerobic treatment |
| | IND_OTH_TRM | Industrial wastewater: other organic | IND_OTH_AERO | Aerobic treatment |
| | | | IND_OTH_ANAE_NON | Anaerobic treatment without biogas recovery |
| | | | IND_OTH_ANAE_USE | 2-stage: Anaerobic with biogas recovery followed by aerobic treatment |

Continued Table 3: Overview of GAINS structure of $CH_4$ emission sources and mitigation options

| Emission source | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| Combustion in stationary sources | DOM -various fuels | Domestic sector: various fuels | none | no option identified |
| | IN_BO -various fuels | Industrial boilers: various fuels | none | no option identified |
| | IN_OC -various fuels | Other industry combustion: various fuels | none | no option identified |
| | PP_EX_OTH -various fuels | Power plants existing: various fuels | none | no option identified |
| | PP_NEW -various fuels | Power plants new: various fuels | none | no option identified |
| | CON_COMB -various fuels | Other combustion sources: various fuels | none | no option identified |
| Combustion in mobile sources | TRA_RD_HDB/HDT/ LD4C/LD4T - various fuels | Road transport/bus/trucks/cars/vans: various fuels | none | no option identified |
| | TRA_OT_AGR/CNS/ INW/LB/RAI - various fuels | Non-road transport: various modes and fuels | none | no option identified |

Table 4: Current legislation affecting $CH_4$ emissions in EU member states.

| **Sector** | **Policy** | **Regional coverage** | **Policy description and implementation in GAINS** |
|---|---|---|---|
| Agriculture | Feed-in tariffs or other subsidies to stimulate co-digestion of manure on farms | Italy, Netherlands, Latvia, Sweden, Cyprus, Austria, Croatia, Germany | Reflected via assumptions on uptake of farm-scale biogas technology consistent with information from EurObserv'ER (2020) on installed capacity. Future uptake follows trend in biogas production from anaerobic digestion as projected in the PRIMES model Reference scenario. |
| | EU Common Agricultural Policy (CAP) and EU Nitrate Directive (EEC/676/1991) with revisions | EU-wide | Reflected in GAINS through input of CAPRI model data on trends in livestock numbers, milk yield and fertilizer use. |
| | Ban on burning of crop residues | EU-wide | Assumed not fully enforced. GAINS uses information derived from satellite images (e.g., MODIS) as approximate estimates of the mass of crop burned on fields. |
| Waste & wastewater | EU Landfill Directive (EC/31/1999) with amendment (EC/850/2018) and EU Waste and Packaging Directives (EC/851/2018, EC/852/2018) | EU-wide | Biodegradable waste diverted away from landfills (relative 1990 by -25% in 2006, -50% in 2009 and -65% in 2016). All landfill sites equipped with gas recovery by 2009. By 2035, countries must not landfill more than 10% of MSW generated. Member states that landfill more than 60% of MSW in 2013 are given a 5 years grace period but must not landfill more than 25% in 2035. GAINS Reference scenario assumes future targets will be met. |
| | EU Waste Management Framework Directive (EC/98/2008) | EU-wide | The following hierarchy is to be respected in waste treatment: recycling and composting preferred to incineration/energy recovery, which in turn is preferred to landfill disposal. Considered in GAINS when simulating pathway for compliance with the Landfill Directive target. |
| | Decree on waste landfill | Slovenia | Decree on landfill of waste beyond EU Landfill Directive. Includes partial ban on landfill of biodegradable waste. |
| | Legislation to replace current composting with anaerobic digestion of food waste | Germany | In GAINS, the current composting of organic waste is phased-out linearly and replaced with anaerobic digestion between 2020 and 2050. |
| | Ban on landfill of biodegradable waste. | Austria, Belgium, Denmark, Germany, Netherlands, Sweden | Complete ban on landfill of untreated biodegradable waste. Reflected in GAINS. |
| | EU urban wastewater treatment directive (EEC/271/1991) | EU-wide | GAINS reflects an "appropriate treatment" of wastewater from urban households (all agglomerations > 2000 people) and food industry must be in place latest by end of 2005. This means discharge must ensure receiving waters meet relevant quality objectives. |

## 3.2 Energy sector

### 3.2.1 Coal mining

Formation of coal produces $CH_4$, which is released to the atmosphere during mining. IPCC (2006, Vol.2, p.4.9) identifies three sources of $CH_4$ emissions from coal mining: liberation of $CH_4$ during breakage of coal in the coal mine, post-mining emissions during handling, processing and transportation of mined coal, and emissions from abandoned coal mines. Following the split of the activity data into brown and hard coal, emissions are calculated separately for the two coal types. All brown coal is assumed to be surface mined, while hard coal is assumed being partly surface mined and partly mined underground. Emissions from peat production are understood to be negligible and have been set to zero.

Emissions from brown and hard coal production are calculated as follows:

$$E_{BC;it} = A_{BC;it} \times \left[ef_{m;i}^{surface} + ef_{postm;i}^{surface}\right] \tag{9}$$

$$\begin{aligned} &E_{HC;it} = A_{HC;it} \times \left[\gamma_{HC;i}^{undergr} \times \left(ef_{m;i}^{undergr} + ef_{postm;i}^{undergr}\right)\right] + \\ &A_{HC;it} \times \left[\left(1 - \gamma_{HC;i}^{undergr}\right) \times \left(ef_{m;i}^{surface} + ef_{postm;i}^{surface}\right)\right] \end{aligned} \tag{10}$$

where

$$ef_{m;i}^{surface} = ef_{m;i;NOC}^{surface} \times \left(1 - r_{dgas}\right) \times CLEappl_{dgas;i}, \tag{11}$$

$$\begin{aligned} &ef_{m;i}^{undergr} = ef_{m;i;NOC}^{undergr} \times \left[\alpha_{VAM;i} \times \left(1 - r_{VAM}\right) \times CLEappl_{VAM;i}\right] \\ &+ef_{m;i;NOC}^{undergr} \times \left[\left(1 - \alpha_{VAM;i}\right) \times \left(1 - r_{dgas}\right) \times CLEappl_{dgas;i}\right] \end{aligned} \tag{12}$$

and where

| | |
|---|---|
| $A_{BC;it}$ and $A_{HC;it}$ | are amounts of brown and hard coal produced in country *i* in year *t*, |
| $ef_{m;i;NOC}^{surface}$ | is a country-specific no control emission factor for surface mining emissions, |
| $ef_{postm;i}^{surface}$ | is the average world IPCC default emission factor for post-mining emissions from surface mines, |
| $ef_{m;i;NOC}^{undergr}$ | is a country-specific no control emission factor for underground mining emissions, |
| $ef_{postm;i}^{undergr}$ | is the average world IPCC default emission factor for post-mining emissions from underground mines, |
| $\gamma_{HC;i}^{undergr}$ | is a country-specific fraction of hard coal being mined underground as opposed to surface mining, |
| $\alpha_{VAM;i}$ | is a country-specific fraction of underground mining emissions being released through the ventilation air as opposed to pre-mine degasification emissions, |

$r_{dgas}$ and $r_{VAM}$ are the removal efficiencies of technologies removing degasification and ventilation air methane, respectively,

$CLEappl_{VAM;i}$ is the application of technology removing ventilation air methane, and

$CLEappl_{dgas;i}$ is the application of technology removing degasification methane.

Country-specific information about emissions released from surface and underground mines in year 2015 were taken from UNFCCC-CRF (2019) together with information about the fraction of hard coal mined in surface mines. Information on the current application of pre-mining degasification was extracted from the same source. The fraction of underground mine gas exiting as ventilation air methane as opposed to being released during pre-mining degasification is taken from USEPA (2003). No country-specific information about post-mining emissions was found and, hence, IPCC (2006) default emission factors for surface and underground mining, respectively, were applied to all countries for post-mining emissions. Finally, the implied emission factor for all coal mining sources was scaled to match GAINS emissions with UNFCCC-CRF reported emissions for year 2015. The resulting derived emission factors for coal-mining in EU-27 countries are presented in Table 5. For clarity, the derived emission factors presented in Table 5 are reproduced in Table 6 in terms of ton $CO_2$eq per TJ coal produced using country-specific calorific values of coal from IEA-WEO (2009).

Emissions from both surface and underground mines can be reduced if $CH_4$ is recovered through pre-mining drainage up to ten years before the mining starts (USEPA, 2008a). Currently in the US, at least 90 percent of degasification emissions from underground coal mines are recovered and utilized (USEPA, 2010a). In GAINS, this is assumed technically possible in other countries as well. There is, however, only one project known to be recovering and utilizing $CH_4$ from pre-mine drainage at a surface mine and details about the removal efficiency of this option are uncertain (Sino-US New Energy Sci-Tech Forum, 2009). In GAINS, it is considered technically possible to recover 90 percent of the drainage gas also from surface mines. Costs for degasification are taken from Thakur (2006) and reflect costs for in-mine drilling, underground pipeline costs, and hydraulic fractioning of vertical wells and other gob wells.

Ventilation air methane (VAM) from underground coalmines can be recovered and oxidized through installation of VAM oxidizers (Mattus and Källstrand, 2010). Although the application on coalmines is still in an early phase, the technology is well known from control of odor and VOC emissions worldwide. The technology oxidizes at least 95 percent of VAM when applied to a ventilation shaft. It uses the energy released during the oxidation to keep the process running, which keeps fuel costs limited to the initial start-up phase. For a thermal oxidation process to run without interruptions the $CH_4$ concentration in the ventilation air needs to be at least 0.3 percent. For some recent installations in China a catalytic oxidation process is in use, which operate with $CH_4$ concentration rates in the ventilation air as low as 0.2% (Somers and Burklin, 2012). Securing this concentration level without increasing explosion risks (i.e. $CH_4$ concentrations in the air should never be in the explosive range between 5 and 15 percent), may in some mines require investments in more efficient ventilation systems. A general assumption is made in GAINSv4 that it is technically possible to keep $CH_4$ concentration levels at a steady rate of at least 0.3 percent, and therefore to install self-sustained VAM oxidizers (Mattus and Källstrand, 2010), on 50 percent of the ventilation air emitted from underground coal mines in all countries. Combining a catalytic oxidation VAM technology with an improved ventilation system is assumed to extend the feasible application of VAM oxidizers to 70 percent of VAM emitted from underground mines in all countries. An improved ventilation system is taken to double the ventilation capacity of the mine compared with a conventional system, thereby doubling the amount of electricity used for ventilation. Costs for VAM oxidation technology and installation are taken from USEPA (2003, p.30) and GMI (2008) and refer to installations in the US and China. Costs for increased electricity use for ventilation in mines are based on information from Unruh (2002) and Papar et al. (1999). No mitigation potential is assumed for post-mining emissions.

For further details on mitigation costs for control of coalmine emissions, see the Supplements of Höglund-Isaksson (2012) and Höglund-Isaksson et al. (2020).

Table 5: Coal mining emission factors for EU-28 countries in the GAINS model.

| | **Coal production 2015 (Mt coal) Source: PRIMES (2020)** | | **Derived no control CH$_4$ emission factors (kt CH4 /Mt coal produced)** | | | | | | | | **Implied emission factors 2015 (kt CH4/Mt coal)** | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Brown coal mining | | | | Hard coal mining | | | | All coal types and sources | |
| Country | Brown coal | Hard coal | total | pre-mining | mining | post-mining | total | pre-mining | mining | post-mining | No control | Including current control |
| Bulgaria | 24.5 | 0 | 1.326 | 0.490 | 0.734 | 0.102 | 24.550 | 9.656 | 14.291 | 0.603 | 1.354 | 1.04 |
| Czech Rep. | 34.7 | 12.9 | 1.717 | 0.393 | 1.243 | 0.082 | 8.545 | 3.295 | 4.876 | 0.374 | 3.567 | 2.82 |
| France | 0 | 0.2 | n.a. | n.a. | n.a. | n.a. | 9.157 | 4.054 | 4.500 | 0.603 | 9.157 | 6.60 |
| Germany | 174.8 | 9.0 | 0.013 | 0.004 | 0.006 | 0.004 | 22.335 | 8.792 | 13.013 | 0.530 | 1.106 | 0.83 |
| Greece | 49.5 | 0 | 1.203 | 0.445 | 0.666 | 0.093 | n.a. | n.a. | n.a. | n.a. | 1.203 | 0.92 |
| Hungary | 5.4 | 0 | 0.027 | 0.008 | 0.012 | 0.008 | n.a. | n.a. | n.a. | n.a. | 0.027 | 0.02 |
| Italy | 0 | 0.1 | n.a. | n.a. | n.a. | n.a. | 18.397 | 7.177 | 10.622 | 0.598 | 18.397 | 13.9 |
| Poland | 58.8 | 83.1 | 1.183 | 0.437 | 0.655 | 0.091 | 9.935 | 3.790 | 5.610 | 0.535 | 6.306 | 4.79 |
| Romania | 27.7 | 0 | 1.057 | 0.391 | 0.585 | 0.081 | n.a. | n.a. | n.a. | n.a. | 1.057 | 0.83 |
| Slovakia | 2.0 | 0 | 8.001 | 0.392 | 7.528 | 0.082 | n.a. | n.a. | n.a. | n.a. | 8.001 | 7.75 |
| Slovenia | 3.3 | 0 | 3.323 | 0.399 | 2.840 | 0.083 | n.a. | n.a. | n.a. | n.a. | 3.323 | 3.07 |
| Spain | 0 | 5.5 | n.a. | n.a. | n.a. | n.a. | 4.510 | 1.785 | 1.928 | 0.797 | 4.510 | 2.06 |
| United Kingdom | 0 | 10.1 | n.a. | n.a. | n.a. | n.a. | 10.110 | 3.801 | 5.626 | 0.682 | 10.110 | 4.21 |

Sources: Calibrated to implied emission factors for UNFCCC-CRF 2019 taking account of current pre-mining control through degasification; USEPA 2003 for assumptions on the fraction of underground mine gas exiting as ventilation air methane as opposed to released during pre-mining degasification; IPCC (2006) for default post-mining emission factors specified for surface and underground mining, respectively. Note that default emission factors have been scaled to better match national implied emission factors reported to the UNFCCC.

Table 6: Coal mining emission factors for EU-28 countries expressed in $tCO_2eq/TJ$ coal produced. Energy conversion factor: 41.868 GJ/toe.

| | **Calorific values used in GAINS (source is IEA-WEO)** | | **Derived no control $CH_4$ emission factors in t CO2eq/TJ coal mined (with CH4 GWP of 28 times that of CO2 over 100 years)** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | toe/t coal | | Brown coal mining | | | | Hard coal mining | | | |
| Country | Brown coal (lignite) | Hard coal (bitumen coal) | total | pre-mining | mining | post-mining | total | pre-mining | mining | post-mining |
| Bulgaria | 0.165 | 0.600 | 3.858 | 1.426 | 2.135 | 0.297 | 19.643 | 7.726 | 11.434 | 0.482 |
| Czech Rep. | 0.301 | 0.435 | 2.739 | 0.627 | 1.982 | 0.130 | 9.430 | 3.636 | 5.381 | 0.413 |
| France | n.a. | 0.631 | n.a. | n.a. | n.a. | n.a. | 6.967 | 3.084 | 3.424 | 0.459 |
| Germany | 0.212 | 0.583 | 0.030 | 0.009 | 0.013 | 0.009 | 18.392 | 7.240 | 10.715 | 0.437 |
| Greece | 0.130 | n.a. | 4.442 | 1.642 | 2.458 | 0.342 | n.a. | n.a. | n.a. | n.a. |
| Hungary | 0.335 | n.a. | 0.039 | 0.011 | 0.017 | 0.011 | n.a. | n.a. | n.a. | n.a. |
| Italy | n.a. | 0.641 | n.a. | n.a. | n.a. | n.a. | 13.778 | 5.375 | 7.955 | 0.448 |
| Poland | 0.203 | 0.525 | 2.798 | 1.034 | 1.548 | 0.215 | 9.085 | 3.466 | 5.130 | 0.489 |
| Romania | 0.181 | n.a. | 2.804 | 1.037 | 1.552 | 0.216 | n.a. | n.a. | n.a. | n.a. |
| Slovakia | 0.301 | n.a. | 12.761 | 0.625 | 12.006 | 0.130 | n.a. | n.a. | n.a. | n.a. |
| Slovenia | 0.312 | n.a. | 5.113 | 0.615 | 4.371 | 0.128 | n.a. | n.a. | n.a. | n.a. |
| Spain | n.a. | 0.539 | n.a. | n.a. | n.a. | n.a. | 4.017 | 1.590 | 1.718 | 0.709 |
| United Kingdom | n.a. | 0.604 | n.a. | n.a. | n.a. | n.a. | 8.036 | 3.021 | 4.472 | 0.542 |

### *3.2.2 Abandoned coal mines*

Countries reporting $CH_4$ emissions to the UNFCCC in the Annex-1 category are expected to enter emissions from abandoned coalmines in the Common Reporting Formats (CRFs). The reported emissions make up the activity data for this source sector in GAINS.

The release of $CH_4$ emissions from abandoned coal mines typically depends on the status of the abandoned mine, i.e., whether it is left open for venting in order to prevent build-up of explosive $CH_4$ pockets underground, flooded to prevent $CH_4$ emissions from escaping, or sealed through cement plugging (USEPA, 2004). For the modelling in GAINS, it is assumed that without regulation the no control case is venting. The control option considered is flooding, which is assumed to prevent 90% of emissions compared to the venting case. Sealing is not considered a $CH_4$ control option in GAINS, because to effectively prevent gas leakage, at least 95% of shafts must be sealed (USEPA, 2004), which likely makes it relatively expensive. In contrast, the cost of flooding abandoned coal mines is likely low or even profitable, as abandoned mines can potentially fill an important role in a future transformation to renewable energy. Abandoned coal mines can be used as pumped storage hydroelectric plants (Pujades et al., 2016; Jessop et al., 1995) or flooded and converted to giant floating solar farms as in Huainan, China (China Daily, 2017). The net cost to society of flooding abandoned coal mines is therefore taken to be zero in GAINS.

### 3.2.3 Oil and gas production and processing

Extraction of crude oil and natural gas gives rise to $CH_4$ emissions, partly as a result of intended flaring or venting of associated gas for security reasons and partly due to unintended leakage of fugitive emissions, which occur along the whole production process from well head to upgrading and storage (IPCC, 2006, Vol.2, Section 4.2). Associated gas is a gas compound mainly consisting of $CH_4$, which is released as oil or natural gas is pumped to the surface. For security reasons, the associated gas needs to be released and is therefore flared off or simply vented. Alternatively, the associated gas can be recovered and utilized for energy purposes provided there is an infrastructure present to transport the recovered gas to consumers.

The methodology for estimating methane emissions from oil and gas production has been described in Höglund-Isaksson (2017). Emissions from venting and flaring of associated gas are calculated separately for fugitive emissions and unintended leakage. Total emissions from oil and gas production are the sum of venting, flaring and leakage emissions:

$$E_{its} = E_{its}^{venting} + E_{its}^{flaring} + E_{its}^{leakage} \tag{13}$$

Venting emissions from production of oil and gas, respectively, are calculated as:

$$E_{it;oil}^{venting} = A_{it}^{oil} * 20 * (1 - r_i) * \left(c_i a_i^{conv} v_i^{conv} + (1 - c_i) a_i^{heavy} v_i^{heavy}\right) \tag{14}$$

$$E_{it;gas}^{venting} = A_{it}^{gas} * 20 * a_i^{gas}(1 - r_i) v_i^{gas} \tag{15}$$

| | | |
|---|---|---|
| where | $\mathrm{A}_{\mathrm{it}}^{\mathrm{oil;gas}}$ | is energy content of crude oil or dry gas produced in country *i* in year *t*, |
| | 20 | 20 kt $CH_4$/PJ conversion of energy content to amount of $CH_4$, |
| | $c_i$ | fraction of conventional oil (as opposed to heavy oil) produced, |
| | $a_i^{conv}$,$a_i^{heavy}$,$a_i^{gas}$ | are associated gas as fractions of the total energy content of conventional, heavy oil, or gas produced in year *t*, |
| | $r_i$ | is the fraction of associated gas recovered for utilization or reinjection in year *t*, and |
| | $v_i^{conv}$,$v_i^{heavy}$,$v_i^{gas}$ | are fractions of unrecovered associated gas that is vented (as opposed to flared). |

Combustion emissions from flaring are derived assuming on average two percent incomplete combustion of $CH_4$ from open flares (Johnson and Kostiuk, 2002). The volume of gas flared is the fraction of unrecovered associated gas that is not vented, i.e. methane emissions from flaring are estimated as:

$$E_{it;oil}^{flaring} = 0.02 * A_{it}^{oil} * 20 * (1 - r_i) * \left(c_i a_i^{conv}(1 - v_i^{conv}) + (1 - c_i) a_i^{heavy}\left(1 - v_i^{heavy)y}\right)\right) \tag{16}$$

$$E_{it;gas}^{flaring} = 0.02 * A_{it}^{gas} * 20 * a_i^{gas}(1 - r_i)\left(1 - v_i^{gas}\right) \tag{17}$$

Country-specific information for the year 2005 on amounts of associated gas and the fraction of associated gas reinjected or recovered are taken from EIA (2015) International Energy Statistics and used as starting point for emission estimates. From EIA (2015) Country Analysis Briefs, types of hydrocarbon produced are identified, i.e. conventional crude oil, heavy crude oil (API gravity <22.3°), oil sands and natural gas, as well as the fraction of offshore production. For gas production only two measurement results have been found for the fraction of associated gas to the energy content of gas produced. These are 0.03 percent for Canada (Johnson and Coderre, 2011), which is adopted as representative also for the EU countries.

Measurement data on unrecovered associated gas vented as opposed to flared from oil and gas wells are extremely rare and no such measurements from European oil and/or gas wells have been found in published sources. We note however that the technology to use satellite observations to continuously monitor methane emissions is under rapid development with the TROPOMI satellite being able to detect major emission events, primarily from oil and gas operations (SRON, 2021). There are several initiatives to improve the methane detection precision of satellites, anticipating the ability to detect methane emissions at the facility level in a few years time from now (e.g., MethaneSAT at www.methanesat.org, Kayrros at www.kayrros.com, and Blue Sky Satellite Communications www.bluesky-sat.com). For now, we resort to a bottom-up approach described in Höglund-Isaksson (2017). It makes use of direct measurement results published by Johnson and Coderre (2011) for over 6000 oil and gas wells active in the province of Alberta in 2008 and specified for different types of hydrocarbons produced (i.e., conventional or heavy oil and conventional natural gas). The results of the Canadian measurements are summarized in Table 7 and the last row shows the fraction of unrecovered associated gas that is vented instead of flared. These factors are assumed representative also for EU circumstances. Note that the assumptions on the total associated gas generated, recovered and reinjected are country-specific and taken from EIA (2015). By calibrating the country-specific recovery rate, the estimated volumes of associated gas vented and flared were matched to the total volumes of associated gas flared as estimated from satellite images of gas flares (NOAA, 2011). The resulting derived emission factors for venting of associated gas are presented in Table 8.

Table 7: Recovery, venting and flaring from oil and gas wells in the province of Alberta 2002-2008. Adapted from Johnson and Coderre (2011).

| | | Conventional oil | Heavy oil | Natural gas |
|---|---|---|---|---|
| Associated gas | % of production | 35.5% | 5.1% | 0.03% |
| | | | | |
| Recovered/ reinjected | % of associated gas | 97.1% | 85.8% | 0% |
| Flared | | 2.1% | 1.7% | 60% |
| Vented | | 0.8% | 12.4% | 40% |
| Sum | | 100% | 100% | 100% |
| | | | | |
| Vented | % of flared/vented | 29.1% | 87.7% | 40% |

Maximum technically feasible reduction of $CH_4$ emissions from associated gas management related to oil and gas production is defined such that for all countries it is assumed possible to recover and utilize 99 percent of the associated gas generated. This high level of associated gas recovery is already achieved in Norway (Husdal et al., 2016a,b; EIA, 2015) and it is therefore assumed possible to achieve the same recovery rate in other countries as well. As shown in Table 8, many EU countries already recover 95 percent or more of the associated petroleum gas generated. Hence, the additional mitigation potential through extended recovery rates for associated gas are limited within the EU. Costs are taken from OME (2001) and described in detail in the Supplement of Höglund-Isaksson (2012).

In addition to extending associated gas recovery rates to 99 percent, it is assumed technically possible to reduce venting through maximized flaring of the 1 percent of associated gas that is not recovered. In a Leak Detection and Repair (LDAR) program (USEPA, 2016; McCabe and Fleischmann, 2014), infrared cameras can be used to frequently scan equipment for methane leakages, which are promptly repaired. If installed to scan flares of associated gas, LDAR would allow for identification and remedy of 'super-emitting' events, reduced routine venting as well as reduced number and duration of temporary flare shut-downs caused by unfavorable weather and wind conditions (Husdal et al., 2016b, p.31). Details on costs and removal efficiency of LDAR technology are described below in conjunction with control of unintended fugitive leakage from oil and gas systems. To our knowledge, LDAR programs have until now been introduced in Europe to control unintended fugitive leakages from gas processing plants and transmission and distribution networks (Saunier et al., 2017), however, not to control venting of associated gas. The applicability and cost of the technology for this purpose is therefore highly uncertain. As a conservative assumption we assume that it is possible to reduce venting of unrecovered associated gas by 30 percent if LDAR is implemented across all oil and gas production facilities. The marginal cost is very high (exceeding 500 €/t $CO_2$eq) as LDAR is assumed applied on top of a 99 percent recovery rate of associated gas and therefore only addressing emissions from the one percent associated gas that is not recovered.

Table 8: Assumptions for deriving emission factors for CH4 venting, flaring and unintended leakage from oil and gas production in EU production countries.

| | Types of crude oil produced as fraction of total production (EIA, 2010) | | Fraction offshore production (EIA, 2010) | **Oil production: venting and unintended leakage 2015** | | | **Gas production: venting and unintended leakage** | | **Inefficient combustion of gas flares** |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Recovery/ reinjection of associated petroleum gas (calibrated to satellite images of gas flares) | Implied emission factor: venting of associated gas | Implied emission factor: unintended leakage (IPCC, 2006, Vol.2, Table 4.2.4) | Implied emission factor: venting of associated gas | Implied emission factor: unintended leakage (IPCC, 2006, Vol.2, Table 4.2.4) | Implied emission factor: flaring of associated gas (98% combustion efficiency, 86 vol% $CH_4$) |
| Country | Conventional oil | Heavy oil | | % | kt $CH_4$/PJ oil produced | | kt $CH_4$/PJ gas produced | | kt $CH_4$/PJ gas flared |
| Austria | 100% | 0% | 0% | 96.0% | 0.052 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Bulgaria | 100% | 0% | 0% | 90.1% | 0.129 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Croatia | 100% | 0% | 0% | 97.0% | 0.039 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Cyprus | n.a. | n.a. | n.a. | n.a. | n.a. | n.a. | 0.00194 | 0.06 | 0.2892 |
| Czech Republic | 100% | 0% | 0% | 89.1% | 0.142 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Denmark | 100% | 0% | 100% | 98.0% | 0.026 | 0.000015 | 0.00194 | 0.000001 | 0.2892 |
| France | 100% | 0% | 80% | 94.1% | 0.078 | 0.012 | 0.00194 | 0.00396 | 0.2892 |
| Germany | 100% | 0% | 0% | 99.0% | 0.013 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Greece | 100% | 0% | 0% | 99.0% | 0.013 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Hungary | 100% | 0% | 0% | 97.0% | 0.039 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Ireland | 100% | 0% | 0% | n.a. | n.a. | n.a. | 0.00194 | 0.06 | 0.2892 |
| Italy | 100% | 0% | 0% | 98.0% | 0.026 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Lithuania | 100% | 0% | 0% | 94.1% | 0.078 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Netherlands | 100% | 0% | 100% | 95.0% | 0.065 | 0.000015 | 0.00194 | 0.000001 | 0.2892 |
| Poland | 100% | 0% | 0% | 9.9% | 1.176[a] | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Romania | 100% | 0% | 0% | 69.3% | 0.401 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Slovakia | 100% | 0% | 0% | 69.3% | 0.401 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| Slovenia | 100% | 0% | 0% | n.a. | n.a. | n.a. | 0.00194 | 0.06 | 0.2892 |
| Spain | 100% | 0% | 0% | 69.3% | 0.401 | 0.060 | 0.00194 | 0.06 | 0.2892 |
| United Kingdom | 100% | 0% | 100% | 97.0% | 0.103 | 0.000015 | 0.00194 | 0.000001 | 0.2892 |

[a] Note that the low recovery rate for Poland and the resulting high implied emission factor are in consistency with national reporting of associated gas vented from oil production (UNFCCC, 2019).

Fugitive $CH_4$ emissions from unintended leakage during oil and gas production are estimated separately from fugitive emissions from venting and flaring of associated gas. Unintended leakage is usually irregular and therefore highly uncertain. In GAINS, fugitive leakage emissions from oil and gas production are calculated as follows:

$$E_{it;oil}^{leakage} = A_{it}^{oil} * \left(\gamma_i ef_{offshore}^{oil} + p_i ef_{oilsand} + (1-\gamma_i - p_i)\left((1-c_i) ef_{onshore}^{heavy} + c_i ef_{onshore}^{conventional}\right)\right) \quad (18)$$

$$E_{it;gas}^{leakage} = A_{it}^{gas} * \left(\gamma_i ef_{offshore}^{gas} + (1-\gamma_i) ef_{onshore}^{gas}\right) \quad (19)$$

| | |
|---|---|
| where $A_{i\,t}$ | is the energy content of marketable oil (or gas) produced in country *i* in year *t*, |
| $\gamma_i$ | is the fraction of oil (or gas) produced offshore, |
| $p_i$ | is the fraction of oil produced from oilsands, |
| $ef_{oilsand}$ | is the IPCC default emission factor for oil production from oilsands, |
| $c_i$ | is the fraction of conventional oil produced, |
| $ef_{offhore}^{oil;gas}$ | is the IPCC default emission factor for fugitive emissions from oil/gas produced offshore, |
| $ef_{onshore}^{conv}$ | is the IPCC default emission factor for fugitive emissions from conventional oil produced onshore |
| $ef_{onhore}^{heavy}$ | is the IPCC default emission factor for fugitive emissions from heavy oil produced onshore |
| $ef_{onshore}^{conv}$ | is the IPCC default emission factor for fugitive emissions from conventional oil produced onshore |
| $ef_{offshore}^{gas}$ | is the IPCC default emission factor for fugitive emissions from natural gas produced offshore |
| $ef_{onshore}^{gas}$ | is the IPCC default emission factor for fugitive emissions from natural gas produced onshore |

GAINS uses IPCC (2006, Vol.2, Tables 4.2.4 and 4.2.5) default emission factors as specified separately for developed countries and developing/transitional countries. All EU countries have here been treated as developed countries, adopting the median of the emission factor range given by IPCC as default. Adopted fugitive emission factors and IPCC ranges for default emission factors are presented in Table 9.

There are several cost-effective and low cost options available to reduce unintended leakage during extraction and processing of oil and natural gas (USEPA, 2014a, 2016; ICF International, 2016). But addressing leakages requires that these are first detected. Recent development of LDAR technologies, in particular the use of infrared cameras, has lowered the costs of leak detection significantly (ICF International, 2016; USEPA, 2016; McCabe and Fleischmann, 2014). In a survey of LDAR programs in Europe installed to reduce unintended leakages from gas production, transportation and storage facilities, Saunier et al., (2017) find that when used regularly and systematically, LDAR effectively detects leakages of which 61 percent are successfully repaired and reducing emissions by at least 90 percent, while 31 percent are less successfully repaired and reducing emissions by less than 50 percent and sometimes even increasing emissions. In an industry survey of US oil and gas facilities, ICF International (2016) finds that if all facilities are subject to annual LDAR emission surveys, an overall emission reduction of 40 percent is feasible. Drawing on these two studies, we assume it possible to reduce emissions from unintended leakages by 45 percent when LDAR technology is implemented across all facilities.

The cost of LDAR programs is likely to be highly site-specific and to vary with the gas price as reduced gas leakages means higher profits from gas sales. After detection of leakages, there is a long list of possible repairs that are available at a wide range of costs (see e.g., Table 3-1 in ICF International, 2016). As we do not have access to industry data on the incidence of different types of leakages in European oil and gas systems, it is not possible to make an assessment of the expected number and types of repairs that will be needed and the associated costs. Such assessments exist for US gas and oil systems, usually based on detailed data reported by industry to the USEPA and complemented by industry surveys (USEPA, 2014a; ICF International, 2016). To estimate costs for gas leakages repairs in the EU countries, we have sought to align the assumptions with the ranges for the US marginal abatement costs estimated for different industry segments (i.e., production, processing, transmission and distribution). It should however be stressed that the estimated abatement costs for these emission sources are highly uncertain and may be subject to revisions in the future if data that is more representative for European conditions become available.

Recently, a few EU countries have shown an interest in exploring the possibilities for large-scale extraction of shale gas. According to EIA (2011) some EU countries hold considerable reserves of technically recoverable shale gas. A survey by AEAT (2012) of the current knowledge of CH$_4$ emissions from shale gas extraction shows that uncertainty in emissions is still very high. The PRIMES (E3modelling, 2020) energy scenario does not include shale gas extraction in future projections. Therefore all CH$_4$ emissions from future gas production in the EU are estimated using emission factors for conventional gas extraction.

Table 9: Default emission factors for unintended fugitive emissions from oil and gas production used in GAINS and in comparison with IPCC (2006, Vol.2, Tables 4.2.4 and 4.2.5).

| | | **Developed countries** | | **Developing/transitional countries** | |
|---|---|---|---|---|---|
| | | **GAINS** | **IPCC (2006) range** | **GAINS** | **IPCC (2006) range** |
| | | **kt CH$_4$/PJ** | **kt CH$_4$/PJ** | **kt CH$_4$/PJ** | **kt CH$_4$/PJ** |
| Oil production | Conventional oil on-shore | 0.06 | 0.00004-0.094 | 0.12 | 0.00004-1.5 |
| | Heavy oil on-shore | 0.1863 | 0-0.3726 | 0.3726 | 0.1863-3.066 |
| | Conventional and heavy oil off-shore | 0.000015 | 0-0.00003 | 0.000015 | 0.000013-0.00013 |
| | Oilsands | 0.0542 | 0.0135-0.095 | 0.0542 | 0.018-0.135 |
| Gas production | Natural gas on-shore | 0.06 | 0-0.12 | 0.12 | 0.1-2.15 |
| | Natural gas off-shore | 0.00974 | 0-0.0195 | 0.00974 | 0.0058-0.034 |

### 3.2.4 *Crude oil transportation and refining*

$CH_4$ emissions from oil refinery and transport are fugitive emissions related to evaporation losses from storage, filling and unloading activities as well as fugitive leaks (IPCC, 2006, Vol.2, p. 4.34). The IPCC (2006, Vol.2, pp.4.52-4.61) guidelines provide emission factors for oil transportation based on the amount of oil transported, while emission factors for refining and storage are based on the amount of oil input to refineries as taken from PRIMES (E3modelling, 2020). Since it is not possible to find systematic data on the amount of oil transported by tanker, trucks or rails by region, GAINS assumes that the amount transported corresponds to the amount of oil refined. Thus, to calculate emissions from this source the activity data used is amount of oil refined combined with IPCC default emission factors for the sum of oil refined and transported:

$$E_{it} = \sum_m \left[A_{it} * \left(ef^{refined} + ef^{transported}\right) * (1 - remeff_m) * Appl_{itm}\right], \tag{20}$$

where $A_{it}$ is amount of oil refined in country *i* in year *t*,

$ef^{refined}$ is the IPCC default emission factor for oil refined,

$ef^{transported}$ is the IPCC default emission factor for oil transported,

$remeff_{im}$ is the removal efficiency of technology m, and

$Appl_{itm}$ is the application rate of technology *m* to emissions in country *i* in year *t*.

IPCC default emission factors for this sector are presented in Table 10 together with GAINS assumptions for no control and controlled emission factors.

The maximum technically feasible reduction is defined by the sum of the lower range IPCC default emission factor for oil refined and a fifty percent reduction in leakage emissions from oil transportation. This corresponds to the relative reduction in leakage emissions considered technically feasible for oil and gas production. Costs for these measures are taken from AEAT (1998).

Table 10: Default emission factors for $CH_4$ emissions from oil refinery and transport. Source: Derived from IPCC (2006, Vol.2, pp.4.52-4.61)

| **Emission source** | **GAINS** | | **IPCC (2006)** | **unit** |
|---|---|---|---|---|
| | **No control** | **Control** | | |
| Fugitive emissions at oil refinery | 0.0455 | 0.0029 | 0.0029 to 0.0455 | kt $CH_4$/Mt oil refined |
| Fugitive emissions from transport by pipeline | 0.0049 | 0.00245 | 0.0049 | kt $CH_4$/Mt oil transported by pipeline |
| Fugitive emissions from transport by tanker, truck and rail cars | 0.0225 | 0.0166 | 0.0225 | kt $CH_4$/Mt oil transported by tanker/truck |
| All sources | 0.0729 | 0.0166 | 0.0303 to 0.0729 | kt $CH_4$/Mt oil refined |

### 3.2.5 Long-distance natural gas transportation

Leakage of $CH_4$ emissions from long-distance gas pipelines arises for several reasons, e.g., untight compressor seals and valves or because pipelines are flushed with gas during start-ups (Ecofys, 1998). For Tier 1 estimations the IPCC guidelines (2006, Vol. 2, p.4.48-4.62) recommend the use of default emission factors per million $m^3$ of marketable gas. We find it problematic to use the IPCC default factors here as emissions from gas transmission are likely to be influenced by both the volume of gas transported, the distance the gas is transported as well as pipeline pressure and maintenance. Leakage rates are therefore likely to be specific to the circumstances prevalent in each country and pipeline system. For this reason, GAINS uses country-specific volumes of gas transported and leakage rates as reported to UNFCCC-CRF (2019) whenever such information is available. Table 11 provides an overview of the data compiled in GAINS for deriving amounts of gas transported and leakage rates for year 2015 in EU-27 countries. The leakage rate for Denmark is adopted as the controlled emission factor, representing the lowest technically feasible leakage rate reported. The percent control implemented is a reflection of the leakage rate reported by countries (UNFCCC-CRF, 2019).

For future years, activity data is assumed to grow proportionally with country gas consumption while leakage rates of 2015 are maintained into the future.

Costs for reducing methane emissions from gas transmission pipelines reflect the cost of replacing, retrofitting and maintenance of high-bleed pneumatics as estimated by USEPA (2006).

Table 11: Documentation of how methane emission factors for long-distance gas transmission pipelines were derived in GAINS for year 2015.

| | Activity data for year 2015 | | | CH4 emissions reported for year 2015 to UNFCCC-CRF (2019) | | | GAINS model estimates for year 2015 | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Country | Transmission network length (Marcogaz, 2017) km | Amount of gas transported through on-shore pipelines PJ | Source activity data | Transmission & Storage kt CH4 | Implied emission factor kt CH4/PJ | Leakage rate when 50 MJ/kg CH4 % leak | Transmission & Storage kt CH4 | No control emission factor when leakage rate is 0.3% ktCH4/PJ | Controlled emission factor (leakage 0.001%) ktCH4/PJ | Assumed current control corresponding to reported implied ef % of max technically feasible control |
| Austria | 3007 | 1242 | TAG (2007; 2015; 2020) | 3.90 | 0.0031 | 0.016% | 3.90 | 0.06 | 0.0001 | 94.9% |
| Belgium | 4057 | 571 | UNFCCC-CRF (2019) | 6.58 | 0.0115 | 0.058% | 6.58 | 0.06 | 0.0001 | 80.9% |
| Bulgaria | 1835 | 500 | UNFCCC-CRF (2019) | 3.69 | 0.0074 | 0.037% | 3.69 | 0.06 | 0.0001 | 87.9% |
| Croatia | 2034 | 96 | UNFCCC-CRF (2019) | 1.21 | 0.0126 | 0.063% | 1.21 | 0.06 | 0.0001 | 79.1% |
| Cyprus | n.a | 0 | UNFCCC-CRF (2019) | NO | n.a | n.a. | n.a | 0.06 | 0.0001 | n.a |
| Czech Republic | 3810 | 1215 | UNFCCC-CRF (2019) | 6.22 | 0.0051 | 0.026% | 6.22 | 0.06 | 0.0001 | 91.6% |
| Denmark | 831 | 201 | UNFCCC-CRF (2019) | 0.03 | 0.0001 | 0.001% | 0.03 | 0.06 | 0.0001 | 99.9% |
| Estonia | 885 | 16 | UNFCCC-CRF (2019) | 0.04 | 0.0025 | 0.013% | 0.04 | 0.06 | 0.0001 | 95.9% |
| Finland | 1318 | 94 | UNFCCC-CRF (2019) | 0.37 | 0.0039 | 0.020% | 0.37 | 0.06 | 0.0001 | 93.6% |
| France | 37246 | 1467 | UNFCCC-CRF (2019) | 24.0 | 0.0163 | 0.082% | 24.0 | 0.06 | 0.0001 | 72.9% |
| Germany | 62500 | 1270 | UNFCCC-CRF (2019) Gas consumed | 103 | 0.0162[a] | 0.081%[a] | 103 | 0.06 | 0.0001 | 73.2% |
| Greece | 1819 | 121 | UNFCCC-CRF (2019) | 0.94 | 0.0078 | 0.039% | 0.94 | 0.06 | 0.0001 | 87.2% |
| Hungary | 5782 | 321 | UNFCCC-CRF (2019) Marketable gas | 2.55 | 0.0079 | 0.040% | 2.55 | 0.06 | 0.0001 | 86.9% |
| Ireland | 2417 | 155 | UNFCCC-CRF (2019) | 0.27 | 0.0017 | 0.009% | 0.27 | 0.06 | 0.0001 | 97.3% |
| Italy | 34415 | 2562 | UNFCCC-CRF (2019) Gas transported | 31.0 | 0.0121 | 0.060% | 31.0 | 0.06 | 0.0001 | 80.0% |
| Latvia | 1193 | 1650 | UNFCCC-CRF (2019) | 0.03 | 0.0000 | 0.000% | 0.17 | 0.06 | 0.0001 | 100.0% |
| Lithuania | 2113 | 385 | UNFCCC-CRF (2019) Derived from reported leakage | 3.54 | 0.0092 | 0.046% | 3.54 | 0.06 | 0.0001 | 84.8% |
| Luxembourg | 1962 | 32 | UNFCCC-CRF (2019) | 0.42 | 0.0130 | 0.065% | 0.42 | 0.06 | 0.0001 | 78.4% |
| Malta | 0 | 0 | UNFCCC-CRF (2019) | n.a | n.a. | n.a. | n.a | 0.06 | 0.0001 | n.a |
| Netherlands | 11896 | 3000 | UNFCCC-CRF (2019) | 7.21 | 0.0024 | 0.012% | 7.21 | 0.06 | 0.0001 | 96.2% |
| Poland | 10077 | 577 | UNFCCC-CRF (2019) Gas consumed | 8.79 | 0.0152 | 0.076% | 8.79 | 0.06 | 0.0001 | 74.7% |
| Portugal | 1298 | 0.168 | UNFCCC-CRF (2019) | 0.04 | 0.2379 | 1.190% | 0.01 | 0.06 | 0.0001 | 0% |
| Romania | 13110 | 489 | UNFCCC-CRF (2019) Gas produced | 3.13 | 0.0064 | 0.032% | 3.13 | 0.06 | 0.0001 | 89.5% |
| Slovakia | 8533 | 2116 | UNFCCC-CRF (2019) | 26.8 | 0.0127 | 0.063% | 26.8 | 0.06 | 0.0001 | 79.0% |
| Slovenia | 1094 | 31 | UNFCCC-CRF (2019) | 0.31 | 0.0100 | 0.050% | 0.31 | 0.06 | 0.0001 | 83.4% |
| Spain | 12987 | 1030 | UNFCCC-CRF (2019) | 2.42 | 0.0023 | 0.012% | 2.42 | 0.06 | 0.0001 | 96.2% |
| Sweden | 600 | 103 | UNFCCC-CRF (2019) Derived from reported leakage | 0.08 | 0.0008 | 0.004% | 0.08 | 0.06 | 0.0001 | 98.9% |
| United Kingdom | 7648 | 1773 | UNFCCC-CRF (2019) | 3.25 | 0.0018 | 0.009% | 3.25 | 0.06 | 0.0001 | 97.1% |

[a] Due to the low domestic gas consumption relative the long km gas transmission networks in Germany, the implied emission factor and leakage rate were adjusted down by a factor 5 to be comparable to other countries.

### 3.2.6 Consumer gas distribution networks

$CH_4$ emissions from gas use come from leakage in consumer distribution networks and during end-use by consumers. The activity data is amount of gas consumed by sector as provided by the PRIMES (E3modelling, 2020) model for the EU-27 countries and United Kingdom. IPCC (2006, Vol.2, Tables 4.2.4 and 4.2.5) provide Tier 1 default emission factors for developed and developing/transitional countries. These correspond to default leakage rates of 0.15 and 0.35 percent, respectively, with uncertainty ranges up to ± 500 percent. Because of the wide uncertainty range of the IPCC default factors, we have calibrated GAINS emission factors to the country-specific leakage rates as reported for year 2015 to the UNFCCC-CRF (2019).

To assess future mitigation potentials we need to attribute reported emissions to residential and non-residential users (i.e. power plants and industry), respectively. Dennett and Vallender (2011) provide measurement results for the UK on how the leakage from residential mains compares to leakage from mains supplying larger users (power plants and industry). The UK measurements show that total gas losses from gas distribution grids in the UK are 0.57 percent of throughput, with 80 percent from low pressure mains systems supplying small consumers. This translates into losses from residential and non-residential sectors of 1 and 0.23 percent of gas consumption, respectively. The same proportional split (i.e., 23%) between losses in residential and non-residential sectors is assumed in GAINS for all EU countries.

A technically feasible range for leakage from gas distribution mains in the EU-28 was determined from the range of leakage reported by countries to UNFCCC-CRF (2019) and maintaining the assumption from the UK measurements that leakage from mains supplying larger users like power plants and industry is 23% of the leakage from mains supplying residential consumers. This corresponds to a leakage rate from 0.003% to 1.15% for residential distribution mains and from 0.0007% to 0.26% for non-residential distribution mains. The calibration in GAINS to the reported emissions is done by adapting assumptions of percent control implemented and assuming the same level of control for residential and non-residential gas users.

For future years emissions are driven by the development in gas consumption by sector as provided by the PRIMES (E3modelling, 2020) model, leakage rates remain constant unless there are efforts made to upgrade the network.

$CH_4$ emissions from consumer distribution networks can be reduced by increased frequency of leakage control using LDAR technology (see Section 3.2.2 for details) or by replacing old town gas distribution networks made from grey cast iron by polyethylene (PE) or polyvinylchloride (PVC) networks. This typically reduces almost all fugitive emissions from this source (AEAT, 1998). In the baseline scenarios developed here, assumptions about the current state of gas distribution networks in the EU member states reflect the relative leakage rates reported by countries to the UNFCCC (2019), see Table 12. The average leakage rate reported for Denmark (0.0077%) has been used as benchmark, as the Danish gas distribution network was first introduced in the mid-1980s and is of relatively recent date compared with other European countries (Aarhus Universitet, 2015). The level of control installed in 2005 and 2010 is derived by relating the reported leakage rates to the benchmark leakage rate as shown in Table 12. The control level of 2015 remains constant in future years. Mitigation costs for replacement of gas distribution networks are taken from AEAT (1998).

Table 12: Documentation of how methane emission factors and implemented control for fugitive emissions from gas distribution networks were derived in GAINS for year 2015.

| | Information taken from UNFCCC-CRF (2019) | | GAINS estimation of CH4 emissions from gas distribtuion networks in year 2015 | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Residential | | Non-residential | | | | | |
| | Gas consumption (PJ) as reported by member states for year 2015 | CH4 from gas distribution as reported by member states for year 2015 | No control emission factor (leakage: 1.15%) | Controlled emission factor (leakage: 0.003%) | No control emission factor (leakage: 0.26%) | Controlled emission factor (leakage: 0.0007%) | Current control as % of max technically feasible control (assumed same for residential and non-residential) | GAINS total emissions match emissions reported to UNFCCC (2019) | | |
| | | | Residential | | Non-residential | | | Residential | Non-residential | Total |
| **Country** | **PJ** | **kt CH4** | **kt CH4/PJ** | **kt CH4/PJ** | **kt CH4/PJ** | **kt CH4/PJ** | | **kt CH4** | **kt CH4** | **kt CH4** |
| Austria | n.a. | 1.55 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 94.9% | 0.86 | 0.69 | 1.55 |
| Belgium | 571.5 | 14.6 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 81.6% | 10.65 | 3.93 | 14.58 |
| Bulgaria | 114.3 | 3.39 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 31.8% | 0.98 | 2.41 | 3.39 |
| Croatia | 29.1 | 0.84 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 91.4% | 0.56 | 0.28 | 0.84 |
| Cyprus | 0 | 0 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 100.0% | 0.0 | 0.0 | 0.0 |
| Czech Republic | 302.3 | 16.3 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 64.7% | 12.10 | 4.15 | 16.25 |
| Denmark | 97.4 | 0.15 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 99.3% | 0.09 | 0.06 | 0.15 |
| Estonia | 15.8 | 0.59 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 80.0% | 0.21 | 0.38 | 0.59 |
| Finland | 7.6 | 0.78 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 90.3% | 0.07 | 0.71 | 0.78 |
| France | 1467.3 | 20.6 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 91.6% | 16.98 | 3.62 | 20.60 |
| Germany | 1270.1 | 113 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 73.9% | 84.95 | 27.83 | 112.78 |
| Greece | 59.0 | 1.68 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 82.9% | 0.70 | 0.98 | 1.68 |
| Hungary | 390.7 | 11.5 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 77.9% | 9.76 | 1.70 | 11.46 |
| Ireland | 71.9 | 2.12 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 87.5% | 1.27 | 0.85 | 2.12 |
| Italy | 1233.2 | 141.8 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 55.2% | 112.23 | 29.58 | 141.81 |
| Latvia | 0.2 | 3.43 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 0.5% | 1.95 | 1.48 | 3.43 |
| Lithuania | n.a. | 8.04 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 0.0% | 2.14 | 4.12 | 6.26 |
| Luxembourg | 32.2 | 0.96 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 81.8% | 0.71 | 0.25 | 0.96 |
| Malta | n.a. | n.a. | 0.23 | 0.0006 | 0.0529 | 0.000138 | n.a. | n.a. | n.a. | n.a. |
| Netherlands | n.a. | 5.73 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 97.2% | 4.38 | 1.35 | 5.73 |
| Poland | 576.8 | 20.6 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 71.9% | 15.20 | 5.41 | 20.61 |
| Portugal | 0.1 | 1.87 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 82.3% | 0.82 | 1.05 | 1.87 |
| Romania | 273.8 | 69.0 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 0.0% | 32.48 | 13.92 | 46.40 |
| Slovakia | 176.0 | 5.10 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 82.1% | 3.87 | 1.23 | 5.10 |
| Slovenia | 30.9 | 0.90 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 64.5% | 0.52 | 0.38 | 0.90 |
| Spain | 1042.5 | 24.4 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 75.5% | 14.03 | 10.40 | 24.43 |
| Sweden | n.a. | 1.36 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 62.9% | 0.61 | 0.75 | 1.36 |
| United Kingdom | n.a. | 149 | 0.23 | 0.0006 | 0.0529 | 0.000138 | 65.3% | 123.08 | 25.79 | 148.87 |

### 3.2.7 Combustion in stationary sources

During combustion processes $CH_4$ emissions are released due to incomplete oxidation of fossil and biogenic fuels. Activity data for combustion emissions from power plants is the energy content of the fuel used, which is taken from the PRIMES (E3modelling, 2020) model. Emission factors are differentiated for different sectors and type of fuel combusted. For industry and power plants, $CH_4$ emission factors were taken from IPCC (2006). For residential boilers, $CH_4$ emission factors are differentiated for different types of boilers and taken from Delmas (1994), Johansson et al. (2004), Kjällstrand and Olsson (2004) and Olsson and Kjällstrand (2006).

No specific mitigation options have been identified for $CH_4$ emissions from combustion sources.

### 3.2.8 Combustion in mobile sources

Incomplete combustion in mobile sources gives rise to $CH_4$ emissions. Activity data is amount of fuel used as adopted from the PRIMES (E3modelling, 2020) model for each vehicle type as specified in Table 3. GAINS uses default emission factors from the European transport emission database COPERT (EMISIA, 2013 http://www.emisia.com/copert/General.html). Emission factors are specified by vehicle category, type of fuel used and the emission control standard of the vehicles (EURO I to EURO VI).

## 3.3 Agriculture sector

### 3.3.1 Livestock

#### 3.3.1.1 Enteric fermentation and manure management emissions

$CH_4$ emissions from livestock emerge from enteric fermentation during the digestive process in the stomachs of ruminants. When the organic content in livestock manure decomposes, emissions of $CH_4$ and $N_2O$ are released. $CH_4$ release occurs when manure is handled under anaerobic conditions, while the formation of $N_2O$ requires aerobic conditions with access to oxygen.

Emissions from livestock are estimated as the sum of the emission types *n* (enteric fermentation and/or manure management) for a certain animal type *s* in country *i* and year *t*:

$$E_{its} = \sum_{bmns}\left[A_{itbs} * ef_{ibns}^{NOC} * (1 - remeff_{mns} * h_{itsm}) * Appl_{itslm}\right], \qquad (21)$$

where $A_{itbs}$ is the number of animals of type *s* in country *i* and year t, with type of manure management *b* (solid or liquid),

$ef_{is}^{NOC}$ is the no control emission factor for emission type *n*, animal type *s* in country *i* and subject to manure management *b*,

$remeff_{mns}$ is the removal efficiency of technology *m* when applied to emissions of type *n* and animal type *s*,

$h_{itsm}$ is a factor correcting for application limitations of technology *m*, e.g., indoor housing rates for feed options or large farm rate for farm-scale anaerobic digestion,

$Appl_{itsbm}$ is the application rate of technology *m* to animal type *s* with type of manure management *b*, in country *i* and year *t*.

GAINS estimates $CH_4$ emissions separately for the animal types dairy cows, non-dairy cattle, pigs, poultry, sheep and goats, buffaloes, and horses. For dairy cows, non-dairy cattle and pigs, animal numbers are further split by whether animals are subject to liquid or solid manure management. The split in the number of animals by liquid or solid manure management is stored in the GAINS model and was recently reviewed by member state experts during IIASA consultations in preparation of the GAINS database for the Clean Air Outlook 1 (in 2018) and 2 (in 2020), see EC (2020). The source for historical animal numbers for historical years is EUROSTAT (2019), except for horses and buffaloes where FAOSTAT (2019) is the source. Projections are based on future trends in animal numbers as estimated by the CAPRI (EuroCare, 2021) model.

A recent improvement in the GAINS model is the split of animal categories dairy cows, non-dairy cattle, pigs, poultry, sheep and goats by five farm size classes, i.e., less than 15 livestock units (LSU), 15 to 50 LSU, 50 to 100 LSU, 100 to 500 LSU, and above 500 LSU. The source for data on historical farm-size distributions is EUROSTAT (2015). Projections for the future development of farm-size classes have been made applying a multi-nominal logistic function weighing in the development observed in historical years from 1990 onwards. The development of farm-size classes has implications for the development of the fractions of animals on liquid and solid manure management and on the future applicability of control technology options. Typically, over time more animals tend to move into the larger farm-size classes with liquid manure management and away from smaller farm-size classes with solid manure management.

Country-specific emission factors corresponding to the implied emission factors reported to UNFCCC-CRF (2019) for year 2015 were adopted for enteric fermentation and manure management emissions, respectively.

For dairy cows, both enteric fermentation emissions and manure management emissions per animal are affected by the milk productivity of the cow. This effect is particularly accentuated for highly productive milk cows. To capture this, the no control emission factor for dairy cows is specified as the sum of a fixed emission factor per animal for cows producing up to 3000 kg per head per year and an additional term describing the emission factor per milk yield for milk production exceeding the productivity level 3000 kg per animal per year, i.e.,

$$ef_{it;cow}^{NOC} = ef_i^{animal} + ef_i^{milk} * (x_{it} - 3000) \tag{22}$$

where $ef_i^{animal}$ is the default emission factor for cows in country *i* producing 3000 kg milk per year,

$ef_i^{milk}$ is the emission factor per kt milk produced above the threshold level 3000 kg milk per animal per year, and

$x_{it}$ is the average milk yield per animal in country *i* and year *t*.

A linear relationship between the CH4 emissions per cow and the milk yield per cow was fitted from default emission factors per cow provided by IPCC (2006, Ch.10) for different world regions representing a range of average milk yield levels. Starting from country-specific implied emission factors reported to UNFCCC for year 2005, the derived linear relationship determine the future emission pathway from the development in milk yield and animal numbers. Hence, as milk yield per cow increases $CH_4$ emissions per cow increase while emissions per kg milk produced decline when fewer animals are needed to produce the same amount of milk. Whether the overall effect on methane emissions is positive or negative will depend on the importance of the effect of increased methane emissions per animal relative the effect of declining animal numbers. The derived $CH_4$ emissions per kg milk produced at different average milk yield levels are well in line with the ranges estimated empirically for Germany between 1990 and 2010 (Piatkowski and Jentsch, 2013).

### 3.3.1.2 Enteric fermentation control options

#### 3.3.1.2.1 Breeding

Breeding through selection offers several different ways to reduce enteric methane emissions from livestock. First, an increase in productivity per animal could mean that a smaller animal stock produces the same amount of produce with potentially lower methane emissions per unit of produce. Milk cow productivity has improved significantly in the EU over the past few decades. This is primarily the result of breeding leading to an increased use of high-producing cow types, e.g., Holsteins, in the stocks. If the increase in milk yield per cow means keeping smaller animal stocks to produce the same amount of milk, then methane emissions per unit of milk produced are likely to fall. In many EU countries, methane emissions per unit of milk produced may, however, not have fallen as a consequence of increased milk yield because the tremendous increase in milk yield in many countries has been accompanied by an increased fraction of unproductive animals in the stocks due to poorer health and fertility of the high-productive cow breeds (see e.g., Lovett et al., 2006; Berglund, 2008; Bell et al., 2011). Highly productive cows are effective in converting feed energy into milk, but may as a consequence lack enough energy reserves to maintain health and reproduction (Berglund, 2008). Many EU countries have seen falling reproductive performance of the milk cows as milk yield increased (Nyman et al., 2014; Nilforooshan et al., 2010; Nilforooshan et al., 2013; Evans et al., 2006), however, the breeding schemes in the Scandinavian EU countries have shown that it is possible through breeding for both productivity and reproduction traits, to achieve top level milk yield *and* maintain fertility and animal health levels (Berglund, 2008). Hence, to achieve lasting methane reductions per unit of milk produced, it is likely that breeding programs must aim at the dual objective of increasing milk yield while maintaining animal health and fertility. This appears to be the case not only for animals fed on concentrates, but also for forage-fed and grazing animals (Bell et al., 2011; O'Brien et al., 2010). Maintaining animal health and fertility levels is likely to better sustain profitability of the milk farmers in the long-run, as it is costly to keep and feed large fractions of unproductive animals in the stock (Evans et al., 2006). According to Bell et al. (2012) and the conclusions of the EU 7th Framework Programme "Animal change" project (Animal change, 2014), genetic improvement through breeding is likely to be a cost-effective measure as it achieves methane emission reductions that are cumulative and permanent. O'Brien et al. (2014a) estimate reduction potentials in 2020 from adoption of accelerated breeding schemes for Irish dairy cows at 555 kt $CO_2$eq per year when using a lifecycle assessment method and by 596 kt $CO_2$eq per year when using the IPCC national inventory approach.[2] The mitigation cost is estimated at -519 and -483 €/t$CO_2$eq with respective methodology. There are no estimates available in the literature of what a dual objective breeding scheme could achieve in terms of methane reductions at an EU level. Several authors however mention that such a development would require large databases on the genetic variation to

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[2] Note that it is not clear from the source from what baseline these emission reductions are estimated. However, if we use GAINS baseline estimates, then the accelerated breeding option would reduce dairy cow emissions in the order of 16 to 18%.

favourably capture the complex interactions of different positive genetic traits (De Haas et al., 2011; Berry and Crowley, 2013). In addition, time is needed for a careful selection process to run its course, often mentioned to at least 10 years (Knapp et al., 2014; De Haas et al., 2011).

Second, breeding can be used to enhance the feed efficiency in animals, which is likely to reduce methane emissions as methane production is essentially a loss of energy. Profitability in meat production systems is closely linked to the feeding costs, which makes it particularly interesting for financial and environmental reasons to use breeding to enhance feed efficiency in animals and simultaneously reduce methane emissions. Berry and Crowley (2013) show in a meta-analysis of 39 scientific publications that genetic variation in feed efficiency in cattle has a pooled heritability for residual feed intake (RFI) and feed conversion efficiency of 0.33 and 0.23, respectively (for cows, the corresponding heritability is much lower at 0.06 and 0.04, respectively). Hence, there appears to be scope for further improvements in feed efficiency through breeding in beef cattle, provided that large quantities of routinely collected feed intake information on individual animals become available (Berry and Crowley, 2013). The link between feed efficiency in cattle and methane emissions is expected. Nkrumah et al. (2006) linked the variation in RFI directly to variation in methane emissions. They found for 27 Canadian hybrid beef steers that methane production was 25% lower for low-RFI individuals than for high-RFI individuals. Hegarty et al. (2007) found among 66 Angus steers that methane emissions were lower for the most feed efficient animals (i.e., with the lowest RFI). The effect on methane emissions from selection for feed efficiency is likely to be the greatest for grazing ruminants, which includes dairy cows, non-dairy beef cattle and sheep (Animal Change, 2014). According to estimates by DEFRA (2012), 10 years of breeding on beef cattle in the UK using current approaches is expected to result in a cumulative increase in farmers' profits of 31 million pounds when measured over a twenty years period and reduce GHG emissions by 3%. Adding more selection indices to the traits currently targeted, is expected to increase profits further and extend the emission reduction potential to 10% below current emission levels.

Third, there is the possibility to use breeding to directly select for individual animals with low formation of methane in the rumen, thereby reducing enteric methane emissions per animal. De Haas et al. (2011) state that, on the basis of the existing genetic variation in Dutch dairy cows, it is theoretically possible to reduce enteric methane production per animal by selecting for more methane efficient individuals. This potential would correspond to enteric methane reductions in the order of 11 to 26% over 10 years. Although theoretically possible, realistic methane reduction rates will depend on several factors, e.g. the accuracy and intensity of selection, the number of records available, and the generation interval (De Haas et al., 2011). De Haas et al. call for an international effort to establish a reference database on genetic variation in dairy cows to be used for genomic selection.

Fourth, breeding has the potential to reduce methane emissions if it is combined with different types of feed as different breeds appear to respond with different levels of enteric methane formation to different types of feed (Bell et al., 2012; Bell et al., 2010; Knapp et al., 2014; O'Brien et al., 2010). In addition to matching feed with the breed, Knapp et al. also mention the importance of optimizing management practices, which could otherwise limit the ability of the animals to reach their full genetic potential.

Finally, in addition to breeding through selection of traits, there are a number of reproductive technologies available that have the potential to reduce methane emissions (Hristov et al., 2013b). These include gender-selected semen, embryo transfer, and hormonal synchronization, which all have the potential to enhance the reproductive efficiency, thereby reducing the number of animals kept in the stock at a systems level. With fewer animals in the system, greenhouse gases are likely to decline per unit of output produced. Hristov et al. (2013b) estimate the methane mitigating effect of assisted reproductive technologies to between 3.5 to 5.5 percent.

We have here described a number of different ways through which breeding through selection and the use of reproductive technologies can achieve persistent reductions in system methane emissions from milk and meat production. The uncertainty of the long-term effects on systems methane emissions is high and difficult to assess with more certainty. At the same time, when taken together it is most likely that these options can offer future opportunities to reduce enteric methane emissions if policies are put in place, which provide incentives for the adoption of reproductive technologies and for large-scale breeding schemes that specifically target reductions in methane emissions.

To estimate the abatement potential from breeding schemes aimed at enhancing productivity and animal health and fertility in dairy cows, we set a benchmark at 14.3 kt $CH_4$/kt milk produced, which corresponds to the average 2015 emission factor for Swedish dairy cows consistent with the reporting to UNFCCC-CRF (2019). The assumption is that with this type of broader breeding schemes set up from 2020 onwards, all dairy cows on farms with more than 50 LSU ought to, from 2040 onwards, be able to reach the same emission level per unit of milk produced as Swedish cows had in 2015. Considering that successful breeding schemes take time, the effect on emissions are assumed to be gradually phased in between 2030 and 2040. The country-specific abatement potentials from breeding of dairy cows are illustrated in Figure 1 as the difference between the baseline emission factor (orange line) and the target emission factor (grey line).

An abatement potential from different types of breeding of 10% is in GAINS assumed possible for non-dairy cattle and sheep from 2030 onwards. Estimates by DEFRA (2012) show that farmers' profits are likely to increase due to enhanced productivity, which would suggest a negative cost for breeding. However, there are likely to be costs involved for establishing a reference database on genetic information and associated research. Although uncertain, a general assumption in GAINS is that breeding as a measure to mitigate methane emissions is cost-effective (i.e., available at zero costs).

#### 3.3.1.2.2 Feed management and feed additives

Changing animal diets or adding substances known to reduce methane formation in the rumen are possible methane mitigation options. Reviews from the FAO by Hristov et al. (2013a) and Gerber et al. (2013a) provide comprehensive information about the current state of knowledge with a global scope. Both reports provide limited information for Europe specifically. Gerber et al., provide information about the expected effect on mixed dairy systems in Western Europe of using lipids as feed additives and find it small (expected reduction in enteric fermentation emissions is 1.2% to 3.6%). The effects of different types of non-lipid additives or feed management options were, however, not evaluated for Western Europe by Gerber et al. (2013a).

Feed management options include mechanical ways to treat the feed to facilitate digestion, ways to combine different types of feed to minimize enteric methane formation, as well as precision feeding, which means very closely monitoring the timing and the feed mix supplied to the animals in order to optimize feeding against both economic and environmental parameters. As an effective feed management is important for the profitability of many European farmers, the scope for further reductions in methane emissions through improved efficiency in feed management is likely to be limited. There may, however, be some scope for mitigation from wider use of precision feeding (Dalgaard et al., 2011).

Knapp et al. (2014) review enteric fermentation methane mitigation measures for intensive dairy systems. They conclude that feeding and nutrition have modest reduction potential (2.5 to 15%) to mitigate enteric methane in intensive dairy operations in developed countries and that "To date, rumen modifiers other than nitrate have not shown sustained reductions in CH4/ECM". Dalgaard et al. (2011) analyze possible measures to reduce GHG emissions in Danish agriculture to 2050. They estimate that a 50-70% reduction in Danish agricultural greenhouse gas emissions is achievable in 2050 relative the 1990 emission level. The contribution from different feed options to methane emission reductions is, however, limited to between 15 and 20% and would involve precision feeding. Results compiled by Hristov et al., (2013a) on the effectiveness of individual feed mitigation measures show a mixed bag with limited potential for intensive European livestock systems from feed management and lipid additives (due to already high efficiency with respect to these parameters in the systems) but with potential effects from some non-lipids additives e.g., nitrate, propionate precursors and phytogenic substances.

Nitrate as feed additive is reported to have significant negative effects on methane emissions, but may also come with negative effects on animal health if the feeding is not closely monitored (Van Zijderveld et al., 2010, 2011b). The expected mitigation effects from propionate precursors as feed additives have been downplayed in recent studies, because effects shown in-vitro have not turned out to deliver in-vivo (e.g., Van Zijderveld et al., 2011b; Ungerfeld and Forster, 2011). Still, developments to bring a product based on propionate precursors to the market are on-going within the European "Clean Cow Project", which claims a minimum 25% mitigation potential to be possible for dairy cows and with plans to launch the additive on the market in 2018 (DSM, 2014). Phytogenic substances have shown some promising results as inhibitors of enteric methane, however, more research is still needed (Flachowsky and Lebzien, 2012). Propionate precursors are not considered as a separate mitigation option in GAINS due to the reasons mentioned in the previous paragraph.

As there are different options for reducing enteric methane through animal feed changes and it is difficult to speculate about the potential future abatement potential from each of the options, we define a broader mitigation category, which is meant to reflect a combined reduction potential from feed additives and/or changes in feed management practices. Despite the inconclusive results and the stated needs for further research, we conclude that the list of (potentially) effective feed options is long and that research is vivid and continues. We consider it likely that at least one or a few will be able to deliver some effect on methane emissions in the future. Hence, in addition to the effects of breeding mentioned above, we assume in GAINS that the combined mitigation effect in 2030 from different feed management changes and feed additives is a 10% reduction of enteric fermentation emissions in dairy cows and 5% of enteric fermentation emissions in non-dairy cattle and sheep during the time that animals are housed indoor. No effects on emissions from feed additives or feed management changes are accounted for during the time animals are kept outdoor. Information on the average number of days per year that animals spend indoor has been collected by animal category in the GAINS database during consultations with experts from EU member states, most recently during IIASA-member state consultations in preparation of the GAINS database for the Clean Air Outlook 1 (in 2018) and 2 (in 2020), see EC (2020).

Costs will depend on the feed additive in question (e.g., nitrate-based additives are likely to be very cheap while additives based on propionate precursors are likely to be more expensive) and on the possibility and costs of effectively distributing the additive in the right dosage in order to avoid negative effects on animal health. DSM (2014) mention that for the launch in 2018 of a new feed additive, they evaluated different options for dosage and distribution in order to offer a financially feasible mitigation option to farmers. With further technological development, we expect the launch of new additives on the market to be accompanied by a development in dosage technologies. With new climate policy incentives in place which effectively promotes the development of new technologies, we consider it feasible by 2030 to expect that new feed additives will become available on the market, which are both effective in terms of reducing emissions and come at a financially feasible cost to farmers. In the absence of information on expected future costs of this option, we consider a financially and politically viable annual cost level to be about 11 Euros per head. This cost level corresponds to an average cost range of about 30 to 60 Euro/t $CO_2$eq when implemented for dairy cows and with higher average cost levels for non-dairy cattle and sheep.

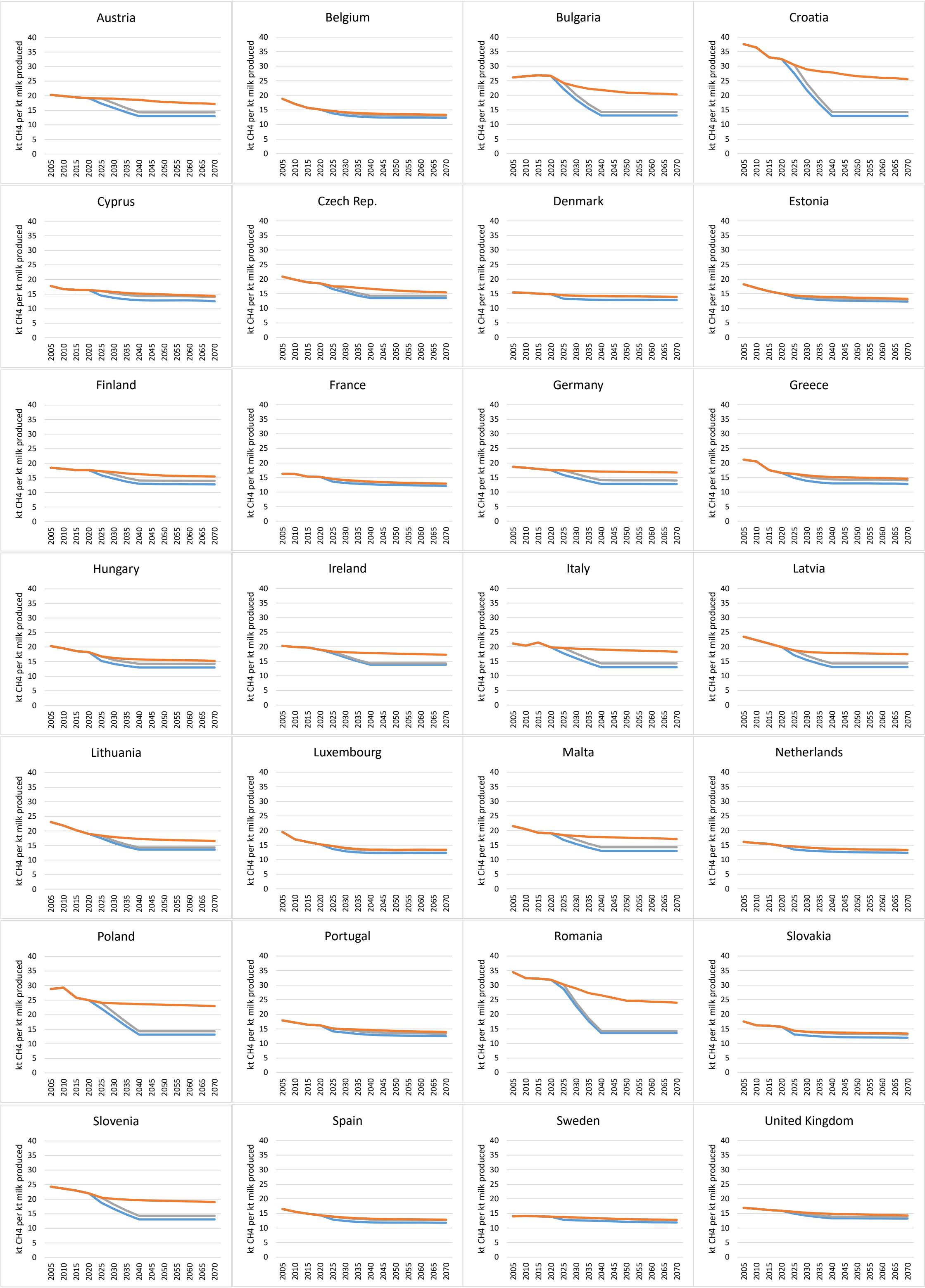

Figure 1: Benchmarking of enteric fermentation $CH_4$ abatement potential from breeding of dairy cows (target is 14.3 kt $CH_4$ per kt milk produced applicable to cows on farms with more than 50 LSU). An additional abatement potential of 10% is assumed possible from animal feed changes applicable to cows when housed indoor on farms with more than 50 LSU.

#### 3.3.1.3 Manure management option

Treatment of animal manure in anaerobic digesters (ADs) that generate biogas can be an efficient way to reduce methane emissions from manure handling at a low cost (Hristov et al., 2013b). The process has the advantage of not only reducing emissions, but also to generate energy that can be used on the farm or sold to external local users, and at the same time produces an odor-free organic fertilizer, which can substitute the use of mineral fertilizers (Sommer et al., 2013).

The minimum electricity generation capacity of co-generation plants currently available on the market is 15 $\mathrm{kW_{electric}}$ and requires at least 100 LSU (Pellerin et al., 2013). Hence, farm AD is likely to be economically infeasible for farms smaller than 100 LSU. In the GAINS model, farm-scale anaerobic digestion (AD) is assumed to be applicable to manure from dairy cows, non-dairy cattle and pigs kept in systems with liquid manure management on farms with at least 100 livestock units (LSU). It is also assumed that manure is only available for anaerobic digestion during the periods that animals are kept indoor. Information on the average number of days per year that animals spend indoor has been collected by animal category in the GAINS database during consultations with member state experts, most recently during IIASA-member state consultations in preparation of the GAINS database for the Clean Air Outlook 1 (in 2018) and 2 (in 2020), see EC (2020). No potential for farm-scale anaerobic digestion is assumed for animals kept on farms smaller than 100 LSU, nor for animals in solid manure management systems, and nor for periods when animals are grazing outdoor.

An additional possible source of manure as input to anaerobic digesters is poultry excretion. This source of substrate has been excluded from this analysis because of its insignificance as a methane emissions source.

The amount of manure available for anaerobic digestion is derived from the average volatile solid excretion rate per animal per day reported by countries to the UNFCCC-CRF (2019) for the year 2015. Given that volatile solids (VS) make up 80% of total solids (TS) in the manure and that the manure (or rather the slurry as it derives from liquid manure management systems) has a water content of 85% (Sommer et al., 2013), the amount of manure generated per head per year ($m$) in country $i$ is calculated as:

$$m_i = 0.001\frac{(365\times VS_i/0.8)}{(1-0.85)}, \tag{23}$$

where $VS_i$ is the country-specific average daily excretion rate for the analyzed animal type.

The cost of farm AD is sensitive to scale. We therefore estimate the methane reduction potential and costs separately for "large farms" defined as farms with 100 to 500 LSU and "extra large farms" with more than 500 LSU. The total amount of manure available for farm-scale anaerobic digestion in country $i$ in a future year $t$ is the sum of manure excreted by animals on large farms and extra large farms during times when animals are kept indoor, i.e.,

$$\mathrm{M_{it}} = \textstyle\sum_{\mathrm{s}} \mathrm{A_{it}^{liquid}} \times \mathrm{VS_i} \times \gamma_{\mathrm{its}}\; M_{it} = \sum_s A_{it}^{liquid} \times m_i \times \gamma_{its} \times h_i, \tag{24}$$

where $A_{it}^{liquid}$ is the number of animals on liquid manure management, $\gamma_{its}$ is the fraction of animals found on farms of size $s$ in country $i$ and year $t$, and $h_i$ is the fraction of a year that animals are housed indoor.

The cost of farm-scale AD is derived as the sum of the annualized investment cost and the operating costs (including costs for labour and additional organic substrate), minus the revenues and cost-savings of utilizing the generated electricity and fertilizers. Hence, the unit cost per head of installing a farm AD plant for treatment of cattle and pig manure is in GAINS defined for country $i$ in year $t$ and for farm scale $s$ in the following way:

$$C_{its} = I_{its} \times \frac{r(1+r)^T}{(1+r)^T-1} + R_{its}p_R + L_{its}w_{it} - 0.6p_{it}^{ind}E_{its} - F_{its}p_f \tag{25}$$

| | | |
|---|---|---|
| where | $I_{its}$ | is the fixed initial investment cost, |
| | $T$ | is the expected lifetime of the equipment, |
| | $r$ | is the interest rate, |
| | $R_{its}$ | is the amount of organic substrate added to the co-digestion, |
| | $p_R$ | is the unit price of organic substrate, |
| | $L_{its}$ | is the fraction of annual work hours spent on operation of AD plant, |
| | $w_{it}$ | is the average annual wage for the agricultural sector, |
| | $p_{it}^{ind}$ | is the average electricity price for the industry sector, |
| | $E_{its}$ | is the amount of energy generated from the AD process, |
| | $F_{its}$ | is the amount of pure fertilizer nutrients (N-$P_2O_5$-$K_2O$) generated from the AD process, and |
| | $p_f$ | is the unit price of fertilizer nutrients. |

The efficiency of energy production of an anaerobic digester loaded with animal manure only is rather poor, because much of the organic matter in manure degrades slowly and with a relatively low content of micronutrients needed by the anaerobic bacteria to form methane. The performance can be considerably enhanced if manure is co-digested with other organic material rich in micronutrients (Sommer et al., 2013). In farm AD plants, which primarily digest manure, it is therefore typical to add at least 20% other organic substrate to enhance the energy-generating performance of the process (ETH, 2008). In GAINS it is assumed that the feedstock contains 20% organic substrate and 80% manure and that both these substrates have a water content of 85%, respectively.

There is a wide range of organic waste materials available, which are suitable as additional substrate to manure in a farm AD plant (EC, 2001). These include straw and other crop residuals like maize stems, food residuals from restaurants and municipalities, food industry waste (e.g., residuals from slaughterers or waste from beverage or fat production), and sewage sludge from wastewater treatment. If there is a risk that the organic substrate contains pathogens, it will need to be pasteurized through the use of a thermophilic process or the digestate resulting from the digestion process cannot later be spread as fertilizer on fields (Sommer et al., 2013). In consistency with the European animal by-products regulation (EC 1069/2009), the digestate needs to be subject to minimum pasteurization equivalent to one hour at 70°C. In equivalence of this, some biogas plants perform pasteurization at a lower temperature, e.g., 53°C, but with the longer retention time of 10 hours (IEA Bioenergy Task 37, 2014b) to better preserve some of the bacterial content of the feedstock.

Because of the wide variety of sources for organic substrate and fluctuations in its availability over time and space, it is hard to make general assumptions about the unit price of organic substrate. It may vary from zero cost for organic waste, which suppliers would otherwise have had to pay to get rid of in an appropriate way, to 150 Euros per ton if feed crops (e.g., maize) are used. In the GAINS mitigation scenarios produced here, we assume the price of organic substrate is 100 Euros/ton.

The amount of energy generated per ton of feedstock from a biogas digester can fluctuate widely depending on e.g., the composition of the feedstock and different operating conditions. From a survey of published case-studies of farm-scale AD plants, we find three European plants currently in operation for which information on both the amount of feedstock loaded and the amount of energy generated, are clearly stated. These are the McDonnell farms in Limerick, Ireland (SEAI, 2014) co-digesting 7600 ton cattle manure per year with 2800 ton food waste and 360 ton glycerine, the Stowell farms in Wiltshire, UK (EnviTec, 2012), co-digesting 4000 ton cow manure with 8000 ton maize silage, and Alviksgården, Luleå, Sweden (Svenska Gasföreningen, 2014) co-digesting 16000 ton pig slurry with 1800 ton slaughter waste. All three farms use or sell the digestate as organic fertilizer. Figure 2 plots the reported net energy generated against the amount of wet substrate loaded for the three plants as well as for a "typical" farm used in model simulations to assess the potential for farm AD in Switzerland (ETH, 2008). For the latter plants, a feedstock of 80% manure and 20% co-substrate is assumed. The energy comes in the form of electricity or heat and typically about 50% is converted to electricity and 50% is utilized as heat.

The three plants currently in operation and illustrated in Figure 2 report net energy generation of 381, 637 and 539 kWh/ton substrate, respectively, while the Swiss study assumes 245 kWh/ ton substrate. In GAINS it is assumed that it is possible to generate on average 380 kWh/ton substrate loaded.

Based on the split between electricity and heat generated as reported by the surveyed farm AD plants, it is assumed in GAINS that half of the 380 kWh/ton substrate generated is converted to electricity, which is sold to local industry at the country-specific industry sector price of electricity and heat taken from PRIMES (E3modelling, 2020). 40 percent is used on farms as heat, and 10 percent is heat used up by the pasteurization process or lost without any economic value.

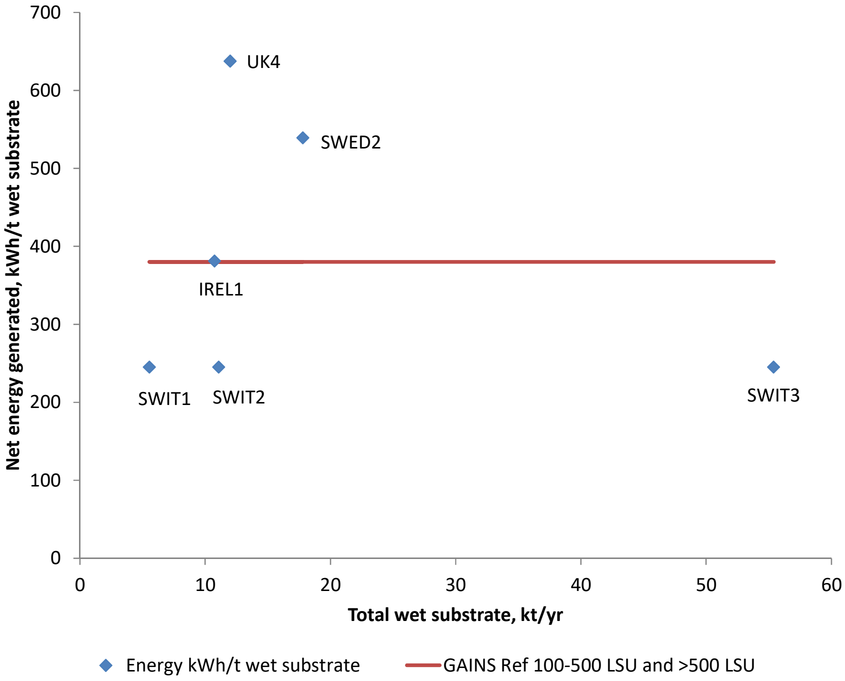

Figure 2: Net energy generation per ton wet substrate loaded for three farm AD installations currently in operation (IREL1, SWED2 and UK4) and for three model simulations (SWIT1, SWIT2, SWIT3) of "typical" Swiss farm AD installations.

Apart from the carbon content of the manure, which has partly been converted into biogas, the nutrients present in the manure feedstock will remain in the digestate after the anaerobic digestion is completed. During digestion, the organic nutrients present in the manure are transformed to inorganic compounds, e.g., organic nitrogen is converted to ammonia. The inorganic compounds can be more readily taken up by the plants than the organic nutrients present in undigested manure (Sommer et al., 2013). The digestate is therefore well suited as organic fertilizer. Table 13 presents typical nutrient contents of animal manure as well as for a few organic waste sources considered suitable as additive organic substrate in co-digestion with manure. The nutrients considered to be essential as fertilizers are nitrogen (N), phosphorous ($P_2O_5$), and potassium ($K_2O$). As shown in Table 13, the range for the nutrient contents is wide. As a conservative assumption, we assume that cattle and pig slurry contains 0.6 percent of nutrients N- $P_2O_5$- $K_2O$ in the proportions 50-17-33 and that added organic waste contains 2.7 percent of nutrients N- $P_2O_5$- $K_2O$ in the proportions 55-30-15. The assumptions give us a basis for estimating the amount of nutrients present in the digestate and therefore available for use as organic fertilizer. The value of pure nutrients in the proportions above is set to 1000 Euros per ton N- $P_2O_5$- $K_2O$, which would correspond to a price of 500 Euros/ton for an organic fertilizer containing 50 percent pure nutrients.

Table 13: Typical nutrient content in animal waste and some organic waste sources in Europe. % of substrate mass weight.

| Substrate source: | | unit | Min | Max | Mean |
|---|---|---|---|---|---|
| Fresh cattle manure | DS content | % of manure | 20 | 50 | n.a. |
| | N | % of manure | 0.55 | 1.21 | n.a. |
| | P2O5 | % of manure | 0.1 | 0.8 | n.a. |
| | K2O | % of manure | 0.25 | 1.2 | n.a. |
| Cattle slurry | Dry solids (DS) | % of slurry | 1 | 18 | n.a. |
| | N | % of slurry | 0.26 | 2.02 | n.a. |
| | P2O5 | % of slurry | 0.1 | 1.2 | n.a. |
| | K2O | % of slurry | 0.2 | 1.5 | n.a. |
| Fresh pig manure | Dry solids (DS) | % of manure | n.a. | n.a. | 25 |
| | N | % of manure | 0.57 | 0.95 | n.a. |
| | P2O5 | % of manure | 0.1 | 0.76 | n.a. |
| | K2O | % of manure | n.a. | n.a. | 0.4 |
| Pig slurry | Dry solids (DS) | % of slurry | 1 | 18 | n.a. |
| | N | % of slurry | 0.41 | 1.96 | n.a. |
| | P2O5 | % of slurry | 0.1 | 1.2 | n.a. |
| | K2O | % of slurry | 0.2 | 0.9 | n.a. |
| Food & drink industry sludge | Dry solids (DS) | % of sludge | 1.3 | 91 | 12 |
| | N | % of DS | 0.73 | 16 | 4 |
| | P2O5 | % of DS | 0.1 | 16 | 2.4 |
| | K2O | % of DS | 0.1 | 16 | 1.4 |
| Slaughter waste - guts content | Dry solids (DS) | % of waste | 2.4 | 21 | n.a. |
| | N | % of DS | 0.2 | 27.2 | n.a. |
| | P2O5 | % of DS | 0 | 3.4 | n.a. |
| | K2O | % of DS | 0 | 1 | n.a. |
| Slaughter waste sludge | Dry solids (DS) | % of sludge | 8 | 25 | 16 |
| | N | % of DS | 2 | 80 | 22.1 |
| | P2O5 | % of DS | 1.7 | 36 | 11 |
| | K2O | % of DS | 0.8 | 4.4 | 1.3 |
| GAINS assumption: cattle & pig slurry | Dry solids (DS) | % of slurry | - | - | 15 |
| | N | % of slurry | - | - | 0.3 |
| | P2O5 | % of slurry | - | - | 0.1 |
| | K2O | % of slurry | - | - | 0.2 |
| GAINS assumption: Organic waste substrate | Dry solids (DS) | % of waste | - | - | 15 |
| | N | % of waste | - | - | 1.5 |
| | P2O5 | % of waste | - | - | 0.8 |
| | K2O | % of waste | - | - | 0.4 |

Source: EC, 2001. Chapter 4: PROPERTIES OF WASTES RELEVANT TO AGRICULTURAL BENEFIT AND ENVIRONMENTAL IMPACT, European Commission-Directorate-General for Environment, Brussels. http://ec.europa.eu/environment/waste/studies/compost/landspreading_4-6.pdf

Figure 3 shows the fixed initial investment cost per ton of wet substrate loaded per year against the wet substrate loading capacity for six anaerobic digestion plants currently in operation and for which information was available on both investment cost and substrate load. In addition to the Swedish and Irish plants presented in Figure 2, these include the Mountstephen farm in Devon UK co-digesting 2373 ton cow slurry and chicken litter with 1095 ton maize and the Langage farm in Devon UK co-digesting 3000 ton cow slurry with 13000 ton food industry waste. Also illustrated in Figure 3 are the assumptions used for model simulations of the potential for farm AD in Switzerland (ETH, 2008). To illustrate the importance of scale for the investment cost of anaerobic digestion plants, Figure 3 also present the investment cost per ton substrate loaded for two large Danish plants co-digesting manure from several farms with organic waste from other sources. For the farm AD plants illustrated, investment costs vary between 75 and 225 Euro/t wet substrate loaded annually over a load range of 3.5 to 55 kt wet substrate per year, while the corresponding costs for the two Danish co-digestion plants are 21 and 37 Euro/t wet substrate for annual loads of 281 and 164 kt wet substrate, respectively. In GAINS the adopted investment cost for AD plants on farms of the size 100-500 LSU is 200 Euro/t wet substrate loaded, while for farms of the size larger than 500 LSU it is 100 Euros/t wet substrate loaded. Because the amount of wet substrate per animal is derived from country-specific volatile excretion rates, the derived investment cost per head is country-specific (see Figure 4). The investment cost is the product of the investment cost per ton wet substrate for farm size $s$ and the country-specific amount of wet substrate loaded per animal head, i.e.,

$$I_{its} = i_s \times m_i \times 1.25. \qquad (26)$$

Note that the total amount of wet substrate loaded per head is 1.25 times the manure generated per head as the total substrate contains 80 percent manure and 20 percent other organic substrate.

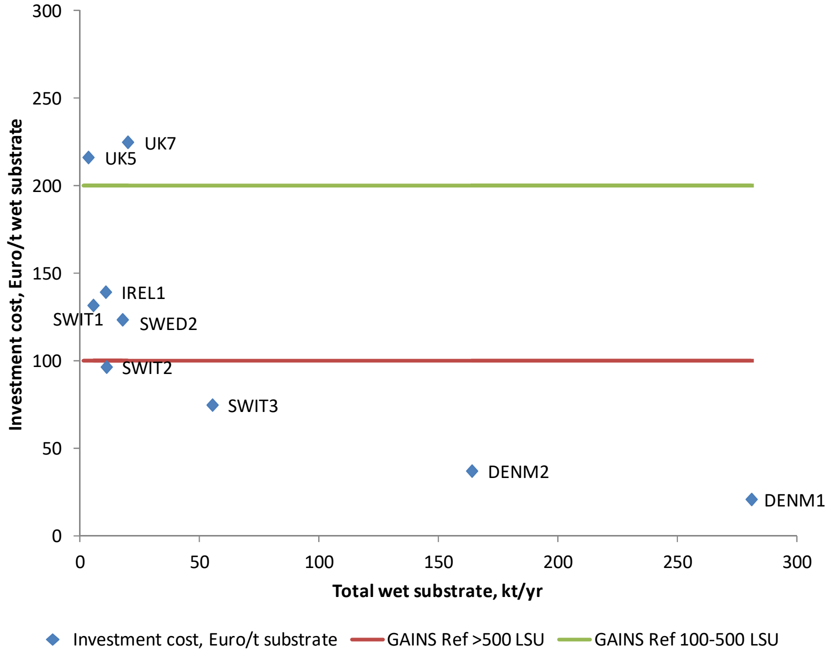

Figure 3: Fixed initial investment cost per ton of wet substrate loaded annually for four currently operating farm AD plants (IREL1, SWED2, UK5, UK7), three model simulation plants (SWIT1, SWIT2, SWIT3) and two large co-digestion plants (DENM1, DENM2) co-digesting manure and industrial organic waste.

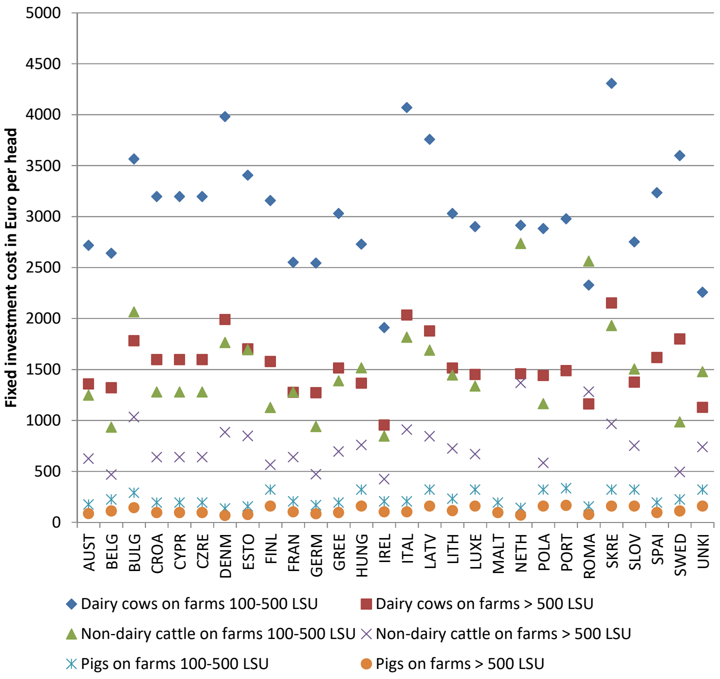

Figure 4: Derived fixed investment cost per animal head.

There are few quantifications of labour costs for the operation of farm AD plants available. There are examples of farmers mentioning high maintenance requirements in particular during the start-up of an AD plant (e.g., Farming Futures, 2010a; Svenska gasföreningen, 2014). When operation and maintenance (O&M) costs are mentioned, these usually reflect the sum of labour costs and the cost of additional organic feedstock, which makes it hard to quantify labour costs separately. Walla (2004) describes two farm AD plants of sizes 100 and 170 kW digesting manure from 140 and 170 LSU dairy cows, respectively. The plants co-digest manure (amount unknown) with 800 and 2300 ton energy crop, respectively, and estimate O&M costs to 34000 and 88000 Euros per year, respectively. USEPA (2014b, 2014c) report O&M costs for two farm AD plants, which only digest manure without adding co-substrate to the process. O&M costs are therefore likely to mainly reflect labour costs. One of the plants has the power of 180 kW and digests manure from 8000 pigs and report annual O&M costs corresponding to 19250 Euros or 2.4 Euros per pig. The other farm is a dairy farm with an AD plant of 215 kW and digesting only manure from 900 dairy cows. The latter reports O&M costs corresponding to 38500 Euros per year, i.e., 43 Euros per dairy cow or 3 Euros/ton manure for cows generating 14 tons/head/year. 3 Euros/ton manure convert to about 0.2 work hours per ton manure if assuming an annual wage of 25000 Euros and 1800 working hours per year. We used the assumption of 0.2 work hours per ton substrate digested in GAINS. The cost of labour per ton substrate is estimated by multiplying 0.2 work hours with the average hourly wage rate for the agricultural sector in each member state as reported by ILO (2010) and projected to grow with the growth of value added in the agricultural sector as provided by the PRIMES model (E3modelling, 2020).

Measurements of leakage of fugitive methane emissions from farm AD plants suggest leakage rates between 3 and 4 percent of gas produced for well managed installations (Flesch et al., 2011; Groth et al., 2015). Considering that a biodigester enhances the conversion of coal into methane, a leakage rate of a few percent of the total generation of methane should be converted to a higher fraction of the methane generated had the manure not been treated in a digester but been subject to alternative manure management practices. Hristov et al. (2013a) mention removal efficiencies for farm AD plants of about 60 to 70% of methane emissions had the manure been treated in a conventional way. The assumed removal efficiency is in GAINS set to 60 percent of methane released from alternative manure management practices for farms of size 100-500 LSU. The assumed removal efficiency for plants installed on farms with more than 500 LSU is 80 percent as the very large farms are expected to be more efficient in controlling the methane slip throughout the process. Pellerin et al., (2013) estimate a removal efficiency of 88% for farm AD installations treating non-dairy cattle and pig manure in France. It is however not clear from Pellerin et al., (2013) if the 88% removal efficiency refers to the fraction removed of the methane generated in the digester or the methane removed in relation to a manure management system without anaerobic digestion.

Table 14 presents a summary of the assumptions that enter unit cost estimations of farm AD in GAINS.

Table 14: Summary of assumptions entering unit cost estimation for farm anaerobic digestion (AD).

| | | Reference scenario assumptions[a] | |
|---|---|---|---|
| Factors entering unit cost estimation | Unit | Farms 100-500 LSU (L) | Farms > 500 LSU (XL) |
| Volatile solids (VS) | % of Total solids (TS) in manure | 80% | 80% |
| Total solids | % of manure | 15% | 15% |
| Water content of manure | % of manure | 85% | 85% |
| Added organic substrate | % of manure | 25% | 25% |
| Water content of added organic substrate | % of organic substrate | 85% | 85% |
| Total wet substrate (manure+organic substrate) | % of manure | 125% | 125% |
| Investment cost -fixed initial investment | Euro per ton wet substrate loaded annually | 200 | 100 |
| Lifetime of investment | years | 20 | 20 |
| Organic substrate price | Euro per ton organic substrate | 100 | 100 |
| Labour hours | work hours per ton wet substrate per year | 0.2 | 0.2 |
| Average wage agricultural sector | Euro per year | Country-specific (ILO, 2010) projected by value added in agricultural sector (PRIMES, 2015) | |
| Energy generated | kWh per ton wet substrate per year | 380 | 380 |
| whereof electricity supplied at industry electrcity price | kWh per ton wet substrate per year | 190 | 190 |
| whereof heat used on farm | kWh per ton wet substrate per year | 152 | 152 |
| whereof heat used in pasteurization process or lost | kWh per ton wet substrate per year | 38 | 38 |
| Electricity price | Euro per kWh | Industry sector by country (PRIMES, 2015) | |
| Heat price | Euro per kWh | 50% of industry electricity price | |
| Organic fertilizer generated from slurry | Pure nutrients ($N\text{-}P_2O_5\text{-}K_2O$) as % of slurry | 0.6% | 0.6% |
| Organic fertilizer generated from organic waste | Pure nutrients ($N\text{-}P_2O_5\text{-}K_2O$) as % of waste | 2.7% | 2.7% |
| Organic fertilizer price | Euro per ton pure nutrients ($N\text{-}P_2O_5\text{-}K_2O$) | 1000 | 1000 |
| Emission removal efficiency | Reduction from no control | 60% | 80% |

[a] Assumptions presented are without effects of technological development (see Section 2.5.1.)

To identify the extent of the current adoption of farm AD in different member states, we first derive the maximum technically feasible output of energy from farm AD plants by animal category and farm-size in the respective member states on the basis of the assumptions summarized in Table 14. We then derive the total energy output produced in 2015 from manure-based anaerobic digesters in consistency with the farm-based biogas production as estimated by the PRIMES model for the same year. The conversion efficiency to electricity is assumed 0.375 and to heat 0.7 (and 50% of the energy output generated is in the form of electricity and 50% in the form of heat as specified in Table 14). The source of information for farm-based biogas production in historical years in the PRIMES model is Eur'Observer (2020). By relating the total energy output from manure-based anaerobic digestion in 2015 to the maximum technically feasible output of energy from manure-based systems, we obtain the percentage of the maximum potential currently exhausted. We exhaust the current potential using the same assumed adoption order for all member states, i.e., starting adoption on pig farms greater than 500 LSU. Once the potential in this category is exhausted, we move on to dairy farms greater than 500 LSU, then non-dairy cattle farms greater than 500 LSU, then pig farms 100 to 500 LSU, then dairy farms 100 to 500 LSU, and finally non-dairy cattle farms 100 to 500 LSU. The control strategy for 2015 was developed in consistency with actual application (Eur'Observer, 2020). The development in implied emission factors for manure management reported by member states to the UNFCCC (2019) for the years 2005 to 2018 was used as an indicator of the development in the uptake of farm AD technology between 2005 and 2015. E.g., if the reported implied emission factor for pigs in 2005 and 2010 is the same as in 2015, then the control in these two years is assumed to be at the same level as in 2015. For future years, the control strategy was developed so as to be consistent with the growth in farm-based biogas production projected by the PRIMES model (E3modelling, 2020).

### 3.3.2 Rice cultivation

$CH_4$ emissions from rice cultivation result from anaerobic decomposition of organic material in flooded rice fields. Emissions depend among other factors on the season, soil characteristics, soil texture, use of organic matter and fertilizer, climate, as well as on agricultural practices (IPCC, 2006, Vol.4, p. 5.45). The emission calculation methodology used in GAINS follows the IPCC guidelines (2006, p. 5.49) and adopts IPCC default emission factors for given water management regimes. The IPCC method is based on the annual harvested area with scaling factors for different water regimes. In GAINS, these are translated into three cultivation activities:

- *Continuously flooded cultivation area:* fields have standing water throughout the growing season and only drying out for harvest.
- *Intermittently flooded cultivation area:* fields have at least one aeration period of more than three days during the growing season. Compared with continuously flooded rice fields, IPCC suggests that intermittently flooded rice fields emit 27 to 78 percent of continuously flooded fields, where the range depends on if the fields are rainfed or irrigated. GAINS uses the assumption of 50 percent emissions per hectare from intermittently flooded compared with continuously flooded fields.
- *Upland rice cultivation area:* fields are never flooded for a significant period of time and are not assumed to emit $CH_4$.

$CH_4$ emissions from rice cultivation in country $i$ in year $t$ are calculated as follows:

$$E_{it} = \sum_{sm} A_{it} * ef_{i;flood}^{IPCC} * h_i * \beta_s * V_{is} * (1 - remeff_{sm}) * Appl_{itsm} \ , \qquad (27)$$

where $A_{it}$ is the rice cultivation area in country $i$ in year $t$,

$ef_{i;flood}^{IPCC}$ is the IPCC default emission factor for $CH_4$ emissions from flooded rice fields (1.3 kg $CH_4$ $ha^{-1}$ $day^{-1}$),

$h_i$ is the duration of the growing season expressed in days per year (=185 days per year),

$\beta_s$ is an emission scaling factor for water regime $s$ (=1 for continuously flooded, =0.5 for intermittently flooded, and =0 for upland rice).

$V_{is}$ is the fraction of rice cultivated land under water regime $s$,

$remeff_{sm}$ is the removal efficiency of technology m when applied to water regime s, and

$Appl_{itsm}$ is the application rate of technology $m$ when applied to water regime $s$.

Methane emissions from rice cultivation in Europe are in GAINS accounted for in Bulgaria, France, Greece, Hungary, Italy, Portugal, Romania, and Spain. Activity data for rice cultivation is measured in million hectares of land and is taken from UNFCCC-CRF (2019) with projections based on OECD-FAO (2012) and with information on country-specific application of water regimes taken from UNFCCC-CRF (2015).

$CH_4$ mitigation options implemented in GAINS to control emissions from rice cultivation include employment of improved water management regimes, use of alternative rice hybrids increasing yields while suppressing methane generation e.g., through shorter stems, and use of soil amendments e.g., biochar or sulphate-containing amendments.

There are several ways to reduce $CH_4$ emissions through improved water management; single mid-season drawdown, alternative wetting and drying, aerobic rice production and dry direct seeding (WRI, 2014). A common feature of all water management options is that they reduce $CH_4$ emissions through decreasing the time that fields are flooded. Differences in local conditions e.g., climatic conditions, traditional farming customs and access to herbicides, water regulation mechanisms or fertilizers, will affect the impact of different water management regimes on yield, labour requirements and methane emissions (WRI, 2014). The choice of preferred water management regime is closely linked to these local conditions. Due to lack of information, we are not able to make a full-fledged assessment of the effectiveness of individual water management regimes in different regions of the world, but will have to resort to making broad assumptions about the effectiveness of water management regimes in general and their associated costs. According to a literature survey by WRI (2014), implementing improved water management regimes on continuously flooded fields have shown to achieve $CH_4$ emission reductions between 30-90%, with the higher relative reductions found for well-managed fields in the US. As a general assumption in GAINS across all flooded rice fields, an average abatement potential of 20% is assumed to be achievable in the next ten years, extending to 40% on an annual basis in 2050. If improved water management is combined with other options e.g., low-$CH_4$ hybrids or different soil amendments (see below for details), the average global abatement potential assumed in GAINS for continuously flooded fields extends to 50%. This estimate takes into account that some areas may be difficult to subject to improved water management due to heavy rainfall during the wet season or due to unreliable water supply systems or fields that are not well levelled (WRI, 2014). These assumptions are somewhat conservative in comparison to Beach et al. (2015) who estimate an overall abatement potential for global rice cultivation in 2030 at 26.5% below baseline and Harmsen et al. (2019) who estimate 61% below baseline in 2050 for the same source.

A cost estimate of improved water management through drying out of continuously flooded rice fields will have to consider associated operation costs, including cost-savings from reduced water use and higher labour costs due to increased weed growth. In particular in poorer regions where farmers lack access to herbicides, longer periods of dry fields increase weed growth (WRI, 2014; Barrett et al. 2004; Ferrero and Nguyen 2004). According to estimates by Barrett et al. (2004), weed growth increases labour costs by an estimated 20 percent, which is equivalent to about 60 additional work hours annually per hectare in developing countries (Heytens, 1991) and 12 additional work hours annually per hectare in developed countries, where herbicides are used for controlling weed (Shibayama, 2001). Dry direct seeding of rice seedlings have shown to be very effective (45-90% reductions in emissions) for reducing CH4 emissions in the US compared with transplanting seedlings into flooded fields (WRI, 2014; Linquist et al., 2015). The abatement effect is attributed to the one month shorter period of flooding as seedlings grow in dried out fields. The option also contributed to reduced labour input and costs, however, this result appears to be conditional on unrestricted access to herbicides and well managed water tables and may therefore be difficult to replicate in many developing countries. According to IRRI (2007), intermittent aeration of continuously flooded rice fields may reduce water use by 16 to 24 percent. Assuming that continuously flooded rice fields need 1000 mm water input per year (Bouman, 2001) and the average cost of irrigated water in Europe is 0.04 Euro per $m^3$ (FAO, 2004), then saving 22 percent of water corresponds to a cost-saving of about 70 Euro per ha.

Certain rice hybrids may affect $CH_4$ emissions. By careful selection of low-$CH_4$ producing hybrids, emissions can be ten percent lower (ADB 1998). ADB (1998) estimates that Chinese rice yields may increase by as much as 10 to 20 percent from switching to low-$CH_4$ hybrids. In other parts of the world, where high yield rice hybrids are already in extensive use, potentials for additional yield increases are likely lower. In GAINS, the assumption is that the potential reduction in $CH_4$ emissions from switching to alternative rice hybrids is 10 percent with a 3 percent increase in crop yield, when applied as the sole option. When applied in combination with other options, like improved water management of continuously flooded fields, the removal efficiency of this option is set to 5 percent.

Application of sulphate-containing substrates to rice fields reduces $CH_4$ emissions because $CH_4$ producing bacteria compete for the same substrate as the sulphate reducing bacteria (van der Gon et al. 2001). Likewise, application of biochar to soils in rice fields improves soil fertility while contributing to reduced $CH_4$ emissions because carbon is added in a stabilized form, which inhibits the abundance and activity of methanogens (Han et al., 2016). The costs associated with these options are the costs of acquiring the sulphate-containing substrates or biochar and spreading them on the fields. In GAINS, a conservative assumption is that application of these types of $CH_4$ inhibitors can remove on average 20 percent of $CH_4$ emissions when applied as a stand-alone option and 5 percent when applied in combination with other options like improved water management.

### 3.3.3 Open burning of agricultural waste

Open burning of crop residuals on fields and other agricultural waste is forbidden in the European Union and regulated under the Council Regulation EC/1306/2013. Many EU member states do not report emissions from this activity. However, remote sensing data (e.g., from MODIS) consistently shows occurrence of hundreds of fires every year. Interpretation of the remote sensing data is associated with several uncertainties and depending on the instrument used, models and data (e.g., land use cover) applied, the results vary significantly. There are annual variability and multi-year trends. We attempt to represent the latter, although, it is largely indicative since the number of fires is not the only indicator that matters. Data is retrieved from the MODIS instrument as used in a GFED product (van der Werf et al., 2010; 2017) and reflect trends and changes in estimated biomass burned. In addition to the remote sensing information, we also compare estimates to bottom up estimates from Bond et al (2004), EDGAR FastTrack2000 data and information received through direct communication with national experts, who often have access to local data, e.g., on local management practices.

The emission factor used is the IPCC default emission factor for open burning of waste (IPCC, 2006, Vol.5, p.5.20) which corresponds to 6.5 kt $CH_4$ per Mt waste burned. Any further potential to reduce methane emissions from this source comes from stricter enforcement of the existing ban. The cost of mitigating methane through this option is set to zero, assuming stricter enforcement does not incur additional costs.

## 3.4 Waste sector

### 3.4.1 Solid waste

$CH_4$ from municipal and industrial solid waste is generated when biodegradable matter is digested under anaerobic conditions in landfills or during temporary storage of waste aimed for different types of treatment. $CH_4$ may also be released during loading or emptying of the reactor when organic waste is treated in anaerobic digesters to produce biogas or during treatment of organic waste in composts. The activity data used in GAINS is the total amount of waste generated before diversion to different types of treatment like recycling, energy recovery or landfill. Amounts of waste generated are first split by municipal or industrial solid waste and then by waste composition for municipal solid waste and by manufacturing industry sub-sector for industrial solid waste.

$CH_4$ from waste deposited on landfills is formed and released with a time delay of up to several decades. IPCC (2006, Vol. 5, Ch. 3) recommends the use of a First-order-decay model taking up to fifty years disposal into account. The GAINS model structure does not allow for implementation of a full First-order-decay model. Instead, a simplified structure is used, where the delay between waste disposal and $CH_4$ release is accounted for as a lag in the activity data of 10 years for fast degrading organic waste like food and garden waste and 20 years for more slowly degrading waste like paper, wood and textile waste. The lags correspond to approximate average half-life values for the respective waste types (IPCC, 2006, Vol.5, Tables 3.3 and 3.4).

$CH_4$ emissions from municipal (or industrial) solid waste in country *i* in year *t* are estimated as the sum of emissions from a certain waste type *s* (or industry sector) summed over emissions from waste diverted to waste treatment option *m* :

$$E_{it} = \sum_{s} \sum_{m} A_{i;(t-y_s);s} * ef_s * (1 - remeff_{sm}) * Appl_{itsm} \tag{28}$$

where $A_{i;(t-y_s);s}$ is amount of waste type (or industry sector) *s* generated in year $t\text{-}y_s$, where $y_s$ is the average lag in $CH_4$ release assumed for waste type (or industry sector) *s*,

$remeff_{sm}$ is the removal efficiency of waste treatment option *m*, and

$Appl_{itsm}$ is the application of waste treatment option *m* to waste type (or industry sector) *s* in country *i* in year *t,* and

$ef_s$ is the IPCC default emission factor for waste type (or industry sector) *s* deposited in a landfill without recovery of landfill gas.

From IPCC (1997, Vol.3, Ch.6, Equation 1) the following expression for the waste type specific emission factor $ef_s$ is obtained:

$$ef_s = DOCm_s * DOCf * \mathrm{MCF} * F * \frac{16}{12} * (1 - OX) \tag{29}$$

Where

| | |
|---|---|
| $DOCm_s$ | is the fraction of Decomposable Organic Carbon (DOC) in waste type/sector *s*, |
| $DOCf$ | is the fraction of DOC that can decompose (default used is 0.5), |
| $MCF_i$ | is the Methane Correction Factor correcting for aerobic decomposition and vary with the management standard of the landfills, |
| $F$ | is the fraction of $CH_4$ in generated landfill gas (default used is 0.5), |
| *16/12* | is the molecular weight ratio $CH_4/C$, |
| $OX$ | is the oxidation factor correcting for increased oxidation from covering of landfills (default used is 0.1). |

IPCC (2006, Vol.5, Tables 2.4 and 2.5) default factors are used for the content of decomposable organic carbon (*DOCm*) in different types of biodegradable waste as well as for the Methane Correction Factor (MCF) applied to different management standards of landfills.

It can be expected that waste generation is positively related to income (World Bank, 2012) and that relative changes in income have a relatively larger effect on waste generation in high income groups than in low income groups. We used country-level data to estimate waste generation elasticities for different average income intervals using annual data on income and historical MSW generation amounts and composition taken from EUROSTAT (2019) for 31 countries (EU-27, Iceland, Norway, United Kingdom and Switzerland) and spanning the time period 1995-2017. To estimate elasticities, the relationship between the relative impact of GDP per capita and urbanization rate (UNstat, 2019) on the per capita MSW generation was formulated as a log-linear function:

$$log(MSWcap_{it}) = \alpha + \beta_1 * log(GDPcap_{it}) + \beta_2 * log(urbrate_{it}) + \varepsilon_{it}, \qquad (30)$$

where

| | |
|---|---|
| $MSWcap$ | is kg MSW per capita per year, |
| $GDPcap$ | is the average annual Gross Domestic Product in Euro per capita, |
| $urbrate$ | is the fraction of the total population living in urban areas, |
| $\varepsilon_{it}=u_i+v_{it}$ | is an error term which is separated into an individual effects term and a residual omitted variables term, and |
| $\varepsilon_{it}\sim IID(0, \sigma_\varepsilon^2)$ | is an error term which is independent and identically distributed, and |
| $\beta_1$ and $\beta_2$ | are estimated elasticities for income and urbanization rate, respectively. |

The MSW waste composition categories considered are Food and garden, Paper, Textile, Wood, Plastics, Glass, Metal, and Other.

Elasticity estimates were generated separately for three different levels of average GDP per capita; < 20000 Euro, 20000 to 40000 Euro, and above 40000 Euro. The results of the elasticity estimates are presented in Table 1. Projections of future generation of total MSW, as well as generation of waste in the period prior to 1995, are estimated applying the elasticities presented in Table 15. When projecting the development in the composition of the MSW, priority was first given to drive generation of food waste using an elasticity estimated only for MSW food waste generation. Thereafter, non-food waste categories were attributed proportionally to the 2010 attribution until constrained by the estimated total MSW generated. This approach take consideration of the inferiority nature of food in relation to other goods, i.e., as income increases we are likely to increase our consumption of food relatively less than we increase consumption of non-food goods.

Table 15: Elasticity estimates for drivers of Municipal Solid Waste (MSW) generation.

| Dependent Variable | Unit | Income Group | Number of Observations | Explanatory variable | OLS | Fixed Effect | Random Effect | LM test | Haussman test |
|---|---|---|---|---|---|---|---|---|---|
| Municipal solid waste -total | Mt per capita | < 20000 | 203 | Constant | -0.001 (-1.00) | _n.a_ | -0.01 (-0.80) | 1.41 | 9.9 |
| | | | | GDP per capita | 0.14 (3.20)** | _0.21(4.75)_ *** | 0.16*** | | |
| | | | | Urbanization rate | 0.14 (0.36) | _-0.76 (-1.69)~_ | 0.004 (3.66) | | |
| | | | | R-square | 0.07 | _0.06_ | n.a | | |
| | | >=20000 - <40000 | 221 | Constant | 0.0001 (0.20) | n.a | _-0.002 (-0.22)_ | 2.81 | 1.25 |
| | | | | GDP per capita | 0.33 (7.62)*** | 0.29(4.48) *** | _0.33*** (6.79)_ | | |
| | | | | Urbanization rate | | | | | |
| | | | | R-square | 0.2 | 0.009 | n.a | | |
| | | >=40000 | 235 | Constant | -0.002 (-0.52) | _n.a_ | -0.005 (-0.77) | 0.17 | 6.68 |
| | | | | GDP per capita | 0.58 (7.61)*** | _0.67(8.07)_ *** | 0.61 (7.74)*** | | |
| | | | | Urbanization rate | -0.85 (-3.91)*** | _-0.95(-4.24)_ *** | -0.89 (-4.05)*** | | |
| | | | | R-square | 0.11 | 0.18 | n.a | | |
| MSW Food waste | Mt per capita | All income groups | 156 | Constant | 4.05 (9.32)*** | n.a | 2.78 (4.29)*** | 40.54 | 9.78 |
| | | | | GDP per capita | 0.05 (1.33) | _0.428 (4.22)_ *** | 0.18 (2.85)** | | |
| | | | | R-square | 0.01 | 0.12 | n.a | | |
| Food, beverages and tobacco industry waste | Mt | All income groups | 70 | Constant | 0.77 (0.90) | n.a | _0.78 (0.72)_ | 18.7 | 0.01 |
| | | | | Value added | 0.83 (7.73)*** | 0.83 (3.27)** | _0.81 (5.82)_ *** | | |
| | | | | R-square | 0.47 | 0.71 | n.a | | |
| Pulp and paper industry waste | Mt | All income groups | 70 | Constant | -0.12 (0.20) | n.a | _-1.35 (1.72)_ | 54.4 | 0.24 |
| | | | | Value added | 0.85 (9.85)*** | 1.07 (7.84)*** | _1.03 (9.56)_ *** | | |
| | | | | R-square | 0.59 | 0.95 | n.a | | |
| Textile, leather and footwear industry waste | Mt | All income groups | 71 | Constant | -1.38 (1.99) | n.a | _-0.53 (0.48)_ | 67.8 | 6.4 |
| | | | | Value added | 0.88 (9.26)*** | -0.32 (0.72) | _0.74 (4.74)_ *** | | |
| | | | | R-square | 0.55 | 0.89 | n.a | | |
| Wood and wood products industry waste | Mt | All income groups | 71 | Constant | 3.12 (4.79)*** | _n.a_ | 3.64 (2.66)** | 173.3 | 0.78 |
| | | | | Value added | 0.47 (4.44)*** | _0.23 (1.39)_ | 0.33 (4.58)*** | | |
| | | | | R-square | 0.22 | _0.83_ | n.a | | |

Detailed information on historical uptake of waste treatment measures in 1990, 1995, 2000, 2005, 2010, 2015 and 2017 were collected from National reporting to the UNFCCC (version 2019) and EUROSTAT (2019). From the Common Reporting Format (CRF) tables, we take information on amounts of waste allocated to various types of landfills, composting and anaerobic digestion in biogas facilities as well as the Methane Correction Factor (MCF) for the different types of landfills used. From EUROSTAT (2019) we take data on overall recycling rates and specific recycling rates for paper waste. In consistency with the reported treatment information and emission factors derived for each waste category and treatment stream applying default IPCC (2006, 2019) methodology, the emission generation potential of waste is determined for every 5 years. The simplified version of the IPCC First-Order-Decay method explained above is then used to shift the timing of the release of emissions 10 or 20 years into the future, depending on if the waste is fast-decaying (food and garden waste) or medium-to-slow decaying (paper, wood, textile, other). Composting and anaerobic digestion are assumed to give rise to limited $CH_4$ emissions, while no $CH_4$ emissions are assumed from recycling of paper, wood or textile waste. Accounting for the decay time of waste in landfills means that to come up with emission estimates from 1990 onwards, we must also estimate the generation and treatment attribution of MSW for the period 1970 to 1990. For this purpose, an extrapolation was made which assumes that in year 1970 all MSW is landfilled except a small fraction that is openly burned (same fraction as reported for 1990). Between 1970 and 1990, a linear transition to the 1990 treatment attribution is assumed.

Projections for future waste treatment streams start from the treatment structure identified for year 2017 and assumes countries meet the targets of the amended EU Waste Directive from 2018, i.e., maximum 10% of total MSW being landfilled in 2035, with a five years grace period granted to Bulgaria, Croatia, Cyprus, Greece, Hungary, Latvia, Romania, Slovakia and Iceland. These countries must meet a target of maximum 25% MSW landfilled in 2035. A linear phase-in of the targets is assumed for the period 2020 to 2035. The targets are met by moving increasing amounts of MSW away from landfill disposal and towards other treatment options (recycling, anaerobic digestion, incineration with energy recovery) following the principles of the EU waste hierarchy.

Figure 5 illustrates resulting EU-27 and EU-27+UK estimates of $CH_4$ emissions from solid waste disposal using the consistent methodology of GAINS and as reported by countries in the CRF tables to the UNFCCC (version 2019). Figure 6 illustrates the same estimates by EU member state and the UK. Note that Figures 5 and 6 illustrate the sum of emissions from MSW and industrial solid waste as only the sum of these emissions are reported in the CRF tables. However, emissions from MSW dominate total solid waste $CH_4$ emissions.

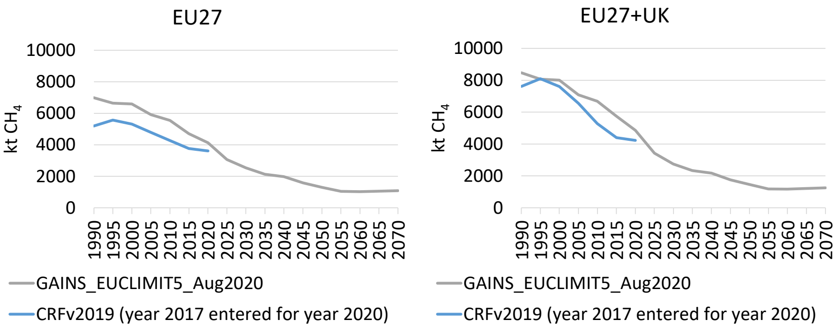

Figure 5: $CH_4$ emissions from solid waste disposal and treatment as estimated in the GAINS model August 2020 and reported in National submissions to the UNFCCC (2019) for EU-27 and EU27+UK.

As shown in Figure 5, GAINS estimates slightly higher emissions from solid waste sources for the EU-27 in the period 1990-2015, however, converging to the reported level in year 2020 (or 2017 as the last reported year is compared to the 2020 estimate in GAINS). Figure 6 shows that GAINS estimates match quite closely the reported emission trend of Austria, Bulgaria, Germany, Netherlands and Sweden. For a number of countries (Belgium, Bulgaria, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Netherlands, Poland, Portugal, Slovenia, Spain and United Kingdom), the reported emissions fall more rapidly in the period 2010-2017 than in GAINS (2010-2020). After 2020, the GAINS estimate often converges to the national estimate of five to ten years earlier. Hence, this may be an effect of these countries being more optimistic than GAINS about how fast landfilled waste decay and accordingly how fast an effect will be seen on emissions from the measures undertaken to fulfill the 1999 EU Landfill Directive targets (-65% of organic waste landfilled between 1995-2020).

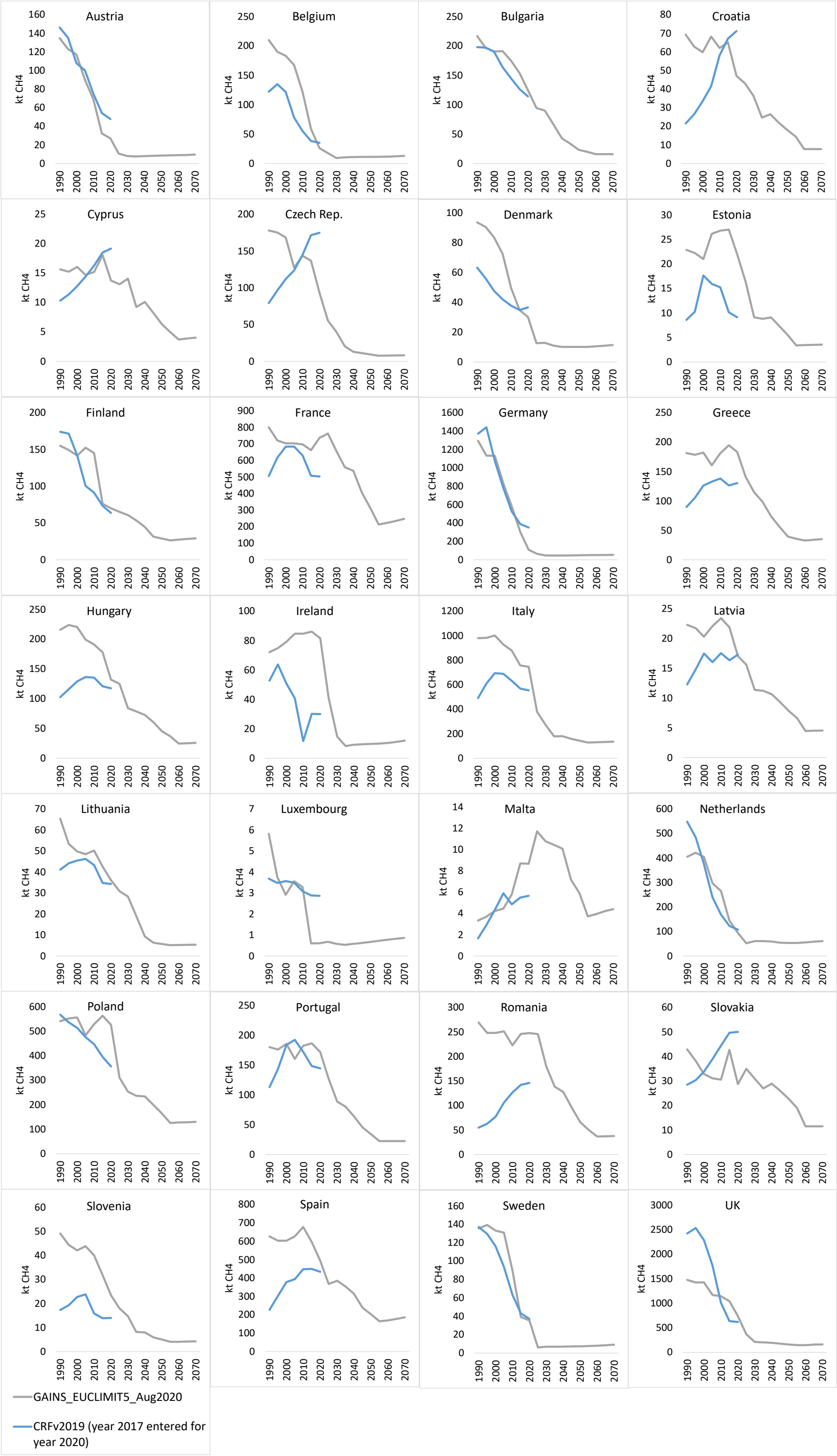

Figure 6: CH4 emissions from solid waste disposal and treatment as estimated in the GAINS model August 2020 and in comparison to emissions reported in the National submissions to the UNFCCC (2019), by country.

In GAINS, the maximum feasible reduction of $CH_4$ emissions in the waste sector is modelled as an “optimal” waste treatment path as defined by the EU waste hierarchy. Source separation of waste for recycling or energy recovery purposes is preferred to landfill disposal with gas recovery. In the optimal case, all biodegradable waste is source separated from the waste stream and none is disposed of to landfills. Some EU member states (Austria, Belgium, Denmark, Germany, Netherlands and Sweden) are close to this optimum and have already implemented a complete ban on landfilling of untreated biodegradable waste.

Costs for source separation and treatment of municipal solid waste are taken from various sources. The additional cost of collecting source separated waste compared to mixed waste is derived from Tanskanen (2000) and assumes 33 Euro/t waste for paper waste and 62 Euro/t waste for food and garden waste. The net cost-saving of reducing the amount of waste disposed of and treated in landfills is estimated at 20 Euro/t waste (AEAT, 1998). The cost for upgrading recovered biogas from 60 to 97 percent is taken from Persson (2003). Cost and potential for energy recovery from incineration of waste are taken from IPPC (2006).

Costs for treating household and industry food waste in anaerobic digesters for biogas recovery are taken from AEAT (1998, 2001). The digestion process is assumed to convert 60 percent of the original waste amount to biogas. Half of the rest product is assumed to be compost material which can be given away for free, a quarter is a liquor which needs further treatment at 12 Euro/t liquor, and the last quarter (i.e. 10 percent of the primary waste amount) is a residual which is landfilled at a cost of 20 Euro/t residual (AEAT, 2001).

The cost of recycling wood industry waste into chipboards is taken from Wilson (2003).

The net cost of household paper recycling is defined as the sum of the additional cost of collecting source separated waste compared to mixed waste and of converting paper waste to recycled pulp minus the cost-savings of depositing less waste to landfills and income from selling recycled pulp at a market value. The investment cost is assessed from the cost for a UK deinking plant producing pulp from 0.07 Mt paper waste per year and amounts to 623 Euro per ton paper waste recycled into pulp when expressed in the 2015 price level (AEAT, 1998). With a plant lifetime of 15 years and an interest rate of 10%, this means an annualized investment cost of 82 Euro per ton paper waste. From Tanskanen (2000), the additional cost of separate collection of paper waste compared with mixed waste collection is assessed at 38 Euro per ton waste. The time lost to households for separation and recycling of paper waste is estimated at 5 minutes per week for every two persons and when about 100 kg paper is recycled per person every year. The loss in time is valued using country-specific average annual wages (ILO, 2010 and projected with PRIMES (E3modelling, 2020). The cost-saving of avoiding landfill disposal is assessed at 25 Euro per ton paper waste (AEAT, 1998). Finally, the cost-saving of selling the recycled pulp on the world market is assessed at 388 Euro per ton recycled pulp (or 346 Euro per ton paper waste recycled), which corresponds to the difference between an average world market price of 676 Euro per ton virgin pulp (FAOSTAT, 2010) and an approximate marginal cost of processing recycled paper into pulp of 288 Euro per ton pulp derived from the cost data provided in AEAT (1998).

When summing up the different cost items described above, the net unit cost will often turn out negative, i.e., with these assumptions there appears to be a relatively large net profit from recycling household paper waste. Why would such opportunities for net profits exist and why have they not been exhausted already in the baseline scenario? We find two possible reasons for why there may still be unexhausted potential for extended paper recycling in the future despite seemingly negative costs. First, the whole recycling chain depends on the willingness of the households to separate the waste and supply it without compensation. Even if there would be profits to be made further down the recycling chain, the initial step of getting the households involved for free may still pose a limitation on the whole chain. Second, the market value of recycled pulp is uncertain and may be distorted upwards due to close integration between primary and secondary markets for pulp, i.e. between virgin and recycled pulp, as such integration has been documented several times (OECD, 2006; Ackerman and Gallagher, 2002). Being close substitutes, it is likely that virgin pulp producers have an interest in keeping the price of recycled pulp at a level where the costs for virgin pulp production are covered (often including costs for forest management).

### 3.4.2 Wastewater

Wastewater treatment plants serve to decompose compounds containing nitrogen and phosphor as well as carbon from the wastewater before discharge. Main gaseous products are $CO_2$ and molecular nitrogen, but during the process also $CH_4$ is released. $CH_4$ is formed whenever wastewater with high organic content is handled under anaerobic conditions.

In the GAINS model, wastewater from households and industry are accounted for separately. The activity data used for estimation of emissions from domestic wastewater is number of people connected to centralized or decentralized collection of wastewater. This basically refers to wastewater from urban and rural population, except for most industrialized countries where wastewater collection services often include some rural areas as well. Country-specific data on fractions of wastewater collected centrally is taken from UNFCCC (2019), EUROSTAT (2019) and OECD (2015).

Uncontrolled emissions are defined as emissions when wastewater is emitted directly to a water body without prior collection and treatment. As anaerobic conditions are formed when large quantities of wastewater are collected and stored, $CH_4$ formation in the uncontrolled case are likely to be limited and to increase for any form of organized wastewater collection. Collection is however a prerequisite for treatment, which is important for combating water pollution from excessive nitrogen and phosphor. Uncontrolled $CH_4$ emission factors are derived following the IPCC guidelines (2006, Vol.5, Equations 6.1 to 6.3):

$$E_{it} = \sum_s A_{it} * h_s * ef_i * (1 - remeff_m) * Appl_{itm} \tag{31}$$

where $A_{it}$ is total population in country *i* and year *t*,

$h_s$ is fraction of total population connected to treatment system *s* (centralized or decentralized treatment)

$remeff_m$ is the removal efficiency of technology $m$,

$Appl_{itm}$ is the application of technology $m$ in country *i* and year *t*, and

The methanogenic process in the treatment of wastewater is sensitive to daily/seasonal temperature variations as temperature affects the microbiological community and the degradation rate of organic matter (Dhaked, Singh and Singh, 2010). Temperature is therefore a relevant factor for the formation of methane during treatment of domestic wastewater, where the lower the temperature, the lower the methane formation (Luostarinen et al. 2007). In GAINS a country-specific temperature correction factor was included when deriving emission factors for domestic wastewater, i.e.,

$$ef_i \quad = BOD_i * B_0 * MCF_0 * TCF_i \tag{32}$$

where $BOD_i$ is amount of biochemical oxygen demand per person in country $i$,

$B_0$ is maximum $CH_4$ producing capacity,

$MCF_0$ is the methane correction factor, i.e. the fraction of BOD converted to $CH_4$, and

$TCF_i$ is the temperature correction factor in country $i$.

The temperature correction factor is derived by weighing the rate of methanogenesis at different temperature intervals with the number of days per year in respective temperature interval, i.e.,

$$TCF_i = \frac{\sum_{j=1}^{4} \alpha_j D_{ij}}{365}, \tag{33}$$

where $\alpha_j$ are the rates of methanogenesis (0, 0.1, 0.6, and 0.9) at the four respective temperature intervals ≤ 5°C, 5 to 15°C, 15 to 30°C and > 30°C, and

$D_{ij}$ are the average number of days (over years 2000, 2005 and 2010) when the maximum temperature in a country falls within the respective temperature intervals.

Data on the rates of methanogenesis at different temperature intervals is adopted from Lettinga, Rebac, and Zeeman (2001), while daily data of the maximum temperature for years 2000, 2005 and 2010 at 25km resolution was taken from the Agri4 Cast Data Portal (JRC, 2015).

Country-specific values for the biochemical oxygen demand per person (BOD) are used when available from UNFCCC-CRF (2014). When unavailable, an IPCC (2006, Vol.5, Table 6.4) default factor for the EU of 31.0 kt $CH_4$/million people is used for the maximum $CH_4$ producing capacity ($B_0$). Methane correction factors ($MCF_0$) of 0.1 for uncontrolled decentralized collection and 1 for uncontrolled centralized collection apply.

Industry sectors identified by IPCC (2006, Vol.5, p.6.19) as potential sources for $CH_4$ emissions from wastewater are food, pulp- and paper industry and other manufacturing industries generating wastewater with an organic content, i.e., textile, leather, organic chemicals etc.

The activity data for estimating methane emissions from industrial wastewater is the amount of COD present in untreated industrial wastewater. These amounts are derived from production volumes combined with COD generation factors as specified in Table 16. Production volumes in ton product are taken from FAOSTAT (2015). Growth in value added by industry is used as driver for future projections.

For pulp- and paper industry, wastewater and COD generation rates reported in literature differ considerably between processes and between developed and developing countries. By comparing reported values from different sources, process specific generation rates are derived as presented in Table 16. For the EU countries, default rates derived for developed countries apply. It should be noted that when using process specific generation rates, the estimated amounts of COD and $CH_4$ generated from this industry come out several times lower than if using the IPCC default factor (2006, Vol.5, Table 6.9) for some food industries and pulp- and paper industry.

(30)

$$E_{it} = \sum_{sm} A_{its} * COD_i * ef_i * (1 - remeff_m) * Appl_{itm} \quad (34)$$

where $A_{its}$ is the amount of product $A$ produced in country $i$ in year $t$,

$COD_i$ is the chemical oxygen demand in untreated wastewater generated per tonne product produced in country $i$,

$remeff_m$ is the removal efficiency of technology $m$,

$Appl_{itm}$ is the application of technology $m$ in country $i$ and year $t$, and where

$$ef_i = B_0^{COD} * MCF_0, \quad (35)$$

where $B_0^{COD}$ is maximum $CH_4$ producing capacity,

$MCF_0$ is the methane correction factor, i.e., the fraction of $CH_4$ generated which is not oxidized but released as $CH_4$.

Values for the maximum methane production capacity *($Bo^{COD}$)* of wastewater from different industrial sectors are based on a literature review presented in Table 16. Weighted averages of the values for each process/product for the year 2010 were used to calculate the methane production capacity by sector and country. An IPCC (2006, Vol.5, Table 6.2) default factor of 0.25 kt $CH_4$/kt COD is applied for the maximum $CH_4$ producing capacity ($B_0^{COD}$) when no value was available from literature. A methane correction factor ($MCF_0$) of 0.5 is applied for the uncontrolled case.

There are no wastewater options available that primarily target $CH_4$ emissions. There are, however, several different ways of treating wastewater, which have different implications for $CH_4$ emissions (Pohkrel and Viraraghavan, 2004 and Thompson et al., 2001). When domestic wastewater is centrally collected and emitted to a water body with only mechanical treatment to remove larger solids, plenty of opportunities for anaerobic conditions and $CH_4$ formation are created. For this type of treatment, the methane correction factor (MCF) used in GAINS is 1. With well managed aerobic or anaerobic treatment, the $CH_4$ formation is effectively mitigated and $CH_4$ emissions can be kept on a negligible level. MCF used in GAINS is 0.01 for aerobic treatment and 0.005 for well managed anaerobic treatment. With less well managed systems the occurrence of anaerobic conditions increase as well as $CH_4$ formation (IPCC 2006, Vol.5, Tables 6.3 and 6.8). Anaerobic treatment has advantages over aerobic treatment like lower costs, smaller volumes of excess sludge produced, and the possibility of recovering useful biogas, which can be upgraded to gas grid quality (Lettinga 1995, Thompson et al. 2001). For industrial wastewater, it is assumed that the most effective way to reduce $CH_4$ emissions is to apply a two-stage process where the water is treated anaerobically with recovery of the biogas in a first stage, which is then followed by an aerobic treatment in a second stage (Latorre et al., 2007). The assumed MCF for this type of treatment is 0.05. In rural areas, domestic wastewater can be collected and treated in latrines, septic tanks or similar anaerobic treatment (USEPA, 1999).

Current applications of different treatment practices for domestic and industrial wastewater in EU countries are taken from UNFCCC (2019) CRF tables complemented with information from EUROSTAT (2019), OECD (data downloaded July 2015) and IPCC (2006, Vol.5, Table 6.5). Investment costs for sewage treatment are taken from EEA (2005) and operation and maintenance costs from Hernandez-Sancho and Sala-Garrido (2008). Rural wastewater treatment costs are from USEPA (1999).

Table 16: Literature review of factors used in calculation of industry wastewater emission factors.

| Industry | Product | Wastewater genertion in m3/ton. (range over different studies) | [COD] in kg/m3 Untreated wastewater. (range over different studies) | Maximum $CH_4$ producing capacity in kg $CH_4$/kgCOD. (range over different studies) | References |
|---|---|---|---|---|---|
| Food | Beer | 4.95$^{a}$ (1.98 - 7.92) | 4$^{a}$ (2-6 /1.2 - 125 UK) | 0.23 ~~$^{a}$~~(0.19-0.27) | Debik and Coskun 2009; Kobya, Senturk, and Bayramoglu 2006; Fountoulakis et al. 2008; Şentürk, İnce, and Onkal Engin 2010; AZBAR et al. 2004; Azbar et al. 2009; Healy, Rodgers, and Mulqueen 2007; Brito et al. 2007; Rodgers, Zhan, and Dolan 2004; Sharda, Sharma, and Kumar 2013; Shivayogimath and Jahagirdar 2015; Maya-Altamira et al. 2008. |
| | Vegetables oils$^{c}$ | 0.8$^{a}$ (0.4 - 1.2) | 45.5$^{a}$ (5 -804) | 0.17$^{a}$ (0.11 -0.24 | |
| | Wine | 2$^{b}$ (0.8-14) | 30.4$^{b}$ (3.1-150) | 0.18$^{d}$ | |
| | Sugar Refining | 0.69$^{a}$ (0.16-1.0) | 6.15 $^{a}$ (2.3 -10 ) | n.a. | |
| | Meat | 13 (IPCC) | 5.4 $^{b}$ (3 -11) | 0.22 | |
| | Dairy Products$^{e}$ | 3.05$^{b\ f}$(0.19-10) | 8.8$^{b}$ (0.18 -25.6) | 0.22 $^{b}$ (0.16 -0.27) | |
| Pulp | Bleached sulphate pulp | 70$^{a}$ (30 -110) | 1.55$^{a}$ (0.10-3.0) | n.a. | Janssen et al. 2009; Ekstrand et al. 2013; Larsson et al. 2015; Karlsson et al. 2011; Tezel et al. 2001; Chaparro and Pires 2011; Dufresne, Liard, and Blum 2001; N H and others 2012; Thompson et al. 2001. |
| | Unbleached sulphate pulp | 50$^{a}$ (20 -80 ) | 1.43$^{b}$ (1.35 -2.44) | n.a. | |
| | Bleached sulphite pulp | 70$^{a}$ (40-100) | 2.10$^{b}$ (0.62 - 8) | 0.22$^{b}$(0.20-0.24) | |
| | Unbleached sulphite pulp | 70$^{a}$ (40-100) | 0.80$^{a}$ (0.20 - 1.4) | n.a. | |
| | Mechanical wood pulp | 20$^{a}$ (5-50) | 6.9$^{b}$ (2.71 - 10.37) | 0.19$^{a}$ (0.12 - 0.27) | |
| | Semi-Chemical pulp | 50$^{a}$(20-80) | 2.19$^{a}$ (0.67 -3.71) | 0.19$^{a}$ (0.11-0.27) | |
| | Recovered pulp$^{g}$ | 20 | 3 | n.a. | |
| | Other fibre pulp | 20$^{g}$ | 8.20$^{a}$ (7.7 -8.7) | n.a. | |
| Paper | Newsprint | 9$^{a}$ (5-15) | 3.5 | n.a. | |
| | Printing and writing paper | 60$^{h}$ (60-227) | 0.81$^{a}$ (0.5-1.11) | n.a. | |
| | Recovered paper | 12$^{a}$ (8 - 16) | 0.51$^{a}$ (0.43 -0.58) $^{i}$ | 0.22$^{a}$ (0.16-0.27) | |
| | Household/sanitary/tissue | 8.50$^{a}$ (5-12) | 1.02$^{a}$ (0.05-2) | n.a. | |
| | Wrapping papers$^{g}$ | 20 | 0.08 | n.a. | |
| | Paper and paperboard othe | 12$^{a}$ (8 - 16 ) | 0.95$^{b}$ (0-11) | n.a. | |

[a] Average
[b] Median
[c] Olive oil (primarily centrifugation and pressing production processes), sunflower and cotton seed oil
[d] Based only on one study
[e] Including milk production, cheese, cheese whey, ice cream and butter
[f] Most of the data (11 total) are below 4.0 (8)
[g] based on Höglund-Isaksson, 2012
[h] 60 m3/ton for UK and 227 m3/ton for Thailand
[i] Collected after the clarifier

# 4 $N_2O$ emission sources

## 4.1 Overview of $N_2O$ emission sources and mitigation options in GAINS

$N_2O$ emission sources in the GAINS model include anthropogenic emissions from agricultural soils, handling of livestock manure, combustion, chemical industry (adipic and nitric acid production, caprolactam production), direct use of $N_2O$ in hospitals and food industry, composting and wastewater treatment (Winiwarter et al., 2018). Table 17 provides an overview of the included $N_2O$ emission sources with corresponding mitigation options. Following the requirements of the amendment (EC/29/2009) of the EU-ETS Directive, nitrous oxide emissions from production of nitric acid, adipic acid, glyoxal and glyoxylic acid are included in the EU-ETS emission cap and subject to emission permits under the EU-ETS system (see Table 18). Caprolactam production, which is not part of the EU-ETS, is considered separately.

Table 17: Overview of $N_2O$ emission sources and mitigation options in the GAINS model.

| Emission source | GAINS sector | Sector description | GAINS mitigation technology | Technology description |
|---|---|---|---|---|
| Combustion | CON_COMB -DC/ GSL/ LPG/ MD | Other combustion sources: various fuels | none | No mitigation option identified |
| | CON_COMB -BC1/ BC2/ HC1/ HC2/ HC3 | Other combustion sources: brown coal/hard coal | ISFCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | CON_COMB -GAS/HF | Other combustion sources: gas/heavy oil | IOGCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | CON_COMB2 -BC1/ BC2/ HC1/ HC2/ HC3 | Other combustion sources: fluidized bed -brown coal/hard coal | FBC_CM | Combustion modification in fluidized bed combustion |
| | DOM -BC1/ BC2/ DC/ GAS/ GSL/ HC1/ HC2/ HF/ LPG/ MD/ OS1/ OS2 | Domestic -various fuels | none | No mitigation option identified |
| | IN_BO -DC/ GSL/ HF/ LPG/ MD/ OS1/ OS2 | Industry boilers -various fuels | none | No mitigation option identified |
| | IN_BO -BC1/HC1/HC2 | Industry boilers -brown coal/hard coal | ISFCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | IN_BO -GAS | Industry boilers -gas | IOGCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | IN_BO -BC1/HC1/HC2/ OS1/ OS2 | Industry boilers: solid fuels | FBC | Fluidized bed combustion -*increases* N2O emissions |
| | | | FBC_CM | Combustion modification in fluidized bed combustion |

Table 17 Cont.: Overview of $N_2O$ emission sources and mitigation options in the GAINS model.

| Emission source | GAINS sector | Sector description | GAINS mitigation technology | Technology description |
|---|---|---|---|---|
| Combustion | IN_OC -DC/ GSL/ HF/ LPG/ MD/ OS1/ OS2 | Industry other combustion - various fuels | none | No mitigation option identified |
| | IN_OC -BC1/HC1 | Industry other combustion - brown coal/hard coal | ISFCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | IN_OC -GAS | Industry other combustion -gas | IOGCSN | $NO_X$ abatement option -*increases* $N_2O$ emissions |
| | IN_OC2 -BC1/HC1 | Industry other combustion: fluidized bed -brown coal/hard coal | FBC_CM | Combustion modification in fluidized bed combustion |
| | PP_EX & PP_NEW -DC/ GAS/ GSL/ HF/ MD | Power plants: existing or new - gaseous and liquid fuels | none | No mitigation option identified |
| | PP_EX & PP_NEW -BC1/ BC2/ HC1/ HC2/ HC3/ OS1/ OS2 | Power plants: existing or new - solid fuels | FBC | Fluidized bed combustion -*increases* N2O emissions |
| | | | FBC_CM | Combustion modification in fluidized bed combustion |
| | PP_IGCC, PP_MOD, PP_ENG | Power plants: Integrated Gasification Combined Cycle; Modern (ultra-, supercritical steam of combined cycle); Internal Combustion Engines | none | No mitigation option identified |
| | TRA_OT_AGR/ AIR_DOM/ CNS/ INW/ LB/ RAI - GSL/MD | Non-road transport: various modes -gasoline/diesel | EU-I to EU-VI | Switching to higher Euro standards |
| | TRA_RD_HDB/ HDT/ LD4C/ LD4T/ M4 -GSL | Road transport: bus/trucks/cars/vans/two-wheels: gasoline/diesel | EU-I to EU-VI | Switching to higher Euro standards |
| Industry | PR_ADIP | Adipic acid production | CR | Catalytic or thermal reduction |
| | | | TWIN_RED | Twin reduction technology |
| | PR_NIAC | Nitric acid production | CR | Catalytic or thermal reduction |
| | | | BAT | Best available technology |
| | PR_CAPR | Caprolactam production | CR | Catalytic or thermal reduction |
| | | | BAT | Best available technology |
| Direct $N_2O$ use | N2O_USE | Direct use of $N_2O$ as anaesthetic gas in medicin and as unreactive propellant in food industry | REDUCE | Apply $N_2O$ in combination with other (liquid) anaesthetics |
| | | | REDUCE2 | Apply even less $N_2O$ in combination with other (liquid) anaesthetics |
| | | | REPLACE | Replace $N_2O$ with alternative: e.g., Xe |
| Waste | COMPOST | Composting | none | No mitigation option identified |
| Wastewater | WW_DOM | Domestic wastewater handling | OPTIM | Process optimization to increase the $N_2/N_2O$ ratio in effluent gases |
| Livestock | MANURE-DL/ DS/ OL/ OS/ PL/ PS/ HO/ SH/ LH/ OP | Manure management - solid/liquid systems for various animal categories | none | No mitigation option identified |

Table 17 Cont.: Overview of $N_2O$ emission sources and mitigation options in the GAINS model.

| Emission source | GAINS sector | Sector description | GAINS mitigation technology | Technology description |
|---|---|---|---|---|
| Soils | APPLIC - MANURE_N | Application of manure nitrogen on soils | FERT_RED | Set of good practice measures to reduce fertilizer input (note: considered to be covered in fertilier projections) |
| | | | FERTTIME | Adjusting fertilizer application to the periods of agricultural demand (note: option superseded by more efficient and cost-effective options) |
| | | | VRT_L,<br>VRT_M,<br>VRT_H | Variable rate technology to minimize fertilizer need (three cost levels) |
| | | | INHIB_L,<br>INHIB_M,<br>INHIB_H | Nitrification inhibitors to reduce emission rates (three cost levels) |
| | | | PRECFARM | Optimization of agricultrual nitrogen efficiency by "precision farming" |
| | APPLIC - SOIL_N | Application of mineral fertilizer nitrogen on soils, including crop residues | FERT_RED | Set of good practice measures to reduce fertilizer input (note: considered to be covered in fertilier projections) |
| | | | FERTTIME | Adjusting fertilizer application to the periods of agricultural demand (note: option superseded by more efficient and cost-effective options) |
| | | | VRT_L,<br>VRT_M,<br>VRT_H | Variable rate technology to minimize fertilizer need (three cost levels) |
| | | | INHIB_L,<br>INHIB_M,<br>INHIB_H | Nitrification inhibitors to reduce emission rates (three cost levels) |
| | | | PRECFARM | Optimization of agricultrual nitrogen efficiency by "precision farming" |
| | APPLIC - RICE_N | Application of mineral fertilizer nitrogen and crop residues on rice fields | FERT_RED | Set of good practice measures to reduce fertilizer input (note: considered to be covered in fertilier projections) |
| | | | FERTTIME | Adjusting fertilizer application to the periods of agricultural demand (note: option superseded by more efficient and cost-effective options) |
| | | | VRT_L,<br>VRT_M,<br>VRT_H | Variable rate technology to minimize fertilizer need (three cost levels) |
| | | | INHIB_L,<br>INHIB_M,<br>INHIB_H | Nitrification inhibitors to reduce emission rates (three cost levels) |
| | | | PRECFARM | Optimization of agricultrual nitrogen efficiency by "precision farming" |
| | GRAZE - EX_CTTL | Grazing-related emissions from cattle, pigs, poultry excreta | INHIB_L,<br>INHIB_M,<br>INHIB_H | Nitrification inhibitors to reduce emission rates (three cost levels) |
| | GRAZE - EX_SH | Grazing-related emissions from sheep excreta | none | No mitigation option identified |
| | HISTOSOLS | Area of organic soils used for agricultural purposes | FALLOW | Abandonment of agricultural use |

Table 18: Current legislation affecting $N_2O$ emissions in EU member states.

| **Sector** | **Policy** | **Regional coverage** | **Policy description and implementation in GAINS** |
|---|---|---|---|
| Agriculture | EU Common Agricultural Policy (CAP) and EU Nitrate Directive (EEC/676/1991) with revisions | EU-wide | Reflected in GAINS through input of CAPRI model data on trends in livestock numbers, milk yield and fertilizer use. |
| Waste & wastewater | Legislation to replace current composting with anaerobic digestion of food waste | Germany | In GAINS, the current composting of organic waste is phased-out linearly and replaced with anaerobic digestion between 2020 and 2050. |
| Industry | EU ETS Directive (EC/29/2009): Production of nitric acid, adipic acid, glyoxal and glyoxylic acid. | EU-wide | Industry needs to aquire tradable emission permits under the EU emission trading system (EU-ETS). |

## 4.2 Energy sector

### 4.2.1 Combustion in stationary sources

$N_2O$ is formed as a combustion by-product, similar to the formation of $NO_x$. Available activity data is energy consumption by sector and fuel, which for the EU countries is provided by the PRIMES model (E3modelling, 2020). Stationary combustion sources include emissions from power plants, industry boilers, other combustion in industry, and residential and commercial sector (see Table 17). Emission factors are specified by sector and type of fuel used and taken from IPCC (2006) and de Soete (1993).

No specific mitigation options have been identified for control of $N_2O$ emissions from stationary combustion sources. However, the use of fluidized bed combustion (FBC) and abatement of $NO_x$ (through selective non-catalytic reduction of flue gas) in power plants and industry boilers increases emission factors for $N_2O$. This may in some cases lead to increasing emissions over time, even if starting from a very low level. For FBC, emission factors used in GAINS have been adopted following Tsupari et al. (2007) and Rentz et al. (2002).

Fluidized bed combustion (FBC) is a technology that allows for an extended contact of solid fuels with air oxygen, minimizing the need to crush or pulverize fuels, while at the same time hampering particle formation. Also, combustion temperatures are kept below the optimum for formation of $NO_x$. Lower $NO_x$ emissions are accompanied with strong increases in $N_2O$ emissions. FBC requires advanced methods to properly regulate combustion air flow and fuel intake to achieve a stable fluidized bed. The GAINS database contains expert estimates of the implementation of FBC, including their future development, in combustion of solid fuels in European countries. This data has been specifically addressed at previous country consultations in order to allow for improvement according to national information. Following information from PRIMES (De Vita, 2013) that FBC is not cost efficient for future power plants, its degree of implementation has been set at the 2005 values for existing power plants and zero for new power plants such that we expect FBC to fade out. In industry, we remain with the GAINS database supplemented with country information.

Methods have been developed and implemented in pilot plants, which allow minimizing $N_2O$ formation connected with the use of FBC, i.e., the GAINS technology “combustion modification in fluidized bed combustion”, see Table 17. Data presented by Winiwarter (2005) indicate that 80 percent of $N_2O$ can be removed (Hendriks et al. 2001). Also cost data was taken from this source. No discrimination has been made for applicability or costs in different countries (considered to be 100 percent).

### 4.2.2 Combustion in mobile sources

$N_2O$ emissions from mobile sources are known to be affected by $NO_x$ abatement applied to the vehicle exhausts. $N_2O$ emissions are linked to the $NO_x$ abatement technology applied in the form of catalytic converters or SCR-technology applied in diesel vehicles. Activity data for the EU countries is adopted from the PRIMES model (E3modelling, 2020). Emission factors depend on several factors like fuel, technology and operating characteristics. GAINS uses default emission factors from the European transport emission database COPERT4 (EMISIA, 2011 http://www.emisia.com/copert/General.html). Emission factors are specified by vehicle category, type of fuel used and the emission control standard of the vehicles (EURO I to EURO VI), while averaging for road types and driving conditions. $NO_x$ abatement may lead to increased $N_2O$ emissions, as were observed in the early generations of catalysts. To reflect this, $N_2O$ emission factors are differentiated by the emission control standard. Apart from this, no $N_2O$ specific mitigation options are identified for mobile combustion sources in GAINS.

## 4.3 Industry sector

### 4.3.1 Adipic acid production

The industrial process generating adipic acid (a compound required in the production of Nylon 66 or polyurethane) involves treating the raw material with concentrated nitric acid, at which large quantities of $N_2O$ are released. Typically, for each ton of product 300 kg of $N_2O$ are formed, making the process an important contributor to overall $N_2O$ emissions, although the amount of production is fairly low compared to production of standard chemicals.

Adipic acid production occurs in only a handful of countries, and since only very few production plants are involved (as of 2010, there were 5 plants operational in EU-28), the CRF tables usually list production data as “confidential”, but the exact sites and production capacities are known (Schneider et al., 2010). Therefore, we supplement activity data with capacity and derive future development by country from PRIMES model (E3modelling, 2020) outputs, specifically by scaling according to the value added in chemical industry.

The small number of producers also allows observing general structural changes efficiently. Industry have made voluntary agreements after a cost-efficient method (catalytic reduction) had been developed to take advantage of the high $N_2O$ concentrations in plume to efficiently remove and even convert back into nitric acid (with 95% efficiency). Most plants had been retrofitted by 2000, with the exception of one in Novara, Italy, where this modification started to be fully operative from 2006 only.

As the method in principle is able to reduce 99.9 percent of the emissions, and the remaining emissions are mainly released during operational shutdown of the abatement device, one of the smaller European plants in Krefeld, Germany, recently installed a backup device. This backup device is expected to reduce 80 percent of the remaining emissions (LANXESS, 2008), such that total abatement of 99 percent (95% +80% of 5% remaining) can be achieved, allowing for some system failure still. Costs for the installations have been estimated to be identical to the original system, but the marginal cost of abating the additional 4%-points (considering the much smaller reductions) is of course considerably higher. The option termed "twin reduction technology" is not available in 2015, but with the expected carbon price level of the ETS system (see Section 2.3), we assume full implementation of this option from 2020 onwards.

In this sector, we also cover emissions from Glyoxal/glyoxylic acid production as the underlying chemical process is similar. Glyoxal production employing the nitric acid pathway (thus causing $N_2O$ emissions) is reported from one plant in EU28 only (in France).

### 4.3.2 Nitric acid production

The oxidation of ammonia to nitric acid is one of the large scale industrial processes. Nitric acid is needed both for the production of fertilizer and of explosives. Nitric acid production occurs in many EU countries, often in several installations, but some countries keep activity data confidential. In such cases, production amounts were inferred using emission results and the default unabated emission factor for the base year. Future production development has been scaled according to the value added in chemical industry, taken from the PRIMES model (E3modelling, 2020) outputs.

As a by-product in the oxidation, nitrous oxide is formed. While the amount lost is by far smaller than with adipic acid production, the level of production makes this an important emission source. In nitric acid production also the concentration of the released gas is considerably smaller, making it more difficult to reclaim. Still, industrial scale production has been proven successful in applying catalytic reduction also to nitric acid production (de Soete 1993; de Beer 2001; Kuiper 2001), leading to emission reductions of 80%. The use of information from a demonstration plant in Linz, Austria, as published in the BREF report (EC, 2007) allows for reasonable estimates of the additional costs incurred. Recent information from the same plant (Muik, 2009) demonstrates that a second level of abatement has been made available allowing to reduce emissions by 94%. As the plant operates under commercially favourable conditions, we assume that this more stringent "best available technology" reduction option is available at no extra costs.

As a consequence of the general introduction of emission reduction technologies across EU, between 2007 and 2012 $N_2O$ emissions from nitric acid production dropped to one quarter (UNFCCC-CRF 2014; 2015; 2019), presumably at least in part in connection with introduction of the EU emission trading scheme. We use this UNFCCC data to establish, by country, the respective abated emission factor to assess the level of control already established in a historic year. As the marginal cost of using catalytic reduction to control emissions from nitric acid production is estimated at less than the expected carbon price in the ETS market, thus full adoption of the "best available technology" option seems reasonable in all Member States from 2020.

### 4.3.3 Caprolactam production

The production of caprolactam, which in a similar manner as adipic acid is used to create the polymer Nylon 6, occurs only in five EU countries at a rather limited scale. As with nitric acid production, nitrous oxide is formed as a by-product during the oxidation of ammonia. Information on abatement technology is scarce, also as the contribution to overall emissions are not large. Only with the decline in emissions from nitric acid production, caprolactam becomes the remaining source of industrial $N_2O$, in some countries arriving at or even exceeding the emissions of nitric acid production.

Due to the similarities in process, GAINS uses the same control technologies, emission reduction efficiencies and costs as for nitric acid production. Caprolactam is dealt with separately as it is not included in the ETS.

### 4.3.4 Direct use of $N_2O$

The specific properties of $N_2O$ are taken advantage of in medicine as an anaesthetic gas, in the food industry as an unreactive propellant, and in specific combustion engine applications providing additional oxygen to the combustion process. At least for the first two applications, virtually all of the $N_2O$ used will eventually be emitted to the atmosphere. In both cases, $N_2O$ enters the human body, where it remains only for a short time and is not metabolised. Based on a handful of assessment to support national emission inventories, Winiwarter (2005) extracted an emission factor by population of a country (i.e., GAINS sector N2O_USE).

Methods to reduce application of $N_2O$ have been derived in hospitals, mostly due to concerns about workplace security for hospital personnel. Medical research allows to supplement or even to fully replace the use of $N_2O$ (Spakman et al. 2003; Nakata et al. 1999). While the use of combined anaesthetics to supplement $N_2O$ application can be observed in practice following national sale statistics, data on replacement are highly speculative.

## 4.4 Waste and wastewater sector

Microbial processes are responsible for conversion of nitrogen compounds contained in waste or wastewater, during which process also $N_2O$ is released. GAINS uses default emission factors derived by IPCC (2006) to quantify these emissions. Specifically, emissions occur during composting and in wastewater treatment. Both emission sources are small sources.

For composting, this default factor refers to the amount of wet waste entering treatment. No differentiation is being made for specific process or type of material. Also, no options are known to mitigate such emissions.

In sewage treatment, the amount of nitrogen added depends on the number of people connected. In consequence, emissions can be estimated by population number on a country scale. GAINS uses a simple emission factor per inhabitant for assessing wastewater related emissions (Behrend et al., 2004). In line with Hendriks et al. (1998), it is assumed that an optimization of process parameters will lead to a decrease of emissions by 40%, without compromising on any of the other operating requirements of the facility, specifically not affecting downstream N content. This measure is assumed to not need any specific investment or operation cost.

# 4.5 Agriculture sector

## 4.5.1 Agricultural and grassland soils

### 4.5.1.1 Emissions of $N_2O$ from soils

Microbial processes in soil convert ammonia into nitrate (nitrification) and further to molecular nitrogen (denitrification). The processes occur in soil under aerobic and anaerobic conditions, respectively, and both release $N_2O$ as a side product. Soil processes are by far the most important source of $N_2O$.

Despite a considerable amount of on-going research, there are still important gaps in knowledge about $N_2O$ release from soils. Especially, the amount of $N_2O$ formed and converted while still in the soil (during diffusion to the surface) seems difficult to assess, but is needed to obtain the overall release rate in a process based approach. Chamber measurements on top of the soil yield highly variable results. As a consequence, uncertainty associated with the emission figures has been estimated as an order of magnitude, when emissions are related to the input of nitrogen (IPCC 2006). Despite of contributing only a minor fraction to overall greenhouse gas emissions, soil $N_2O$ emissions are typically responsible for the major part of uncertainty in a national greenhouse gas inventory (Winiwarter and Rypdal, 2001; Winiwarter and Muik, 2010). Lately there has been some evidence that, averaged over long time periods and large areas, inventories using IPCC methodology provide a very useful tool to understand the real world emissions. On a global scale, Mosier et al. (1998) proved that emissions are in agreement with observed trends of $N_2O$ accumulation in the atmosphere. Using inverse modelling, Bergamaschi et al. (2015) demonstrated in general reasonable agreement between observations on tall towers in Europe, and European inventories. Moreover, assessments of global $N_2O$ developments based on more generic parameters (freshly created reactive nitrogen, or total mineral fertilizers and total manure only) note that the IPCC total “is not inconsistent with the total derived by the top-down methodology” developed therein (Crutzen et al., 2008; Davidson, 2009).

Nitrous oxide emissions are typically assessed as a fraction of the nitrogen deposited on soils. Nitrogen input in GAINS is derived from nitrogen contained in mineral fertilizer, animal manure and crop residue left on the field. Information on mineral fertilizer use and projections till 2070 derive from results of the CAPRI (EuroCare, 2021) model, which also provides numbers of animals on farms. Assessing the amount of animal manure also requires nitrogen excretion rates, data that are available in the national reports (UNFCCC, 2019) and have been integrated in the GAINS model with the help of national experts during consultations. The amount of crop residues and their nitrogen content is again taken from national reports (UNFCCC, 2019), with no temporal trend assumed.

As from 2015, countries change reporting methodology in their national greenhouse gas inventory. With completion of the Kyoto period, using the IPCC (2006) guidelines has become mandatory. One major difference for agricultural $N_2O$ emissions is a different treatment of indirect emissions due to nitrate leaching. Nevison (2002) discovered and described in detail these issues that lead to a reevaluation of the situation by IPCC (2006), which under default conditions decreases indirect emissions due to leaching by a factor of more than 3, such that overall soil-related $N_2O$ emissions (direct & indirect) are reduced by about a third.

Consequently, also the algorithm for assessing soil $N_2O$ emissions in GAINS has been updated. As previously, there is no differentiation between direct and indirect emissions. Instead, IPCC default data has been used to assess the level of indirectly produced $N_2O$ (as a result of atmospheric redeposition of ammonia, or nitrate leaching) and aggregated into an overall emission factor. IPCC (2006) reports distinctively different emissions from grazing (of which sheep have distinctively lower emissions than cattle) and from direct application of fertilizers (mineral fertilizers, animal manure and crop residue nitrogen have largely similar impacts) to soils. Wetland rice production is associated with a lower emission factor. Avoiding a category split not supported by data, the following activity clusters having the same emission factor were created in GAINS: grazing (all animals except sheep), sheep grazing and manure application, mineral fertilizer and crop residues application on soil and grassland, and fertilizer application on wetland rice (see Table 19).

Table 19: Emission factors of $N_2O$ from soils as used in GAINS. Default release rate (given in % of N applied), shares of re-emissions and leaching were taken from IPCC (2006), the conversion to total $N_2O$ emitted uses a stoichiometry factor of 1.57

| IPCC 2006 | manure / grazing (sheep) | grazing (all other animals) | urea | other mineral fertilizer | crop residues | flooded rice |
|---|---|---|---|---|---|---|
| direct soil emissions [%] | 1.000 | 2.000 | 1.000 | 1.000 | 1.000 | 0.300 |
| indirect / atmos deposition [%] | 0.200 | 0.200 | 0.150 | 0.030 | 0.000 | 0.150 |
| indirect leaching [%] | 0.225 | 0.225 | 0.225 | 0.225 | 0.225 | 0.225 |
| Total $N_2O$-N [%] | 1.425 | 2.425 | 1.375 | 1.255 | 1.225 | 0.675 |
| Total $N_2O$ [kg $N_2O$/kg N applied] | 0.022 | 0.038 | 0.022 | 0.020 | 0.019 | 0.011 |
| | | | | | | |
| GAINS emission factors [kg $N_2O$/kg N applied] | 0.022 | 0.038 | 0.020 | | | 0.011 |

Technological options to reduce $N_2O$ emissions may target either on the nitrogen input, or on the release rate of $N_2O$. Following the tier 1 level of the IPCC (2006) methodology, reducing nitrogen input (increasing NUE) is the only applicable way to get emission reductions accepted, thus the majority of measures available in the literature focusses on reduced inputs. Increased NUE's have been observed over the last years in Europe (OECD, 2008), in part as a consequence of fertilizer savings also triggered by environmental considerations. Further such changes are implemented in the CAPRI (EuroCare, 2021) fertilizer projections, improvements deemed technologically possible may in part be factored in already as part of the projections. Using CAPRI's Nitrogen budgets, it is evident that an improvement of NUE takes place over the projected period. Measures implicitly assumed by CAPRI in part reflect reduction technologies and are therefore accounted for when assessing additional potentials for N-input reductions in future abatement in the GAINS model.

Thus, with the update of the algorithm to assess soil $N_2O$ emissions, GAINS also updates abatement technologies. Based on a thorough review of available scientific literature (Winiwarter and Sajeev, 2015), emission reductions have been reassessed and technologies available in agriculture have been updated. The resulting reduction efficiencies are described in detail below. Using further literature with a specific focus on the European situation (e.g., Tavella et al., 2010, from the "FutureFarm" research project), cost information has been further refined. Cost data are now split into investments, operation costs and (financial) benefits derived from implementing a given technology.

#### 4.5.1.2 Emission abatement technologies for $N_2O$ emissions from soils

As with emissions generally, there are two directions abatement could take: controlling emissions, or reducing activities. In the case of $N_2O$ from soil, lowering fertilization rate (i.e., reducing input activities) is clearly a chance to also reduce emissions wherever the nitrogen use efficiency is low. Obviously such measures would not change the amount of manure or crop residues available to deliver nutrients, but if used more effectively such organic nutrients would be able to further reduce mineral fertilizer application. Further technology options exist to reduce emissions while maintaining fertilizer levels. While combination of both types of technologies is conceivable, evidence seems not to suffice for inclusion in GAINS.

The concept of abatement technology used here basically follows the concept developed by Gale and Freund (2002) and updated for use in GAINS according to Winiwarter (2005). It describes, for the same set of "activity", N application on agricultural soils, subsequent abatement stages of increasing stringency and at the same time increased costs. In addition to the original approach, now also improvements in nitrogen use efficiency are considered, and the cost-dependence ("economy of scale") of the size of farms has been investigated. In this new structure, the three technologies effectively considered are: "variable rate" application of fertilizers, agrochemical "inhibitors" (limiting the microbial processes that release $N_2O$) and "precision farming" (as a high-tech tool to minimize and optimize agricultural production).

#### 4.5.1.3 Fertilizer reduction measures and efficiency improvements

The simplest and cheapest form of reducing excess fertilizer application is to reduce its wastage. "Fertilizer reduction" subsumes a set of good practice approaches to avoid fertilizer losses to ditches and to set-asides, or excess application by inappropriate or misadjusted spreaders. Following previous studies, Winiwarter (2005) assumed 6% of fertilizer could be saved this way.

As GAINS obtains external information on future fertilizer application, specifically using the results of the CAPRI model (Britz and Witzke, 2014), it is important to provide consistent assumptions. CAPRI internally estimates fertilizer demand in part from plant requirements, and it also projects agronomic improvements based on past trends. On an EU average, the modelled improvement between 2005 and 2030 amounts to 6.6%, which comes close to the 6% reduction expected from the simple measures to reduce fertilizer application. We thus conclude that improvements expected by CAPRI largely refer to measures defined as simple improvements, and "fertilizer reduction" should therefore be assumed to be already fully implemented without costs in the baseline scenario by 2030. No emission reductions will result as implicitly the reduced fertilizer consumption rates will cover these reductions.

#### 4.5.1.4 Farm sizes

Choosing agronomic practice and developing economically viable strategies for farms will often also depend on the farm size. This is also the case for emission abatement. In the case of air pollutants (ammonia, specifically), an elaborate scheme has been worked out to assess abatement costs on the basis of animal numbers per farm averaged by country (Klimont and Winiwarter, 2015). The same "economy of scale" principle also applies to agronomy, but in contrast to animal husbandry it is the farm area that will determine the scale of activities. Especially for one-time costs that can be distributed over a large area on large economic units, such as investments or one-time training costs, cost advantages may occur for larger units. Smaller units, however, do also have opportunities to take advantage of expensive technology etc., which they can more easily share as needed only for a short time period to cover the smaller area. Traditionally this has been organized in machinery rings or by using the service of a contractor.

We use here farm sizes as collected by EUROSTAT in their agricultural data on farm structure, specifically those for arable crops (Table ef_alarableaa, available at http://ec.europa.eu/eurostat/data/database). For each EU member country, the shares of arable area in a given size class of utilized agricultural area (>150 ha, 30-150 ha, and <30 ha) have been extracted und used for further differentiation. We apply these shares also to grassland and permanent crops as arable land constitutes the largest share of fertilized agricultural area in almost all countries, and we understand that the farm size structure of a country is well represented. An alternative EUROSTAT table (ef_oluaareg) only provides differentiation up to 100 ha utilized agricultural area per farm and was therefore not selected. In contrast to the methods adopted to describe increasing animal numbers per farm (see section 3.3), the agricultural area of 2010 has been used throughout to depict farm sizes, even if data from different years indicate a consistent trend towards increased size could be derived here, too.

#### 4.5.1.5 Variable rate technology

Precision agriculture provides nutrients to plants only to the extent they need it. At high prices of agricultural commodities and technical progress, there is a long-standing discussion (see e.g. Auernhammer, 2001) whether saving fertilizer at a certain point would not compensate any expenditure for technology, such that precision agriculture might become profitable. Looking into the lower-cost options of precision agriculture, here we consider the use of sensors and yield maps to estimate the fertilizer requirements on the scale of square meters, combined with applicators that individually regulate the dose spread according to the locally optimized needs (Variable rate technology, VRT). Winiwarter and Sajeev (2015) compile the results of ten individual studies and reviews which, using sensor technology, on the average allow to reduce nutrient inputs by 24% to arrive at comparable yields. In a situation of 2030, assuming general efficiency improvements having taken place (6%, see above), implementation of such a technology still would decrease application rates by 19%. It is interesting to note that some authors (Sehy et al., 2003) observed $N_2O$ emission reductions clearly higher than the fertilizer application decrease (34% at 18% less fertilizer) which could be explained by less excess nitrogen available – but has not been used here. Instead, emissions remain to be considered proportional to N fertilizer application.

Very little information is available on costs, even while the technology actually exists and is on the market. Cost assessment, in some cases, referred also to different farm sizes, or at least was provided for a given farm size. Operation costs and investment for sensor and spreaders have been made available from U.S. studies (ICF, 2013; Koch et al., 2004). In order to apply instrumentation for a European situation, here we decided to apply the cost estimates of the “FutureFarm” project (Tavella et al., 2010), who (for a 500 ha farm) require one set of sensor & spreader. Moreover, in order to maintain consistency with the emission reductions derived from our literature survey from the sensor system, we include also costs of an “auto guidance” system. This systems provides detailed information on the location of agricultural machinery and allows, by precise GPS guidance, to fully prevent overlaps in spreading. One system is needed per 250 ha (Tavella et al., 2010). Applying this to a 250 ha farm (which we use as representative for all farms at >150 ha), we arrive at total investments of EUR 56000, annualized over ten years, plus 5% maintenance costs and flat costs for the precision GPS signal. Note this investment is about 2-3 times that assumed in U.S. studies (ICF, 2013; Koch et al., 2004). Benefits are fertilizer savings due to a 19% lower application rate, which we estimate at 1.11 €/kg N (following an estimate by KTBL, personal information transmitted to JRC). With costs and benefits closely matching, results are volatile with respect to assumptions taken – i.e., under certain real conditions costs may be higher while for other situations VRT may actually be profitable (see detailed emission factors and cost data provided in Table 20).

Basically the same considerations apply to the use of VRT for manure spreading. As manure cannot be saved, we understand that it can be distributed to a larger area and so save mineral fertilizer elsewhere. Savings, however, are estimated only at 80% of the N content.

For farms smaller than 150ha, investments needed would soon render VRT very costly, while the equipment would remain idle for most of the year. Sharing of equipment, under such conditions, seems a viable option. While some studies (ICF, 2013) have ignored that “because 70–80% of farmers who currently use [a specific VRT] technology purchase it”, the current market situation cannot be seen representative for the costs of a future activity. We use the additional costs estimated by Koch et al. (2004) for contracting, scale it by the significantly higher investments assumed for the European situation, and apply the resulting costs to farms between 30 and 150 ha size. For even smaller farms, we understand an organization overhead of 20% is needed in addition. Revenues in both cases are reduced fertilizer application, and again resulting costs reflect an average condition and may be different in individual situations as local variation cannot be adequately modelled here.

#### 4.5.1.6 Inhibitors

Enhanced efficiency fertilizers have been developed to allow a continuous supply of nutrients during the growth phase of plants. As urease inhibitors and nitrification inhibitors, key compounds of the so-called “stabilized fertilizers”, prevent processes that release $N_2O$, we specifically looked into the effect of reduced emissions as a consequence of such inhibitors. Following the literature survey by Winiwarter and Sajeev (2015), 14 studies were evaluated covering different parts of the world and often several types of enhanced efficiency fertilizers. Inhibitors were among the most efficient, by average demonstrating an emission reduction of 38%. Here we do not account for possible simultaneous fertilizer reductions, and we also will not consider the resulting economic benefit – consistent with the way emission reductions are being reported in the background literature. Compared to an improved 2030 situation due to improved nitrogen use efficiency already in the baseline (see above), emission reductions are still as high as 34%.

The effectiveness and considerable achievable emission reductions have been noted before (see e.g. the review by Snyder et al., 2014). However, inhibitors are substances that affect the soil microflora (Freibauer, 2001) and may exhibit possible unintentional side effects, which call for monitoring programs.

Limited information on costs is available, which basically is a consequence of the incremental cost charged to treated fertilizer – inhibitors typically will be applied in combination with or as compounds of fertilizers. Estimates from fertilizer industry indicate costs to be clearly below 10% of the original fertilizer. Costs derived from market prices (Carson and Ozores-Hampton, 2014) matched both the industry quote and the estimate used by Winiwarter (2005). We apply this estimate and add 10% training and other transaction costs (for a 250 ha farm). The same training costs in absolute terms, also applied to smaller far sizes, lead to cost increases for smaller farms, such that the training element for 20ha farms exceed the fertilizer cost increments. Training costs alone have been quoted at 300 GBP per farm every 5 years (Godwin et al., 2003), as annual costs less than an order of magnitude smaller than training and transaction costs of 365 EUR per farm and year which have been derived as above. Resulting emission factors and cost data are provided in Table 20.

Inhibitors can be generally used, also for manure and even animal droppings and urea patches. While use during application is rather straightforward, additional labor is needed to treat animal excreta on pasture. O'Brien et al. (2014b) estimate inhibitors are applicable to urine (60% of N excreted) in 50% of cases, reducing direct & indirect emissions by 50%, while Di and Cameron report reductions of  direct emissions by 80%. Here we use 80% generally, applicable to half of the urine excreted, thus total reductions amount to 24% (0.6x0.5x0.8). This reduction applies to pastures and thus is unaffected by the improved nitrogen use efficiency relevant in other cases. Costs were converted from data provided by O'Brien et al. (2014b), reflect the tedious additional work needed and thus represent an upper level of the cost curve (see Table 20).

#### 4.5.1.7 Precision farming

Soil testing, yield mapping, additional sensors including airborne equipment all have been discussed as means to further optimize nutrient supply to plants. Both efficiency in emission reduction and costs exceed those of VRT. While little information is available, we allow abatement marginally higher than inhibitors, and unit costs as already selected by Winiwarter (2005) and Gale and Freund (2002). Precision farming thus appears at the top of the cost curve (Table 20).

#### 4.5.1.8 Cost curve parameters

Resulting from the above assumptions, Table 20 displays the detailed emission factors and cost assumptions used in agriculture, based on IPCC (2006) guidelines.

Table 20: Emission factors and abatement costs in agriculture (except histosols), expressed as € of 2010. Large farms (>150, nominal average set at 250 ha), Medium farms (30-150, nominal average set at 80 ha) and Small farms (<30, nominal average set at 20 ha) are considered separately for some abatement measures. Default fertilizer input used is 170 kg N/ha, the effectiveness of manure as fertilizer has been set to 80%.

| Sector | Abatement | Cost structure | Investment [M€/ (kt N/yr)] | Lifetime [yr] | O&M [M€ / kt N] | Savings [M€ / kt N] | Uncontrolled emissions [kt $N_2O$ / kt N] | Controlled emissions [kt $N_2O$ / kt N] |
|---|---|---|---|---|---|---|---|---|
| Mineral fertilizer | VRT | Large farms | 1.32 | 10 | 0.034 | 0.21 | 0.020 | 0.0162 |
| | | Medium farms | | | 0.249 | 0.21 | 0.020 | 0.0162 |
| | | Small farms | | | 0.299 | 0.21 | 0.020 | 0.0162 |
| | Inhibitors | Large farms | | | 0.095 | | 0.020 | 0.0132 |
| | | Medium farms | | | 0.113 | | 0.020 | 0.0132 |
| | | Small farms | | | 0.194 | | 0.020 | 0.0132 |
| | Precision Farming | | | | 0.302 | | 0.020 | 0.0132 |
| Manure | VRT | Large farms | 1.32 | 10 | 0.034 | 0.17 | 0.022 | 0.0178 |
| | | Medium farms | | | 0.249 | 0.17 | 0.022 | 0. 0178 |
| | | Small farms | | | 0.299 | 0.17 | 0.022 | 0. 0178 |
| | Inhibitors | Large farms | | | 0.095 | | 0.022 | 0.0145 |
| | | Medium farms | | | 0.113 | | 0.022 | 0.0145 |
| | | Small farms | | | 0.194 | | 0.022 | 0.0145 |
| | Precision Farming | | | | 0.302 | | 0.022 | 0.0140 |
| Grazing (cattle) | Inhibitors | grazing (all except sheep) | | | 0.810 | | 0.038 | 0.0289 |

VRT – Variable Rate Technology

O&M – Operation and Maintenance

Note that in GAINS no emission abatement options are foreseen for the categories "grazing sheep" and "flooded rice" production, hence they have been excluded from the table above.

### 4.5.2 Organic soils

Soil processes in organic soils do not differ from those in other soils, but the larger amount of carbon available provides "feed" for micro-organisms which become more productive. Organic soils (histosols) are thus treated separately in national greenhouse gas inventories (IPCC, 2006). Thus the area of histosols used for agricultural purposes (HISTOSOL, presented in Mha area) is taken from the national submissions to the UNFCCC (2019).

As emissions are large compared to other soils, and the overall area of organic soils under cultivation is fairly low in all countries, the obvious abatement option is to stop utilizing these soils for agricultural purposes. This option has been implemented in GAINS, even if studies on abandoned Finnish histosols (Maljanen *et al.*, 2004) indicate that banning cultivation may in reality not return the emission situation to the natural background. Emission reductions are taken as the difference between natural background and cultivated histosols according to IPCC (2006) which indicate 94% reductions, and costs have been estimated as lost revenues from agricultural production of 500 €/ha (e.g. Riester et al., 2002).

### 4.5.3 Livestock

When the organic content in livestock manure decomposes, emissions of $CH_4$ and $N_2O$ are released. While $CH_4$ release occurs when manure is handled under anaerobic conditions, formation of $N_2O$ occurs during aerobic conditions, i.e., with access to oxygen. Activity data is number of animals by animal category and type of manure management system (solid/liquid). Sources for historical animal numbers are EUROSTAT (2019) for EU countries with projections based on future trends taken from the CAPRI model (EuroCare, 2021).

Emissions of $N_2O$ from manure handling are calculated as a fraction of the total nitrogen excretion, where the size of the fraction depends on the type of manure management. Both animal number and nitrogen excretion rates required for this calculation are elements of the national submissions to UNFCCC (2019), projections of animal numbers are available from the CAPRI (EuroCare, 2021) model.

For dairy cows, both enteric fermentation emissions and manure management emissions per animal are affected by the milk productivity of the cow. This effect is particularly accentuated for highly productive milk cows. To capture this, the no control emission factor for dairy cows is specified as the sum of a fixed emission factor per animal for cows producing up to 3000 kg per head per year and an additional term describing the emission factor per milk yield for milk production exceeding the productivity level 3000 kg per animal per year, i.e.,

$$ef_{it;cow}^{NOC} = ef_i^{animal} + ef_i^{milk} * (x_{it} - 3000) \tag{36}$$

where $ef_i^{animal}$ is the default emission factor for cows in country *i* producing 3000 kg milk per year,

$ef_i^{milk}$ is the emission factor per kt milk produced above the threshold level 3000 kg milk per animal per year, and

$x_{it}$ is the average milk yield per animal in country *i* and year *t*.

Increased nitrogen excretion associated with high milk yields (above 3000 kg/hd) is considered at an increase rate provided by national experts during consultations (Klimont, pers. communication).

Methodology and default emission factors specified by animal categories and manure management system (liquid vs. solid) follow IPCC (2006) guidelines (Table 21). No emission abatement technologies are considered in GAINS for $N_2O$ emissions from manure handling.

Table 21: Emission factors for manure handling following IPCC (2006), Vol. 4, Table 10.21. The conversion to total $N_2O$ emitted (from originally $N_2O$-N) uses a stoichiometry factor of 1.57.

| GAINS manure system | Emission factor [kg $N_2O$/kg N excreted] | | IPCC characterization |
|---|---|---|---|
| poultry | 0.0016 | | Poultry manure with/without litter |
| solid manure systems | 0.0157 | | Cattle and Swine deep bedding |
| liquid manure systems | 0.0079 | | Liquid/slurry with cover |

Changes over time occur, however, in terms of the manure systems, with general trends towards an emission reduction due to liquid manure systems becoming more prevalent. Information on such a change over time derived from country experts during consultations (for a few countries only), or from the extrapolated trend in increasing animal numbers per farm (see section 3.3). As large farms (with a high number of animals) tend to be on liquid systems, it is assumed that the separation between liquid and solid systems is strictly along a given number of animals on a farm, and that farms above a certain (country specific) size automatically will use liquid systems. This allows deriving a liquid/solid share for any future scenario, which is applicable for pigs, dairy cows and other cattle, for which such solid/liquid share information is available.

# 5 F-gas emission sources

## 5.1 Overview of F-gas emission sources and mitigation options in GAINS

The GAINS model framework accounts for the F-gas emissions hydrofluorocarbons (HFCs), hydrochlorofluorocarbons (HCFCs), perfluorocarbons (PFCs) and sulphur hexafluoride ($SF_6$). For the emission scenarios produced for the EUCLIMIT5 project, only HFCs, PFCs and $SF_6$ emissions are considered as these were regulated under the Kyoto Protocol during its first commitment period. In the post-2012 period, nitrogen trifluoride ($NF_3$)[3] –a potent greenhouse gas- was added to the 'basket' of greenhouse gases covered by the Kyoto Protocol. $NF_3$ is manufactured by the reaction of hydrochloric acid (HCL) and ammonia ($NH_3$) and primarily used in the cleaning of silicon wafers in semiconductor applications (Arnold et al., 2013). In the EUCLIMIT5 project, $NF_3$ emissions are reported separately in addition to the other F-gases.

F-gas emissions have increased significantly in recent years and are estimated to rise further in response to increased demand for cooling services and a phase out of ozone-depleting substances (ODS) under the Montreal Protocol (Gschrey et al. 2011, Velders et al. 2015; Purohit and Höglund-Isaksson, 2017; Purohit et al., 2020). In compliance with the Montreal protocol (UNEP, 2007), many applications that formerly used chlorofluorocarbons (CFCs) for cooling and refrigeration purposes changed rapidly to applications employing HCFCs with lower ozone-depleting effects or to HFCs with no ozone-depleting effects. Later amendments to the Montreal protocol require a complete phase-out of all ozone-depleting substances including HCFCs (UNEP, 2007). In GAINS, all use of CFCs in mobile air conditioners in the EU-27 is assumed phased out by 2005, while in stationary sources about 20 percent of the total demand for HFCs/HCFCs is still covered by HCFCs in 2005 followed by a gradual phase-out completed by 2020.

The Kigali Amendment (KA) to the Montreal Protocol (UNEP, 2016) went into force in January 2019 and is expected to phase-out global HFC consumption by 2050, thereby significantly reducing HFC emissions over the period leading up to 2050 (Purohit et al., 2020). The EU F-gas regulations (EC) No 517/2014 and (EC) No 842/2006 together with the EU Mobile air conditioners (MAC) Directive are expected to overachieve on the commitments EU have under the KA. In the EUCLIMIT5 scenarios, effects on baseline emissions from adopted F-gas regulation are therefore reflecting EU F-gas regulation, thereby also fulfilling commitments under the KA.

Table 22 presents the structure of F-gas emission sources (for the Kyoto protocol gases HFCs, PFCs and $SF_6$) in GAINS with corresponding mitigation options. HFCs are primarily used as cooling agents in refrigeration and air-conditioning appliances, as solvents in industrial processes, as fire-extinguishing agents, for foam blowing and as aerosol propellants. Emissions of HFCs from these sources can be reduced through good practices like leakage control and end-of-life recollection and recycling. For more substantial removal of emissions, the HFCs need to be replaced by alternative substances with low or no global warming potential (GWP). The production of HCFC-22 for feedstock use in industry also gives rise to HFC emissions

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[3] $GWP_{100}$ = 16,100 as per IPCC/AR5

unless these are controlled through incineration. Sources of PFCs accounted for in the GAINS model are primary aluminium production and the semiconductor industry. The $SF_6$ sources considered are high and mid- voltage switches, magnesium production and casting, soundproof windows and some small sources like tyres and sports equipment. For more details on activity data, emission factors, mitigation options, and emissions estimation methodology applied in GAINS, see respective sector descriptions below.

An overview of the EU legislation currently in place to control the release of F-gas emissions is presented in Table 23. Baseline scenario emissions are in GAINS estimated under the assumption that all EU Member States at least meet the requirements of the regulations adopted EU wide. From 1st January 2015 the placing on the market and use of F-gases within the EU has to follow the rules given in Regulation (EU) No 517/2014. This Regulation has repealed Regulation (EC) No 842/2006 with effect from 1st January 2015.

The EU F-gas Regulation (EC 842/2006) was implemented in 2006 to control the release of F-gases from stationary cooling and refrigeration equipment as well as from aerosols, foams and a few other minor sources. Emissions from air conditioners in motor vehicles are regulated in the EU MAC Directive (2006/40/EC) and in the EU Directive on end-of-life vehicles (EC 53/2000). These F-gas regulations stipulate e.g., that good practice measures like leakage control and end-of-life recollection of refrigeration and air-conditioning equipment should be put into place by 2008. Allowing for a certain phase-in period in countries with no prior F-gas regulation, GAINS assumes the full effect of good practice measures is attained from 2015 onwards. From 2011, the use of HFC-134a in mobile air conditioners should be replaced by a cooling agent with a GWP of less than 150 in all new vehicle models placed on the market (Minjares, 2011). In addition, the EU MAC directive (2006/40/EC) bans the use of HFC-134a in mobile air-conditioners in all new cars and vans from 2017.

The new EU F-gas regulation (517/2014) came into force from 1st January 2015, replacing the previous version EC (842/2006). Key changes include: product and equipment bans, service and maintenance bans, cap and phase down and responsible supply. The regulation puts restrictions on the placing on the market of certain refrigeration and air conditioning equipment, foams and propellants using F-gases, and of $SF_6$ in small magnesium foundries. It limits the use of higher GWP gases, such as HFC-404A and HFC-507A, in existing refrigeration equipment from 2020. Further, the new regulation puts a cap on the placing on the market of F-gases and phase down on the supply of HFCs. Only companies with EU quotas will be able to supply HFCs to the EU market. Numerous additional changes include requirements for leakage control, checks and detection systems, end-of-life recovery, training and certification programmes, labelling and reporting (EC 517/2014).

In addition, the new F-gas Regulation requires an increased use of alternative blowing agents for one-component foams, use of alternative propellants for aerosols, leakage control and end-of-life recollection and recycling of high- and mid- voltage switches, $SF_6$ replaced by $SO_2$ in magnesium production and casting, and a ban of use of $SF_6$ in soundproof windows, sports equipment etc. The EU's 2006 regulation only restricted use of $SF_6$ in magnesium die casting operations. The bans on the use of $SF_6$ in the recycling of magnesium die casting alloys are new to the 2014 regulation.

The new EU F-gas regulation (517/2014) phases out the total amount of HFCs that can be sold from 2015 to one fifth of today's sales by 2030. This is expected to reduce the EU's F-gas emissions by some two-thirds in 2030 compared to the 2014 level. Table 24 summarizes how different control options have been implemented in the GAINS baseline scenario in order to meet the F-gas emission reduction target specified in the 2014 F-gas regulation.

Following the requirements of the amendment (EC/29/2009) of the EU-ETS Directive, PFC emissions from the primary aluminium industry are included in the EU-ETS emission cap.

There is a voluntary agreement among semiconductor producers worldwide to reduce release of PFC emissions to 10 percent below the 1995 emission level by 2010 (WSC 2008). The European semiconductor industry has signed up to this agreement (ESIA, 2006). HFCs used in the manufacture of semiconductors are excluded from the phase-down in the new F-gas regulation (EC 517/2014).

In addition to EU wide legislation, there is national legislation targeting F-gases in Austria, Belgium, Denmark, Germany, Netherlands and Sweden. These regulations were put in place earlier or go further in their stringency or address more sources than the EU wide regulation.

Table 22: Overview of F-gas sources and mitigation options in the GAINS model.

| Gas | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| HFCs | CAC_SMALL_B | Small commercial air conditioning, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Propane (R290) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | LOW_GWP | Alternative refrigerant: HFO-1234yf, HFO-1234ze |
| | CAC_SMALL_S | Small commercial air conditioning, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Propane (R290) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | LOW_GWP | Alternative refrigerant: HFO-1234yf, HFO-1234ze |
| | CAC_LARGE_B | Large commercial air conditioning, emissions banked in equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | ALT_WATER | Alternative refrigerant: Water |
| | | | LOW_GWP | Alternative refrigerant: HFO-1234yf, HFO-1234ze |
| | CAC_LARGE_S | Large commercial air conditioning, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_WATER | Alternative refrigerant: Water |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | LOW_GWP | Alternative refrigerant: HFO-1234yf, HFO-1234ze |
| | COMM_SMALL_B | Small commercial refrigeration, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-600a, HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | COMM_SMALL_S | Small commercial refrigeration, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-600a, HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | COMM_LARGE_B | Commercial refrigeration, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | COMM_LARGE_S | Large commercial refrigeration, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-152a, HFC-32, HFC-446A, HFC-447A) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | DOM_S | Domestic small hermetic refrigerators, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Isobutane (HC-600a) |
| | IND_SMALL_B | Small industrial refrigeration, including food and agricultural sectors, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-446A, HFC-447A) |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234ze |
| | | | ALT_NH3 | Alternative technology: Ammonia (NH3) |
| | IND_SMALL_S | Small industrial refrigeration, including food and agricultural sectors, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-446A, HFC-447A) |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234ze |
| | | | ALT_NH3 | Alternative technology: Ammonia (NH3) |
| | IND_LARGE_B | Large industrial refrigeration, including food and agricultural sectors, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-446A, HFC-447A) |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234ze |
| | | | ALT_NH3 | Alternative technology: Ammonia (NH3) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | IND_LARGE_S | Large industrial refrigeration, including food and agricultural sectors, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-446A, HFC-447A) |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234ze |
| | | | ALT_NH3 | Alternative technology: Ammonia (NH3) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | RAC_B | Residential air conditioning, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-32) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | LOW_GWP | Placeholder for speculative future HFO alternative refrigerant |
| | RAC_S | Residential air conditioning, emissions from scerapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-32) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | | | LOW_GWP | Placeholder for speculative future HFO alternative refrigerant |
| | TRA_REFB | Refrigerated transport, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-32) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_REFS | Refrigerated transport, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Hydrocarbons (e.g., HC-290, HC-1270) |
| | | | ALT_HFC | Alternative HFC refrigerant (e.g., HFC-32) |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |

Continued: Table 22: Overview of F-gas sources and mitigation options in the GAINS model.

| Gas | GAINS sector | Sector description | GAINS technology | Technology description |
|---|---|---|---|---|
| HFCs | TRA_RD_HDB_B | Mobile air conditioning in buses, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_RD_HDB_S | Mobile air conditioning in buses, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_RD_LD4C_B | Mobile air conditioning in cars, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_RD_LD4C_S | Mobile air conditioning in cars, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_RD_LD4T_B | Mobile air conditioning in light and heavy duty trucks, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | TRA_RD_LD4T_S | Mobile air conditioning in light and heavy duty trucks, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HFO | Alternative refrigerant: HFO-1234yf |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | AERO | Aerosols | ALT_HFC | Alternative propellant HFC-152a |
| | | | ALT_HC | Alternative propellant: Propane (R290) |
| | | | ALT_HFO | Alternative propellant HFO-1234ze |
| | OC | Polyurethane one component foams | ALT_HFC | Alternative blowing agent: HFC-152a |
| | | | ALT_HC | Alternative blowing agent: c-pentane, iso-pentane, n-pentane, etc. |
| | | | ALT_CO2 | Alternative technology: CO2 and supercritical CO2 |
| | | | ALT_HFO | Alternative blowing agent: HFO-1234ze, HFO-1233zd, HFO-1336mzz |
| | OF | Other foams | ALT_HFC | Alternative blowing agent: HFC-152a |
| | | | ALT_HC | Alternative blowing agent: c-pentane, iso-pentane, n-pentane, etc. |
| | | | ALT_CO2 | Alternative technology: CO2 and supercritical CO2 |
| | | | ALT_HFO | Alternative blowing agent: HFO-1234ze, HFO-1233zd, HFO-1336mzz |
| | GSHP_B | Ground source heat pumps, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | ALT_HC | Alternative refrigerant: Propane (R290) |
| | | | ALT_HFC | Alternative HFC refrigerant: HFC-152a |
| | | | ALT_HFO | Alternative blowing agent: HFO-1234ze, HFO-1233zd, HFO-1336mzz |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | GSHP_S | Ground source heat pumps, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | ALT_HC | Alternative refrigerant: Propane (R290) |
| | | | ALT_HFC | Alternative HFC refrigerant: HFC-152a |
| | | | ALT_HFO | Alternative blowing agent: HFO-1234ze, HFO-1233zd, HFO-1336mzz |
| | | | ALT_CO2 | Alternative technology: pressurized $CO_2$ |
| | FEXT_B | Fire extinguishers, emissions banked in equipment | GP | Good practice: leakage control, improved components |
| | | | FK | Alternative agent: Fluoro-ketone (FK-5-1-12) |
| | FEXT_S | Fire extinguishers, emissions from scrapped equipment | GP | Good practice: end-of-life recollection |
| | | | FK | Alternative agent: Fluoro-ketone (FK-5-1-12) |
| | SOLV_PEM | Solvents | BAN | Ban of use |
| | HCFC22_E | HCFC-22 production for emissive use | INC | Post combustion of HFC-23 |
| | HCFC22_F | HCFC-22 production for feedstock use | INC | Post combustion of HFC-23 |
| PFCs | ALU_SWPB | Primary aluminium production using side-worked prebake technology (SWPB) | RETSWPB | Retrofitting SWPB technology |
| | | | CONVSWPB | Conversion of SWPB to PFPB technology |
| | | | NEWSWPB | Conversion to speculative new technology eliminating emissions[a] |
| | ALU_VSS | Primary aluminium production using vertical stud Söderberg technology (VSS) | RETVSS | Retrofitting VSS technology |
| | | | CONVVSS | Conversion of VSS to PFPB technology |
| | | | NEWVSS | Conversion to speculative new technology eliminating emissions[a] |
| | ALU_CWPB | Primary aluminium production using center-worked prebake technology (CWPB) | RETCWPB | Retrofitting CWPB |
| | | | CONVCWPB | Conversion of CWPB to PFPB technology |
| | | | NEWCWPB | Conversion to speculative new technology eliminating emissions[a] |
| | ALU_PFPB | Primary aluminium production using point-feeder prebake technology (PFPB) | NEWPFPB | Conversion to speculative new technology eliminating emissions[a] |
| | SEMICOND | Semiconductor industry | NF3 | Switch from PFC to $NF_3$ |
| $SF_6$ | HMVES | High -and mid voltage switches | GP | Good practice: leakage control and end-of-life recollection and recycling |
| | MAGNPR | Magnesium production and casting | ALT_MAGN | Alternative protection gas: $SO_2$ |
| | WIND_B | Soundproof windows | BAN | Ban of use |
| | SF6_OTH | Other $SF_6$ sources | BAN | Ban of use |

a New technologies refer to either of the following technologies mentioned in IEA's Energy Technology Perspectives 2010 (IEA/OECD 2010): wetted drained cathods, inert anodes, carbothermic reduction or kaolinite reduction.

Table 23: Current legislation regulating F-gas emissions in EU member states.

| Sector | Gas | Policy | Regional coverage | Policy description and implementation in GAINS |
|---|---|---|---|---|
| Industry | PFCs | EU ETS Directive (EC/29/2009): Primary aluminum production | EU-wide | Industry needs to aquire tradable emission permits under the EU emission trading system (EU-ETS). |
|  | PFCs | Voluntary agreement in semiconductor industry | EU-wide | Semiconductor producers to reduce PFC emissions by 2010 to a level at 10 percent of 1995 emissions. Accounted for in GAINS to the extent it is reflected in national emission inventories to the UNFCCC. |
| F-gases | HFCs, PFCs, SF6 | EU F-gas regulation (EC 517/2014) | EU-wide | Phase-down of F-gas sold on the market, banning of use in applications where alternatives to F-gases are readily available, and preventing emissions from existing use of F-gases through leakage control and end-of-life recovery. |
|  | HFCs | EU MAC Directive (EC 40/2006) | EU-wide | Mobile air conditioners: replacing the use of high GWP HFCs with cooling agents GWP100 < 150 in all new vehicle models placed on the market. |
|  | HFCs | EU Directive on end-of-life vehicles (EC 53/2000) | EU-wide | Scrapped mobile air conditioners: recovery and proper handling |
|  | HFCs, PFCs, SF$_6$ | National F-gas regulations more stringent than EU regulation | Austria ("HFKW-FKW-SF6-Verordnung"), Belgium (end-of-life regulation from 2005 for large-scale refrigeration), Denmark (deposit-refund scheme since 1992, tax since 2001 and ban on import, sale and use since 2002), Germany ("Chemikalien-Klimaschutzverordnung" specify maximum leakage rates), Netherlands ("STEK" since 1992), Sweden (envirionmental fees since 1998, specific regulation since 2007) |  |

Table 24: Summary of assumptions on the control adopted to meet implemented EU and national F-gas regulations.

| Gas | Sector | Control adopted in the Reference scenario 2020 to meet implemented F-gas regulations |
|---|---|---|
| HFCs | Aerosols | Alternative substance propane (HC-290) implemented to 66% from 2020 onwards in all EU-27 countries. The consumption of HFCs in metered dose inhalers (MDIs) is exempted according to Article 15 (2) of the F-gas Regulation. |
| | Commercial air conditioning -small | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance HFCs with low-GWP implemented to 20% in 2020 and 40% in 2030 in EU-27, except in the six member states with national targets/legislation (Austria, Belgium, Denmark, Germany, Netherlands and Sweden), where it is implemented to 30% in 2020 and extended to 50% in 2030. |
| | Commercial air conditioning -large | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance water (R-178) implemented to 20% in 2020 and to 60% in 2030 in EU-27 in large units. |
| | Commercial refrigeration -small | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance hydrocarbons (e.g., HC-290) implemented to 25% in 2020 and to 70% in 2030 in EU-27. |
| | | Alternative substances HFCs with low-GWP (e.g., HFC-32) implemented to 5% in 2020 and to 10% in 2030 in EU-27. |
| | Commercial refrigeration -large | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance pressurized CO2 (R-744) implemented to 25% in 2020 and to 70% in 2030 in EU-27. |
| | | Alternative HFC refrigerant (e.g., HFC-32) implemented to 5% in 2020 and to 10% in 2030 in EU-27. |
| | Domestic refrigeration | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance hydrocarbons (e.g., HC-600a) implemented to 40% in 2020 and to 100% in 2030 in EU-27. |
| | Fire extinguishers | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Fluoro-Ketons implemented to 37% in 2020, 79% in 2030 and 100% in 2050 in all EU-27 countries. |
| | Ground source heat pumps | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | HCFC-22 production | Post-incineration of HFC-23 is 100% implemented from 2005 onwards (except Greece but plant has been closed). |
| | Industrial refrigeration - small | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | Industrial refrigeration - large | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance ammonia (NH3) implemented to 20% in 2020, 50% in 2030 and 80% in 2050 in all EU-27 countries, except Denmark with a 100% implementation already from 2005 onward. |
| | | Alternative substance HFO-1234yf implemented to 5% in 2020 and 20% from 2030 onwards in all EU-27 countries, except Denmark with a 0% implementation. |
| | One component foams | Alternative substances propane, pressurized CO2, HFC-152a and HFO's (e.g. HFO-1234ze, HFO-1233zd, HFO-1336mzz) implemented (in equal proportions) to 60% in 2020 and to 100% from 2025 onward in all EU-27 countries. |
| | Other foams | Alternative substances propane, pressurized CO2, HFC-152a and HFO's (e.g. HFO-1234ze, HFO-1233zd, HFO-1336mzz) implemented (in equal proportions) to 60% in 2020 and to 100% from 2025 onward in all EU-27 countries. |
| | Residential air-conditioning | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance propane (HC-290) implemented to 5% in 2020, 15% in 2030 and 20% from 2035 onward in all EU-27 countries. |
| | | Alternative substance HFC-32 implemented to 20% in 2020, 60% in 2030 and 80% from 2050 onward in all EU-27 countries. |
| | Solvents | Alternative substance HFO's (e.g., HFO-1336mzz (Z), HFO-1233zd(E)) implemented to 100% from 2020 onward in all EU-27 countries. |
| | Mobile air-conditioning | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance HFO-1234yf implemented to 20% in 2020, extending to 90% in 2030 and 100% in 2040 in all EU-27 countries. |
| | Transport refrigeration | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | | Alternative substance pressurized CO2 implemented to 10% in 2020, extending to 40% in 2030 and 50% from 2040 onward in all EU-27 countries. |
| | | Alternative substance propane implemented to 5% in 2020, extending to 20% from 2030 onward in all EU-27 countries. |
| | | Alternative substance HFCs with low-GWP implemented to 5% in 2020, extending to 20% in 2030 and 25% from 2045 onward in all EU-27 countries. |
| PFCs | Primary Al production | Only point-feeder pre-bake (PFPB) technology in use from 2020 onwards. |
| | Semiconductor industry | Emissions consistent with national reporting to the UNFCCC (2019) |
| SF6 | High and mid- voltage switches | Good practices (GP) implemented to 100% in all years (except Bulgaria and Romania in 2005 and Croatia in 2005/10) |
| | Magnesium prod. and casting | Alternative gas SO2 implemented to 100% from 2015 onwards. |
| | Soundproof windows | Use of sound-proof windows with SF6 phase-out in 2030. |

## 5.2 HFC emission sources

### 5.2.1 Air conditioning and refrigeration

#### 5.2.1.1 Residential air-conditioning

To estimate emissions from stationary air conditioners in the residential sector, we apply a method similar to what has been used in a model described by McNeil and Letschert (2007). HFC use for air conditioning depends both on the average HFC consumption per household using air conditioning (kg HFC/unit) and on the fraction of households who own air conditioners (penetration).

$$HFC\ consumption\ =\ Households\ \times Penetration\ \times\ Average\ charge\ size \tag{37}$$

The number of households was calculated by dividing total population by average household size. Data and scenario values for average household sizes are taken from the UN Global Report on Human Settlements 2005 (UN-HABITAT 2005).

We assume that both energy consumption per appliance and the proportion of households owning air conditioners (penetration) depend on climate and income, being higher in warmer and richer places. Penetration in a certain region is formulated as a function of the climate maximum saturation for that region and of the percentage of the climate maximum saturation achieved at that time in the region (availability).

$$Penetration\ =\ Availability\ \times\ Climate\ Maximum\ Saturation \tag{38}$$

The climate maximum saturation is derived from the assumption that current penetration rates in the USA are the maximum for a climate with a given amount of cooling degree days (CDD's). The relationship between maximum saturation and CDD is exponential, as developed by Sailor and Pavlova (2003) and corrected to give a maximum of 100 percent by McNeil and Letschert (2007) whose equation we have used here. Availability of air conditioners as a function of income is assumed to develop along a logistic function, with a threshold point beyond which ownership increases rapidly. Using data on present day air conditioner penetration in various countries from McNeil and Letschert (2007) we find availability as a function of income

$$Availability = \frac{1}{1+e^{4.152}\times e^{-0.237\times Income/1000}} \tag{39}$$

where income is defined as GDP per capita in purchasing power parity (PPP) and converted to constant Euro 2010.

GDP and population data is taken from the GAINS model in consistency with the relevant external macroeconomic scenario, i.e., EC/DG ECFIN (2020). Data on cooling degree days along with the expected future increase in regional CDDs and household size is taken from IEA (2018) and UN-HABITAT (2005), respectively. Once the number of stationary air conditioners is estimated, the HFC consumption is estimated assuming the average size of each appliance is 2.62 kW (Adnot et al. 2003) and the average refrigerant charge is 0.25 kg/kW (UNEP 2011a). An annual leakage rate of 10 percent is assumed from equipment in use mainly due to losses during refill (Schwartz and Harnisch 2003). The emission factor has been estimated as a product of leakage rates and country/sector specific GWPs, taken from the national communication to the UNFCCC (2019). Figure 7 presents the share of HFC's used in stationary air-conditioning sector as reported by EU-27 countries for 2015 (UNFCCC, 2019). For countries not reporting

HFC consumption in this sector, the German shares have been adopted as default. At the end-of-life the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery through recycling or destruction.

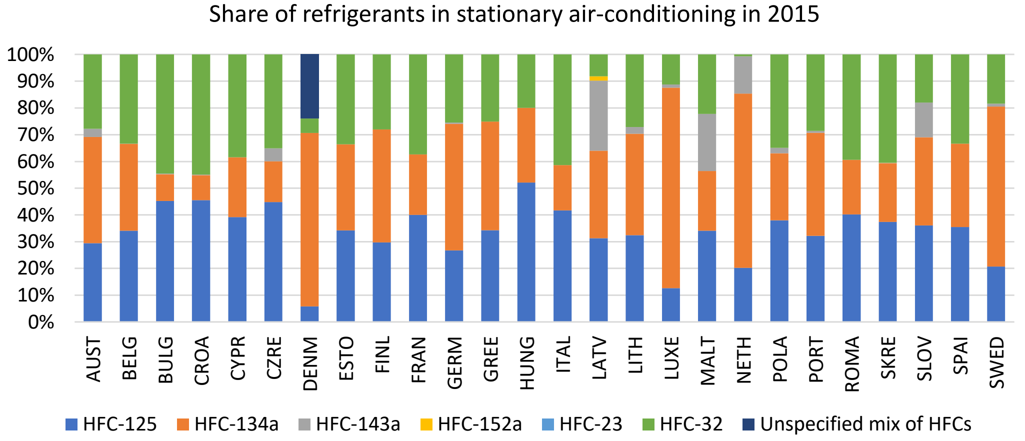


Figure 7: Share of refrigerants in stationary air-conditioning sector in 2015 as reported to UNFCCC (2019).

The control options available for this source are different good practise options including leakage control, improved components and end-of-life recollection. These options are assumed to remove 30 percent of emissions banked in equipment in use and almost 90 percent of scrapping emissions. Good practice options are being implemented in the EU as part of the different regulations controlling F-gases (see Table 23). In countries with no prior national F-gas regulation, full adoption of good practice options is assumed from 2015 onwards. For substantial further emission reductions, the use of HFC-410A (50% HFC-125 and 50% HFC-32) and HFC-134a ($GWP_{100}$=1300) using IPCC (2013) and other high GWP blends need to be replaced by an alternative low GWP refrigerant such as HFC-32 ($GWP_{100}$=677), propane ($GWP_{100}$=3) or pressurized $CO_2$ ($GWP_{100}$=1) (Table 22).

A problem with the HFC alternatives with low GWPs is that they often suffer from poorer performance, e.g., in terms of flammability or efficiency (IPCC/TEAP, 2005; USEPA, 2008b; Minjares, 2011). In recent years, companies like Honeywell© and Dupont© have developed and marketed alternative substances with better performances and very short lifetimes of less than a few months. These are known as hydrofluoroolefins (HFOs) (or unsaturated HFCs). E.g., HFO-1234ze with a $GWP_{100}$ of less than 1 can be used in foam products and HFO-1234yf with a $GWP_{100}$ of less than 1 can be used in mobile air-conditioners (IPCC, 2013). The suitability of these substances for stationary air conditioners has not yet been confirmed and they are therefore currently not applied in GAINS for this source.

Another option would be to use other non-HFC substances with low or zero GWP like hydrocarbons, $CO_2$, dimethyl ether and other diverse substances used in various types of foam products, refrigeration, air-conditioning and fire protection systems. In view of the new EU F-gas regulation (EC517/2014), HFC's with a GWP above 750 will be banned from 2025 in 'single split' systems that contain less than 3 kg of refrigerant. Single split systems involve one cooling coil connected to a remote condensing unit. There are no bans for larger air-conditioning or heat pump systems, e.g. chillers or larger split systems.

#### 5.2.1.2 Commercial air-conditioning

The GAINS model stores data on commercial floor space area for Annex-1 countries (Cofala et al., 2009). The primary data source for this data is the PRIMES (2015) model. For year 2005 the data on commercial floor space area was correlated with GDP/capita as illustrated in Figure 8.

Fitting a linear trend line, the following relationship was retrieved:

$$\left[\left(\frac{Commercial\ floor\ space}{Capita}\right) = 0.0003\left(\frac{GDP}{Capita}\right) + 7.1984\right] \qquad (R^2 = 0.6737) \qquad (40)$$

Using GDP per capita as driver, projections for future growth in commercial floor space area were obtained for each country. To estimate the HFC consumption in commercial air conditioning, a sector specific HFC consumption value of 0.02 kg/m$^2$ was applied.

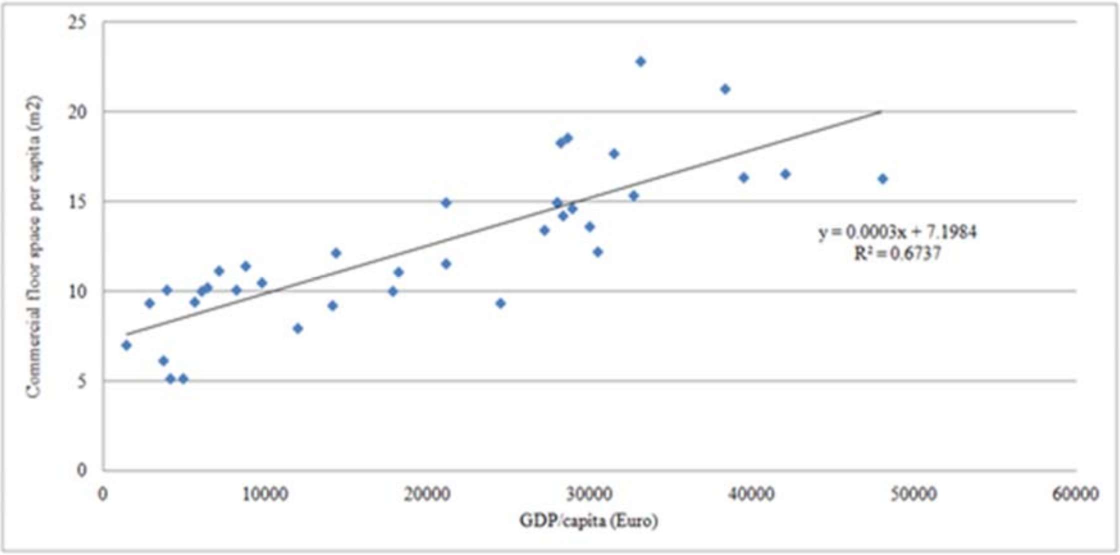


Figure 8: Variation of commercial floor space per capita w.r.t. GDP per capita in year 2005. Source: PRIMES model (2015).

An annual leakage rate of 10 percent is assumed from equipment in use mainly due to losses during refill (Schwartz and Harnisch, 2003). The share of HFC's used in stationary air-conditioning sector as reported by EU-27 countries for 2015 (UNFCCC, 2019) are presented in Figure 7. As mentioned in the previous section, German shares have been adopted as default for countries not reporting HFC consumption in this sector.

Figure 9 shows the default GAINS leakage rate applied to all EU-27 countries in comparison to the different leakage rates reported by member states to UNFCCC (2019) for2015. At the end-of-life the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery, recycling or destruction.

Control options available for this source are similar to the options discussed for residential air conditioning (previous section).

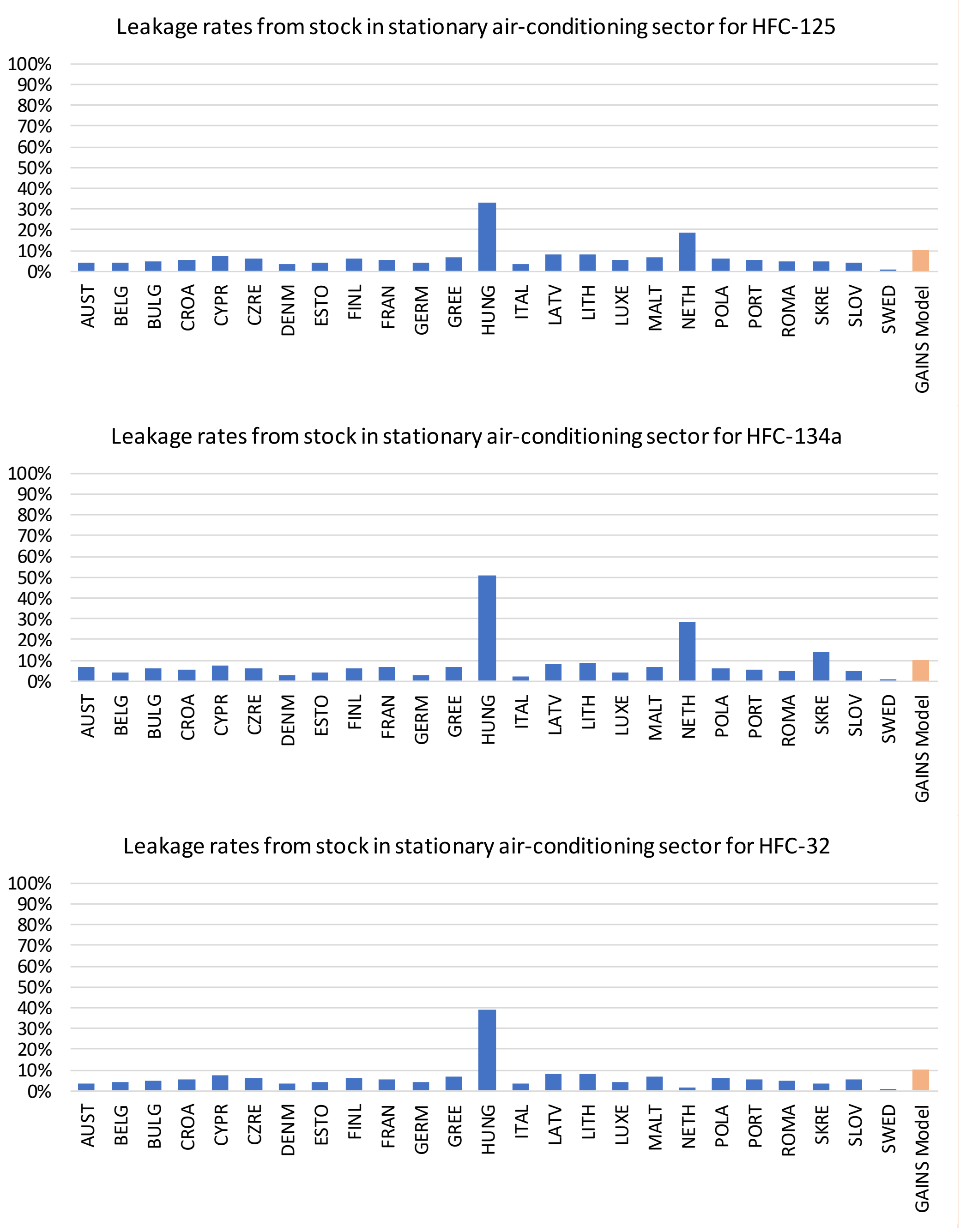


Figure 9: Leakage rates from stock in stationary (residential and commercial) air-conditioning in 2015 for different HFCs as reported to UNFCCC (2019) and adopted as assumptions in GAINS.

#### 5.2.1.3 Domestic refrigeration

The activity data for refrigeration in the domestic sector is number of refrigerators owned by households. The rate of ownership of refrigerator(s) per household is derived using a function estimated by the PAMS model (2012). The general form of the function for the rate of refrigerator ownership per household is given by:

$$Sat_{DOM} = (K \times I_t)^{\lambda_a} \times \left[1 - e^{-\left(bE_t{}^{\lambda_b} + cU_t{}^{\lambda_c}\right)}\right]^a \tag{41}$$

where $Sat_{DOM}$ represents the saturation (rate) of domestic refrigerator ownership, $I$ is the monthly household income given by GDP per household in the country, $U$ is the national urbanization rate, $E$ is the national electrification rate, and $t$ is the year of the projected saturation. The econometric parameter estimates from the PAMS model were applied to derive the rate of refrigerator ownership per household in GAINS. The number of refrigerators in a country was calculated by multiplying the ownership rate by the number of households in a country (UN-HABITAT, 2005). GDP and population data is taken from the GAINS model in consistency with the relevant external macroeconomic scenario, i.e., EC/DG ECFIN (2020) whereas data on household size is taken from UN-HABITAT (2005). Growth in number of refrigerators is driven by population growth and trends in appliance ownership as estimated above. Once the number of refrigerators is estimated, an average refrigerant charge of 0.125 kg HFC per unit (Schwartz et al., 2011) is used to estimate the HFC consumption in domestic refrigerators.

As domestic refrigerators are hermetic there is no risk of leakage during use, but there is a risk of emission release during the scrapping phase. At the end-of-life, the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery with recycling or destruction.

The control option available for this source is good practice during end-of-life scrapping, which is assumed to remove 85 percent of emissions. The option is already in place in the EU through the F-gas Regulation 2006 (see Table 23). For substantial further emission reductions, the use of HFC-134a ($GWP_{100}$=1300) need to be replaced by alternative low GWP refrigerants such as iso-butane ($GWP_{100}$=3). HFCs with GWPs above 150 are banned in domestic fridges and freezers as per new F-gas regulation (EC 517/2014).

#### 5.2.1.4 Commercial refrigeration

Commercial refrigeration includes refrigerated equipment found in supermarkets, convenience stores, restaurants, and other food service establishments (Girotto et al. 2004). Equipment in this end-use typically lasts approximately 15–20 years. The commercial refrigeration sector accounts for approximately 32 percent of global HFC consumption, or 40 percent of HFC consumption in the refrigeration/AC sector (USEPA 2010c).

Starting point for the estimation of emissions from commercial refrigeration in EU-27 in GAINS is the HFC consumption reported by member states to the UNFCCC (2019) for this sector for the years 2005, 2010 and 2015. As we are not able to fully explain the variations in the reported consumption, e.g., by having access to information on consumption patterns for refrigerated goods, we adopt HFC consumption as reported. Projections for future HFC consumption are driven by growth in service sector value added. For countries not reporting HFC consumption in this sector, the German consumption per service sector value added has been adopted as default. Projections for service sector value added are adopted from the macroeconomic scenario by EC/DG ECFIN (2020).

An annual leakage rate of 18 percent from equipment in use is applied consistently across all member states (Gschrey et al., 2011) since no evidence has been found available from direct measurements which would suggest that leakage rates differ substantially between EU countries for this source sector. The leakage from this source mainly reflects losses during refill (Schwartz and Harnisch 2003). Figure 10 presents the share of HFC's used in commercial refrigeration sector as reported by EU-27 countries for 2015 (UNFCCC, 2019). It is observed that HFC-134a (39%), HFC-143a (29%) and HFC-125 (28%) are mostly used for commercial refrigeration sector in EU.

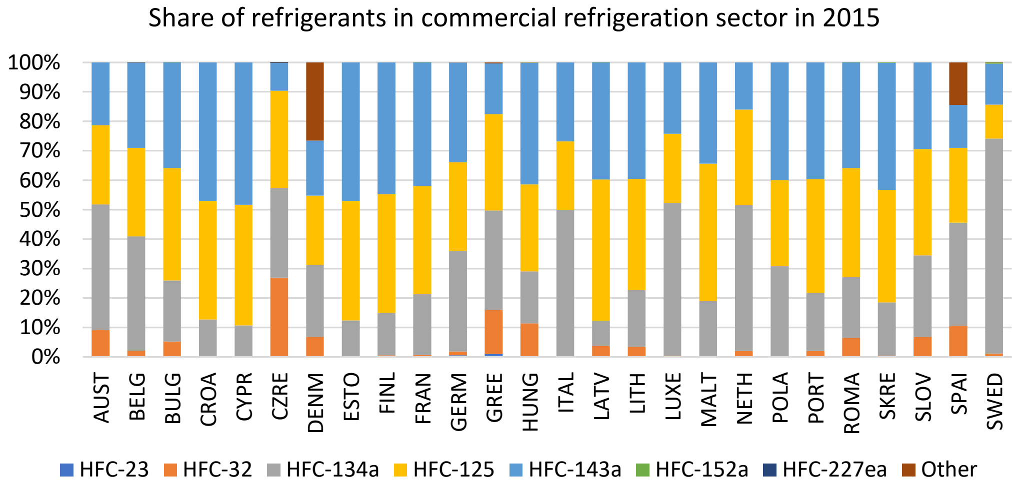
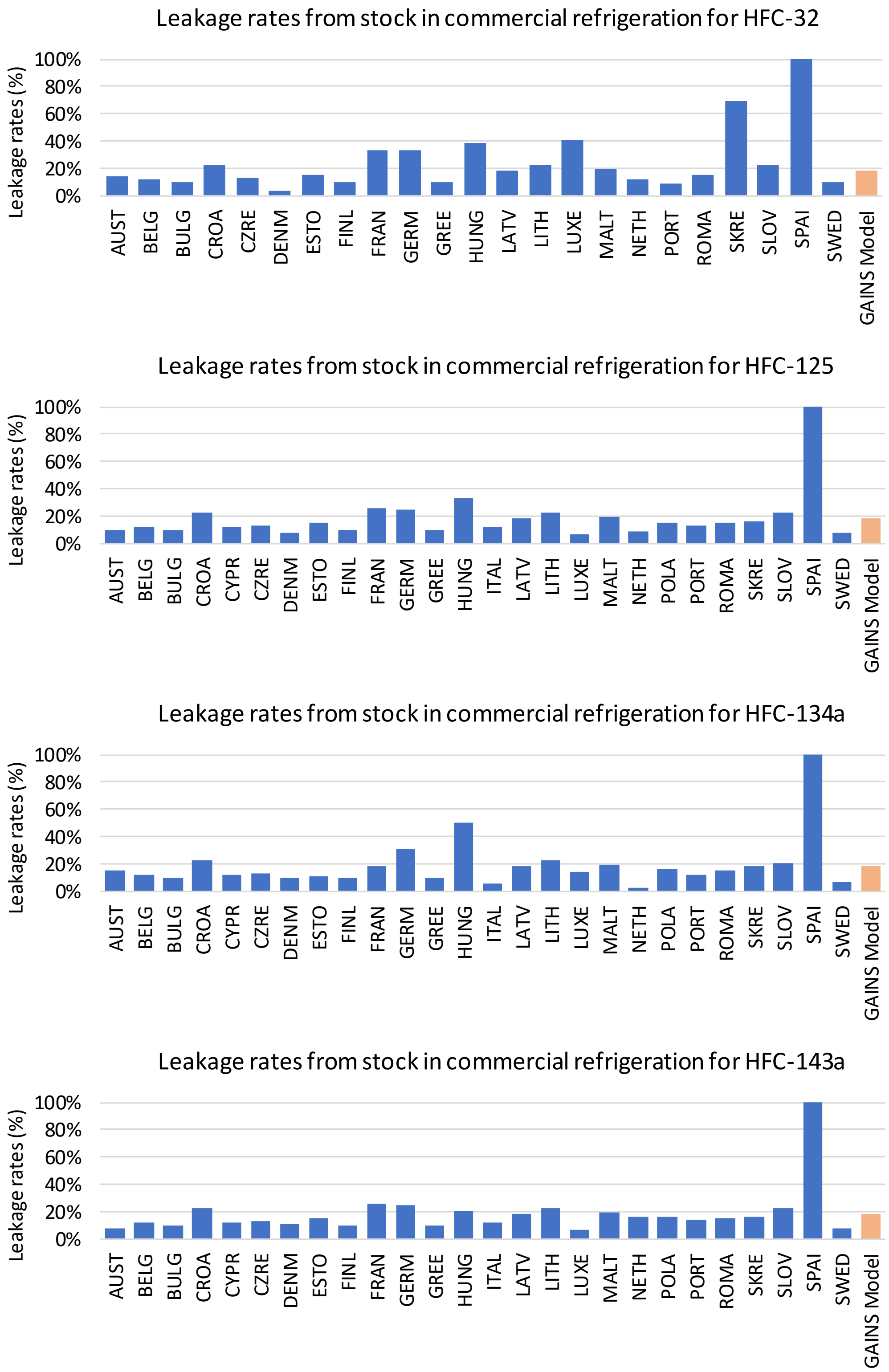


Figure 10: Share of refrigerants in commercial refrigeration sector in 2015 as reported to UNFCCC (2019).

Figure 11 shows the default GAINS leakage rate applied to all EU-27 countries in comparison to the different leakage rates reported by member states to UNFCCC (2019) for 2015. At the end-of-life the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery, recycling or destruction.

The control options available for this source are different good practise options including leakage control, improved components and end-of-life recollection. These options are assumed to remove 42 percent of emissions banked in equipment in use and almost 90 percent of scrapping emissions (Tohka, 2005). Good practice options are already implemented in the EU as part of the different regulations controlling F-gases (see Table 23). For substantial further emission reductions, the use of HFC-134a ($GWP_{100}$=1300), HFC-125 ($GWP_{100}$=3170), HFC-143a ($GWP_{100}$=4800) and other high GWP blends need to be replaced by alternative low GWP refrigerants such as HFC-152a ($GWP_{100}$=138), HC-290 ($GWP_{100}$=3) or pressurized $CO_2$ ($GWP_{100}$=1) using IPCC (2013). HFCs with GWPs of more than 2,500 will be banned in commercial refrigeration systems as per new F-gas regulation (EC 517/2014).


Leakage rates from stock in commercial refrigeration for HFC-32

Leakage rates from stock in commercial refrigeration for HFC-125

Leakage rates from stock in commercial refrigeration for HFC-134a

Leakage rates from stock in commercial refrigeration for HFC-143a

Figure 11: Leakage rates from stock in commercial refrigeration in 2015 for different HFCs as reported to UNFCCC (2019) and assumption adopted in GAINS.

#### 5.2.1.5 Industrial refrigeration

Food processing and cold storage is an important application of industrial refrigeration used for preservation and distribution of food while keeping nutrients intact. On a global scale this application is very significant in size and economic importance (Mohanraj et al. 2009). The application includes cold storage (at temperatures from −1°C to 10°C), freezing (−30°C to −35°C) and the long-term storage of frozen products (−20°C to −30°C). The preferred HFCs used are HFC-134a and HFC blends with a small temperature glide such as HFC-404A, HFC-507A and HFC-410A. Ammonia/$CO_2$ cascade systems are also being used, as are hydrocarbons as primary refrigerants in indirect systems (IPCC/TEAP 2005).

Starting point for the estimation of emissions from industrial refrigeration in EU-27 in GAINS is the HFC consumption reported for this source by member states to the UNFCCC for the years 2005, 2010 and 2015. As we are not able to explain the variations in the reported consumption, we adopt it as activity data as reported. Projections for future HFC consumption are driven by growth in value added for manufacturing industry. For countries not reporting HFC consumption in this sector, the German consumption per value added has been adopted as default. Projections for manufacturing industry value added are adopted from the macroeconomic scenario by EC/DG ECFIN (2020).

An annual leakage rate of 11 percent is applied consistently across member states (Gschrey et al., 2011) from equipment in use mainly due to losses during refill (Schwartz and Harnisch 2003). No evidence has been found available from direct measurements which would suggest that leakage rates differ substantially between EU countries for this source sector. Country/sector specific GWPs are taken from the national communication to the UNFCCC (2019). Figure 12 presents the share of HFC's used in industrial refrigeration sector as reported by EU-27 countries for 2015 (UNFCCC, 2019). It is observed that HFC-134a (41%), HFC-125 (31%) and HFC-143a (18%) are mostly used for industrial refrigeration sector in EU-27 countries. For countries not reporting HFC consumption in this sector, the German shares has been adopted as default.

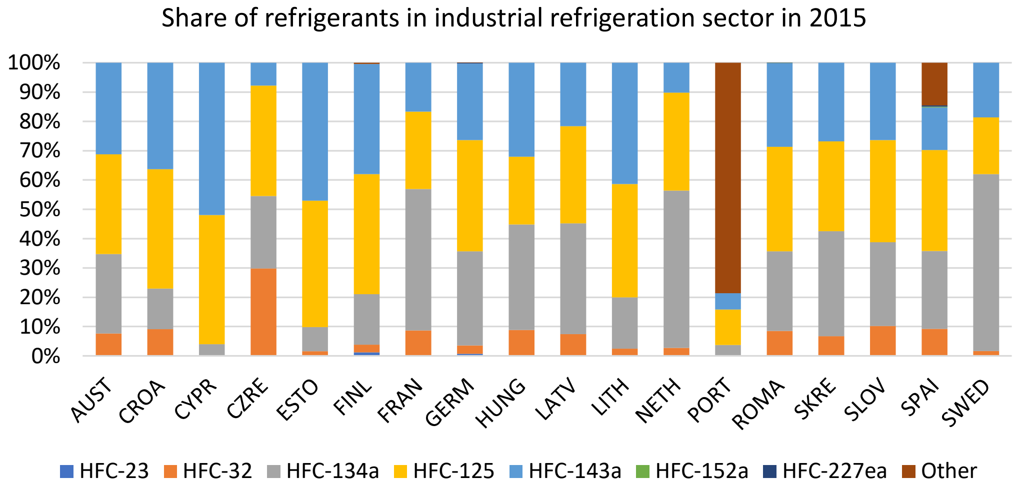


Figure 12: Share of HFC's in industrial refrigeration sector of EU countries in 2015 as reported to UNFCCC (2019).

Figure 13 shows the default GAINS leakage rate applied to all EU-27 countries in comparison to the different leakage rates reported by member states to UNFCCC (2019) for 20152015. At the end-of-life the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery with recycling or destruction.

The control options available for this source are different good practise options including leakage control, improved components and end-of-life recollection. These options are assumed to remove 42 percent of emissions banked in equipment in use and almost 90 percent of scrapping emissions (Tohka, 2005). Good practice options are already implemented in the EU as part of the different regulations controlling F-gases (see Table 23). For substantial further emission reductions, the use of HFC-134a ($GWP_{100}$=1300), HFC-125 ($GWP_{100}$=3170), HFC-143a ($GWP_{100}$=4800) and other high GWP blends need to be replaced by alternative low GWP refrigerants such as ammonia ($GWP_{100}$=0), HFC-32 ($GWP_{100}$=677) or pressurized $CO_2$ ($GWP_{100}$=1) (see Table 22).

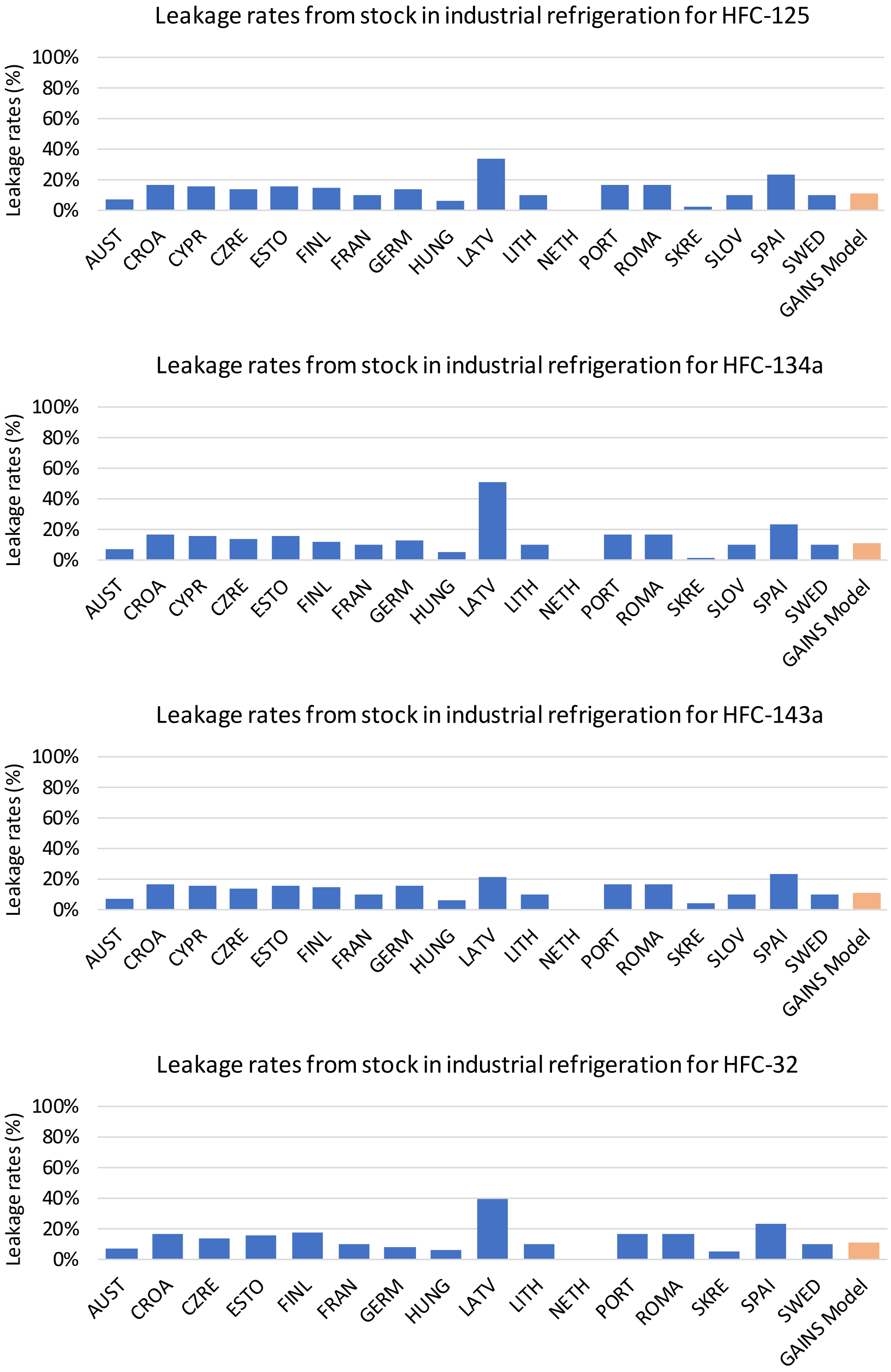


Figure 13: Leakage rates from stock in industrial refrigeration in 2015for different HFCs as reported to UNFCCC (2019) and assumption adopted in GAINS

#### 5.2.1.6 Refrigerated transport

Refrigerated road transport includes transportation of food products (fresh, frozen or chilled), pharmaceutical products, and plants/flowers. The type of vehicles used for such transportations are trailers, heavy and small trucks, and vans. Refrigerated road transport vehicles have different capacities; vans are typically below 3.5 tonnes, small trucks and trailers vary between 3.5 to 7.5 tonnes, and heavy trucks have a capacity of more than 7.5 tonnes.

Approximately 650000 refrigerated road vehicles were used within the EU during 2003-04 (Tassou et al. 2009; 2010) that has increased up to one million in 2014-15 (Dearman, 2015).These units predominantly use HFC-404A and HFC-410A as refrigerants. HFC-134a is also used for chilled distribution only vehicles. It is reported that the emission leakages from transport refrigeration systems are higher than those from stationary refrigeration because the former operate under more severe conditions (IIR 2003). The operating environment involves vibration, which will depend on road surface and a wide range of weather conditions and operating temperatures. Annual leakage figures reported are 10–37 percent of the refrigerant charge (IPCC/TEAP 2005). A study reported by Koehler et al. (2003), which assumed a 10 percent leakage rate showed the direct emissions (refrigerant leakage) from the refrigeration system to be 21 percent of indirect emissions (engine fuel consumption) for HFC-404A and 13 percent for HFC-410A.

Due to the country-specific variation in the amount of freight transported as well as the type of vehicles used (see UNECE 2010; Eurostat 2010; OECD 2010; USDOT 2010), GAINS derive HFC consumption per unit of freight transportation in 2005, 2010 and 2015 based on the consumption reported by countries for this source to the UNFCCC (2019). For countries not reporting HFC consumption specific for this sector, the rate reported for Germany (1.53, 2.03 and 1.47 kg HFC per unit GDP for 2005, 2010 and 2015 respectively) is adopted as default. Projections of HFC consumption in refrigerated transport have been assumed to follow proportionately the growth in GDP (EC/DG ECFIN, 2020).

An annual leakage rate of 25 percent from equipment in use is adopted consistently across all member states (Gschrey et al., 2011). This leakage is mainly due to losses during refill (Schwartz and Harnisch 2003). Figure 14 presents the share of HFC's used in transport refrigeration sector as reported by EU-27 countries for 2015 (UNFCCC, 2019). It is observed that HFC-134a (35%), HFC-125 (29%), HFC-143a (25%) and HFC-32 (11%) are mostly used for transport refrigeration sector in EU-27 countries.

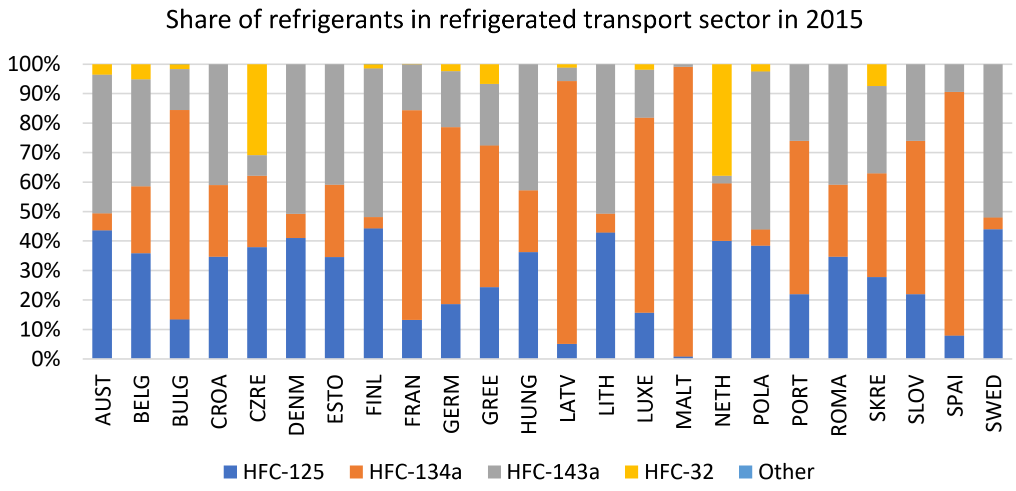
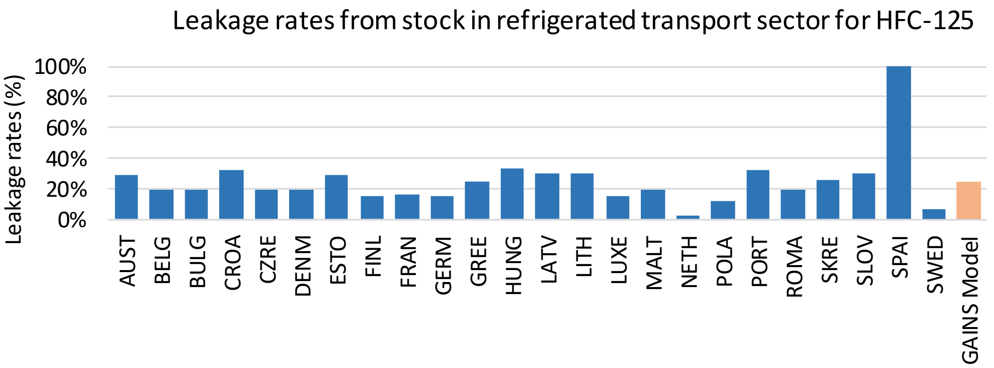
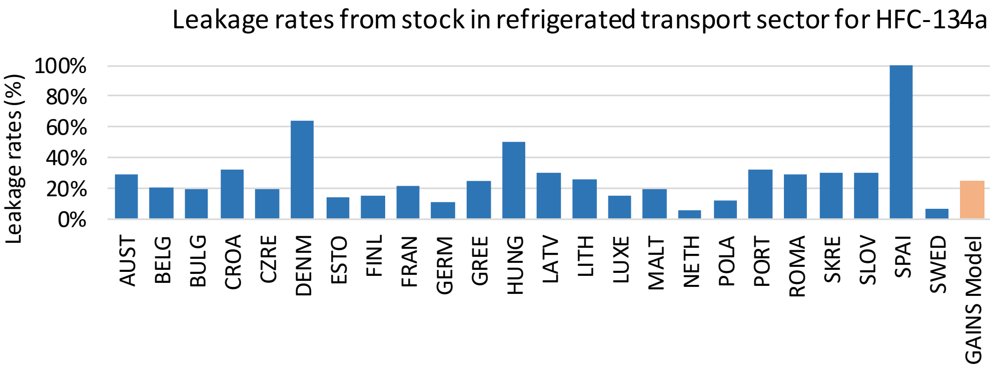


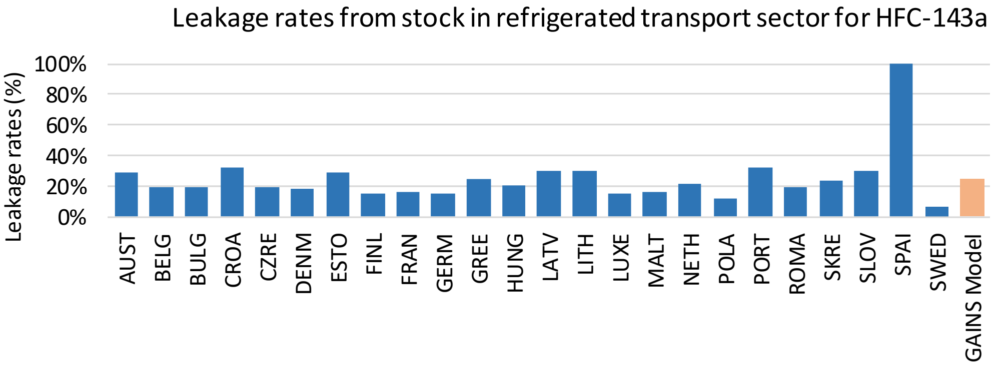
Figure 14: Share of HFC's in transport refrigeration sector of EU countries in 2015 as reported to UNFCCC (2019).

No evidence has been found available from direct measurements which would suggest that leakage rates differ substantially between EU countries for this source sector. Figure 15 shows the default GAINS leakage rate applied to all EU-27 countries in comparison to the different leakage rates reported by member states to UNFCCC (2019) for 2015. At the end-of-life the scrapped equipment is assumed to be fully loaded with refrigerant which needs recovery, recycling or destruction.

The control options available for this source are different good practise options including leakage control, improved components and end-of-life recollection. These options are assumed to remove 33 percent of emissions banked in equipment in use and 80 percent of scrapping emissions. The latter emissions are controlled in the EU through the end-of-life vehicles Directive (see Table 23). Control of leakage from equipment in use for refrigerated transport was not part of the EU F-gas regulation but assumed to be adopted with full effect from 2015 in Germany, Denmark and the Netherlands due to national F-gas regulations.  Further emission reductions from this source can be achieved through switches to alternative refrigerants like HFC-32 or pressurized $CO_2$ (see Table 22). HFCs with GWP of more than 2,500 will be banned in transport refrigeration systems as per new F-gas regulation (EC 517/2014).


Leakage rates from stock in refrigerated transport sector for HFC-125


Leakage rates from stock in refrigerated transport sector for HFC-134a


Leakage rates from stock in refrigerated transport sector for HFC-143a

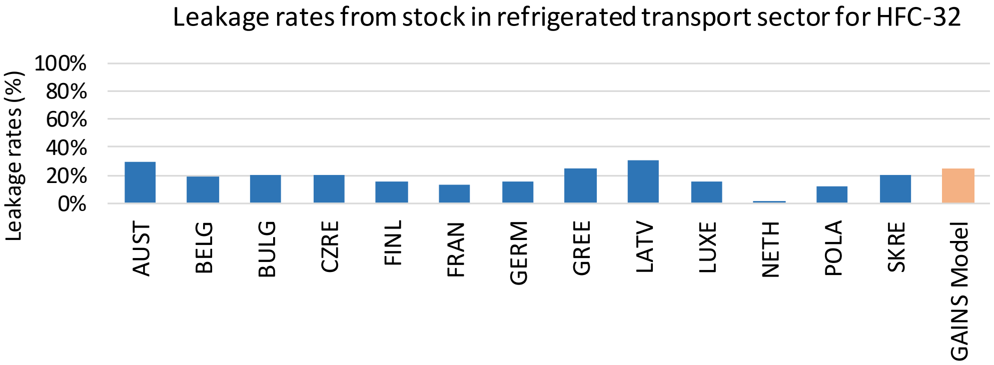

Leakage rates from stock in refrigerated transport sector for HFC-32

Figure 15: Leakage rates from stock in refrigerated transport in 2015 for different HFCs as reported to UNFCCC (2019) and assumption adopted in GAINS.

#### 5.2.1.7 Mobile air-conditioning

A major source of F-gas emissions from the transport sector is emissions from mobile air-conditioners (MAC). Air conditioning in cars became common in the United States in the 1960s. Mass installation in Europe and developing countries started only later, around 1995. The most commonly used refrigerant MACs in the EU is HFC-134a. The emissions of HFC-134a take place during accidents, through leakage and servicing and at disposal. Global recovery rates are generally low (DeAngelo et al. 2006), except for the EU where end-of-life recollection has been mandatory since 2000 due to the Directive on end-of-life vehicles (EC 53/2000).

In the GAINS model, emissions from MAC are accounted for in cars, light and heavy-duty trucks, and buses, as shown in Table 22. The number of vehicle types in different GAINS regions is extracted from the GAINS model and for EU member states consistent with transport fuel use in the PRIMES model (E3modelling, 2020). The penetration rates for air-conditioners in different vehicle types were derived from a literature survey (IPCC/TEAP 2005; Kanwar 2004; Hu et al. 2004; CSI 2009; Uherek et al. 2010). The adopted penetration rates for MAC in EU-27 in 2005 are applied consistently across all member states assuming 75 percent in cars and 60 percent in buses, light and heavy duty trucks. By 2020 all vehicles are assumed equipped with MAC. Using the average charge size for different vehicle types, the HFC consumption from MACs is estimated (Repice and Schultz 2004; Tohka 2005). Average charge sizes used are 0.67 kg for cars and light duty trucks, 1.34 kg for heavy duty trucks (Tohka, 2005) and 12 kg for buses (Schwarz et al., 2011). The leakage rate assumed from MAC in use is 10 percent and at the end-of-life the scrapped MAC is assumed to be fully loaded with coolant which needs recovery, recycling or destruction. Figure 16 shows the default GAINS leakage rate from equipment in use applied to all EU-27 countries in comparison to the different leakage rates reported by member states to UNFCCC (2019) for 2015. No evidence has been found available from direct measurements which would suggest that leakage rates differ substantially between EU countries for this source sector.

Following adherence to the MAC Directive (see Table 23), HFC-134a ($GWP_{100}$=1300) is expected to be replaced by a low GWP substance ($GWP_{100} < 150$) in all new models put on the market from January 2011 onwards. Moreover, new vehicles fitted with air conditioning cannot be manufactured with fluorinated greenhouse gases having GWPs greater than 150 from January 2017 onwards. There are a few possible alternatives to HFC-134a including replacement with $CO_2$, HFO-1234yf ($GWP_{100}$ is less than 1) or HFC-152a ($GWP_{100}$ is 138). The modelling of additional future potential for adoption of $CO_2$-based systems versus the use of HFO-1234yf in mobile air conditioners, depends in GAINS on the marginal cost of the respective alternatives when implemented in different types of vehicles.

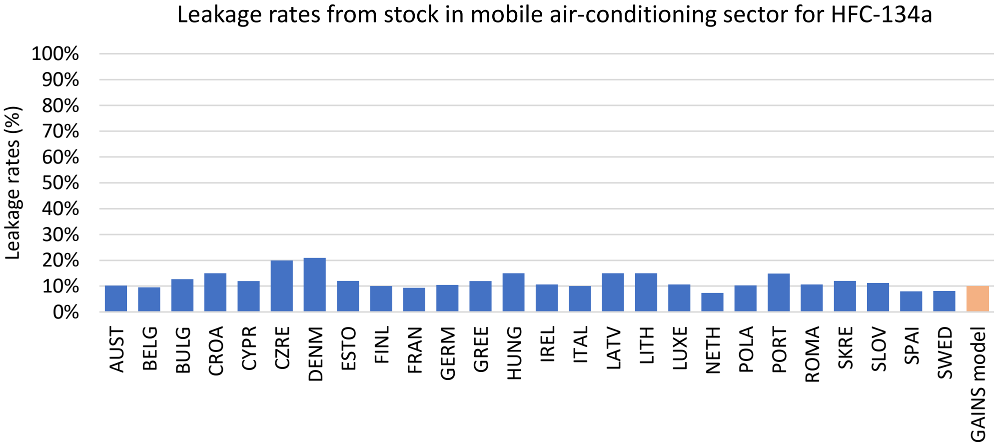


Figure 16: Leakage rates from stock in mobile air-conditioning in 2015 for HFC-134a as reported to UNFCCC (2019) and assumption adopted in GAINS

## 5.2.2 *Foams*

### 5.2.2.1 Polyurethane one component foams

Foams became a significant application for HFCs as part of the phasing-out of CFCs under the Montreal Protocol. HFCs are used as blowing agents in a solidifying matrix of a polymer (UNEP, 2006). The main application of polyurethane (PU) one component (OC) foam is to fill cavities and joints when installing inner fixtures in housing constructions. Since one component foams come in pressurized canisters and cylinders, they are also called aerosol foams. One component blowing agents are typically gaseous and function as propellant for the foam. They volatilize upon application, except for small residues that remain for at most one year in the hardened foam (Schwarz and Leisewitz 1999).

To estimate emissions from one component foams we adopt HFC consumption in one component foams as reported by member states to the UNFCCC (2019) for years 2005, 2010 and 2015. Projections of blowing agents for the one component foam sector follow growth in GDP (EC/DG ECFIN, 2020). The annual release of HFCs from foams is assumed 15 percent of the stock banked in foams. Figure 17 presents the share of HFC's used in foam sector as reported by EU-27 countries for 2015 (UNFCCC, 2019).

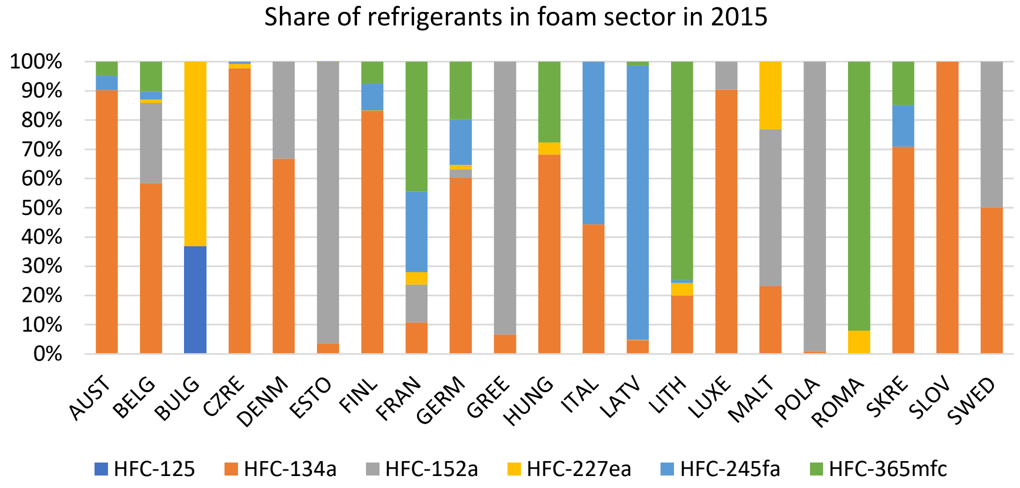


Figure 17: Share of HFC's in foam sector of EU countries in 2015. Source: UNFCCC (2019)

Emissions from foams can be controlled by replacing HFC-134a and other high GWP blends with alternative low GWP blowing agents, like hydrocarbons or $CO_2$. A recent mitigation option for applications where high pressure is essential is the replacement with HFO-1234ze ($GWP_{100}$ of less than 1). According to the manufacturer Honeywell© it is already available for use as blowing agent for one component foams (http://www51.honeywell.com/sm/lgwp-uk/applications.html) however the cost is unclear.

The F-gas Regulation (see Table 23) requires that all EU member states from 2008 stop using HFCs in one component foams unless this is required to meet national safety standards. According to Kaschl (2012) the most common current replacement options for HFCs in foams is $CO_2$. For some applications the performance of $CO_2$ in foam blowing is limited (UNEP/TEAP 2010). In GAINS the options considered available for replacement of HFCs in one component foams are $CO_2$, different hydrocarbons like propane, butane (i.e., i-butane, n-butane), pentane (i.e., isopentane, cyclopentane) and HFO-1234ze.

#### 5.2.2.2 Other foams

The sector for other foams (OF) in the GAINS model includes about 10 different polyurethane foam types (viz. PU appliances, PU/PIR/Phen laminates, PU disc panel, PU cont panel, PU blocks, PU spray, PU pipe, XPS) and extruded polystyrene (XPS). It is difficult to estimate product life emissions and lifetime of the foam product. End of life emissions depend greatly on the end of life treatment. If the product is land filled, the emission factor depends mainly on the properties of the plastic. If the product is recycled, all gases can be emitted into the atmosphere if fugitive emissions during the recycling process are not incinerated or collected. If the product is incinerated, the emission factor can be close to zero, depending on the incineration temperature. To estimate emissions from the other foam sector we adopt HFC consumption in other foams as reported by member states to the UNFCCC for years 2005, 2010 and 2015. Projections of refrigerants for one component foam sector follow GDP as taken from

(EC/DG ECFIN, 2020). The annual release of HFCs from foams is assumed 15 percent of the stock banked in foams. The share of HFCs used in foam sector as reported by EU-27 countries for 2015 (UNFCCC, 2019) is presented in Figure 17.

Emissions from foams can be controlled by replacing HFC-134a and other high GWP blends with an alternative blowing agent like $CO_2$ or hydrocarbons (propane, butane, pentane) or HFO-1234ze. According to Harvey (2007) a water/$CO_2$ mixture has been used in Europe (with a 10 to 20 percent market share by 2000) for solid PU in building applications. The use of HFO's (i.e., HFO-1234ze, HFO-1336mzz, HFO-1233zd) with a $GWP_{100}$ of less than 1 is also available for use as blowing agent for foams.

According to Kaschl (2012) 80 percent of XPS board foams in the EU use $CO_2$. However, $CO_2$ has some limitations with respect to thermal resistance and product thickness (UNEP/TEAP 2010). The remaining 20 percent will therefore need to use some other alternative, e.g., a mix of HFCs, HCs and water could be possible, but also HFO-1234ze is an interesting possible option (UNEP/TEAP 2010). In GAINS the options considered available for replacement of HFCs in one component foams are $CO_2$, different hydrocarbons like propane, butane and pentane, and HFO's (i.e., HFO-1234ze, HFO-1336mzz, HFO-1233zd).

All F-gases with GWPs of more than 150 will be banned as the refrigerant or foam blowing agent in any hermetically sealed system. This will mainly affect the use of HFC-134a as a refrigerant and HFCs-245fa and HFC-365mfc as foam blowing agents (EC 517/2014). From 2020, HFCs with GWPs of more than 150 will be banned in extruded polystyrene foam (XPS). From 2023 HFCs with GWPs of more than 150 will be banned in all other foams, including polyurethane.

### 5.2.3 Aerosols

HFC is used as propellant for aerosols released from cans and metered dose inhalers, e.g., medical asthma inhalers. In the EU, the use of HFCs as propellant for aerosols in all applications for entertainment and decorative purposes has been prohibited since 2008 (see Table 23). From 2018 HFCs with a global warming potential above 150 will be banned in technical aerosols (EC 517/2014). The release from this source is therefore mainly from medical dose inhalers (MDIs), where high pressure is essential and the approval of new medical drugs is very expensive and time-consuming (UNEP/TEAP 2010). In 2015, approximately 34% HFC-134 reported for the aerosol sector was used in MDIs (UNFCCC, 2019) We assume that for the EU, the current use of HFCs in MDIs is limited to severe cases and that the primary gases used are HFC-134a with some use of HFC-152a (Schwarz et al., 2011). To estimate HFC consumption in aerosols, we adopt the HFC consumption reported by member states to the UNFCCC for years 2005, 2010 and 2015 (UNFCCC, 2019). For countries not reporting HFC consumption in this sector, the German consumption per person has been adopted as default. Population growth is used as driver for future HFC use in aerosols. Figure 18 presents the share of HFC's used in aerosol sector as reported by EU-27 countries for 2015 (UNFCCC, 2019).

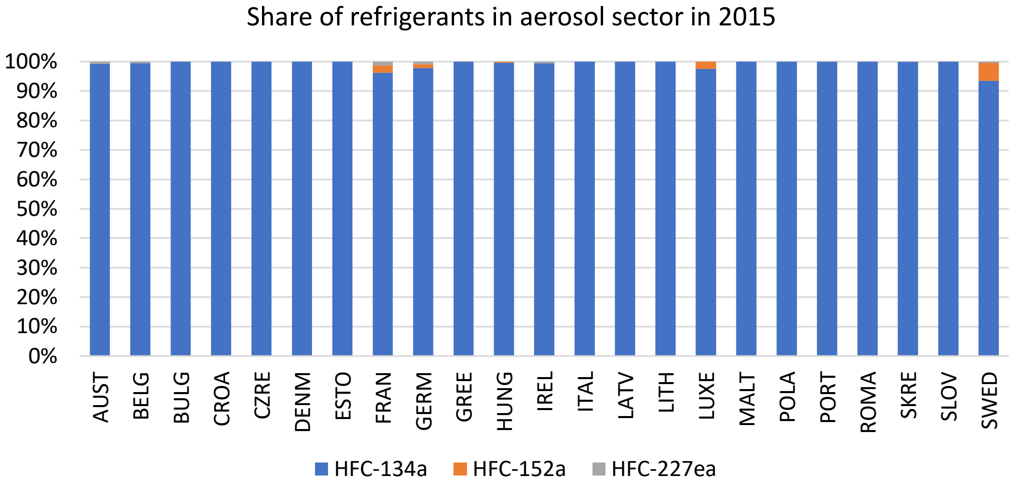


Figure 18: Share of HFC's in aerosol sector of EU countries in 2015

The primary alternatives to HFC-134a as propellant in medical dose inhalers are dry powder inhalers (DPI) or HFC-152a, which has a $GWP_{100}$ of about 140. The relative cost of these options is similar to the cost of MDIs in developed countries (UNEP/TEAP 2010), however, for medical reasons MDIs are still preferred in severe cases.

For severe cases, where high pressure is essential, there is the option to replace HFC-134a with HFO-1234ze ($GWP_{100}$ of less than 1), which according to the manufacturer Honeywell© is already available for use as propellant for aerosols (http://www51.honeywell.com/sm/lgwp-uk/applications.html). As no cost information is available for this option, GAINS assumes that the cost is about 50 percent higher than the cost of using HFCs.

### *5.2.4 HCFC-22 production*

HCFC-22 (chlorodifluoromethane) is a gas used in refrigeration and air-conditioning systems, in foam manufacturing as a blend component of blowing agents, and in the manufacturing of synthetic polymers. Since it is an ozone-depleting substance its release is regulated in the Montreal Protocol (MP). The phase-out schedule of HCFCs in the latest revision of the MP (UNEP, 2007) is presented in Table 25 for Article 5 (mostly developing countries) and non-Article 5 (mostly industrialized countries) countries. In addition to the phase-out of the use of HCFCs, the MP also requires the production and sales of HCFC-22 for emissive use to end completely after 2040.

In contrast to production of HCFC-22 for emissive use, the production and use of HCFC-22 as feedstock in industry is not regulated in the MP as it does not contribute to emissions of HFC-23. Production of HCFC-22 is however a source of HFC-23 emissions, which is a strong greenhouse gas with $GWP_{100}$ of 12400 times that of $CO_2$ (IPCC, 2013).

Table 25: Phase-out schedule of HCFCs for emissive use in the Montreal protocol (UNEP 2007).

| | **Article 5 (developing countries)** | | **non-Article 5 (developed countries)** |
|---|---|---|---|
| | **Pre 2007 revision of MP** | **Post 2007 revision of MP** | **Pre- and post 2007 revision of MP** |
| **1996** | | | freeze in emissions |
| **2004** | | | -35% |
| **2010** | | | -75% |
| **2013** | | freeze at average of 2009 & 2010 levels | |
| **2015** | freeze in emissions | -10% | -90% |
| **2020** | -35% | -35% | -99.50% |
| **2025** | | -67.5% | |
| **2030** | | -97.5% | -100% |
| **2035** | | | |
| **2040** | -100% | -100% | |

To calculate HFC-23 emissions from HCFC-22 production in the EU, GAINS applies an IPCC default emission factor of 3 percent related to the volume of HCFC-22 production for emissive (HCFC22_E) and feedstock (HCFC22_F) applications (IPCC, 2005). Activity data are based on reported production levels for historic years (UNEP, 2011) and UNEP's phase out schedule for HCFC products for future years (UNEP, 2007). Projections of HCFC-22 production for feedstock use are assumed to grow proportionately with value added in manufacturing industry with a levelling off after 2030 due to an assumed saturation in demand.

HFC-23 emissions from HCFC-22 production can be almost eliminated through post combustion during which HFC-23 is oxidized to carbon dioxide, hydrogen fluoride (HF) and water. The marginal abatement cost for destruction of HFC-23 emissions from HCFC-22 production is very low, less than 1 Euro/t$CO_2$eq (Schneider 2011; IPCC/TEAP 2005).

HCFC-22 production in the EU is recorded in the emission inventories submitted to the UNFCCC and takes place in France, Germany, Greece, Italy, Netherlands and United Kingdom. All countries except Greece are in the baseline scenario assumed to have full post-incineration of emissions in place already in 2005. Greece has full control from 2010 onwards.

### 5.2.5 Ground source heat pumps

Geothermal heat pumps or ground source heat pumps (GSHP) are systems combining a heat pump with a ground heat exchanger (closed loop systems) or being fed by ground water from a well (open loop systems). The earth is used as a heat source when operated in heating mode, with a fluid as the medium which transfers the heat from the earth to the evaporator of the heat pump, thus utilizing geothermal energy (Sanner et al., 2003). In cooling mode, heat pumps use the earth as a heat sink. With borehole heat exchangers (BHE), geothermal heat pumps can offer both heating and cooling at virtually any location, with great flexibility to meet demands.

The growth in Europe's GSHP systems has been accelerated by national policies[4] stimulating installation, e.g., through subsidies, efficiency standards to new buildings and heating demand

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[4] Heat pumps are considered a renewable energy technology in EU, where they are expected to account for between 5% and 20% of the EU's renewable energy target for 2020. For example, the revised National

mandates for heat pumps (EurObserv'ER, 2009). Many European countries have identified barriers that mirror those seen in the United States, namely higher investment costs, lack of knowledge and awareness among end users, and underdeveloped institutional and financial support (EHPA, 2008). In the EU, Sweden (>320,000) and Germany (>150,000) show the highest absolute numbers of GSHPs as shown in Figure 19.

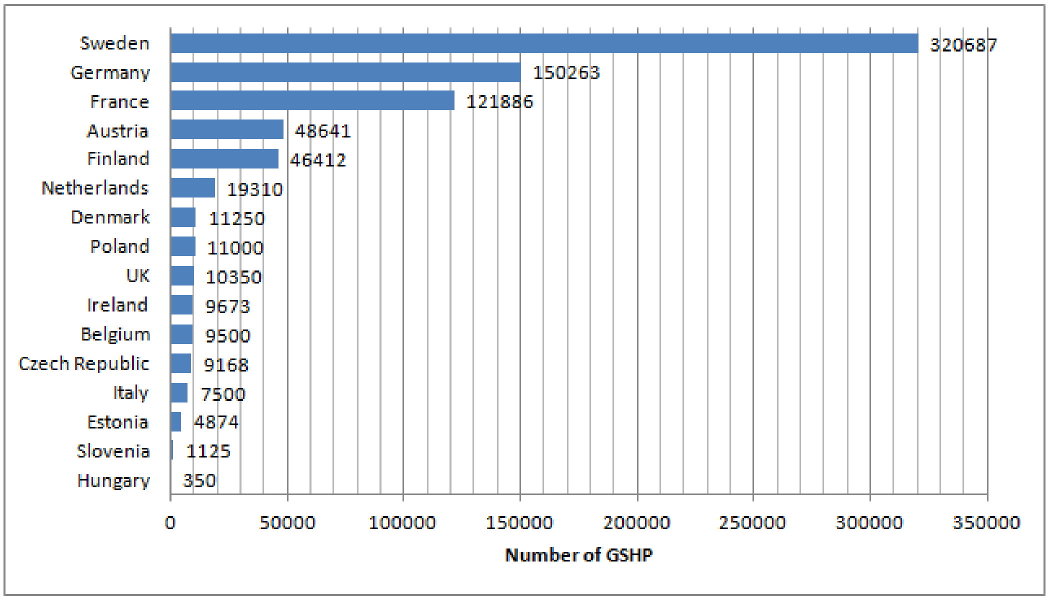

Figure 19: Number of existing GSHPs (with more than 10 MW heat capacity) in 2008. Source: EGEC (2009).

Lund et al. (2016) estimated installed capacity of direct use geothermal in 2014 approximately at 71 GWth, distributed in 82 countries. The previous estimates from the same group estimated installed capacity of direct use geothermal in 2009 at 51 GWth, distributed in 78 countries (Lund et al., 2011), while Goldstein et al. (2011) estimated direct use at 60 GWth at the end of 2009.

Direct use (ranging from 60 to 120°C) by type and relative estimates as given by Lund et al. (2016) were space heating (10.7%), greenhouse heating (2.8%), aquaculture pond heating (1.0%), agricultural drying (0.2%), industrial uses (0.9%), bathing and swimming (12.9%), cooling/snow melting (0.5%), and others (0.1%).  GSHP contributed to 71% (50.3 GWth) of the global installed geothermal heating capacity in 2014 and is the fastest growing of all forms of geothermal direct use since 1995 (Rybach 2005; Blum et al., 2010; Thorsteinsson and Tester, 2010; Lund et al. 2011; Schimschar et al. 2011; Ogola et al., 2012). Although, most of the installations occur in North America, Europe, and China, the number of countries with installation increased from 33 in 2005, to 43 in 2010, and to 48 in 2015. The equivalent number of installed 12 kW units (typical of US and Western European homes) is approximately 4.18 million (Lund et al., 2016).

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Renewable Energy Action Plan (NREAP) of Malta listed that 1.6% of heat pumps is required in order to reach the RES target by technology.

To estimate HFC consumption in the GSHP sector in the GAINS model we have taken information about the GSHP installed capacity from Lund et al. (2005, 2011, 2016), EurObserv'ER (2009), EHPA (2010) and Bayer et al. (2012). For projections, it is assumed that the annual growth in GSHPs using HFCs follows the growth in GDP (EC/DG ECFIN, 2020). Based on available technical information, the current fluid/refrigerant used in GSHP is most likely HFC-410a (IPCC/TEAP, 2005; Johnson, 2011). An average refrigerant charge of 0.22 kg HFC per kW installed capacity (Schwartz et al., 2011) is used to estimate the HFC consumption in the GSHP sector. Annual leakage from equipment is assumed at 2.5 percent per year. Emissions can be controlled through good practice options and switching to alternative substances. In the near future, the key alternatives for HFC-410A use in GSHP are assumed to be HC-290 direct, $CO_2$ and HFO-1234yf (Schwarz et al., 2011).

### 5.2.6 Other HFC emission sources

GAINS includes two additional HFC emission sources: fire extinguishers and solvents. Emissions from fire extinguishers are derived from the HFC consumption in fire extinguishers per unit GDP as reported for some EU countries to UNFCCC (2019) for years 2005, 2010 and 2015. We adopt reported values when available and derive a default factor per GDP for countries not reporting consumption from this source. Emissions from solvents are only included for those countries reporting emissions from this source to UNFCCC (2019) for years 2005, 2010 and 2015. For projections, emissions from both sources are assumed to grow proportional to GDP. Control options identified for these sources are good practice during scrapping and switching to alternative HFCs with low GWP. PFCs have been banned in fire protection systems since 2007. From 2016, HFC-23 are banned in new fire protection systems (EC 517/2014).

## 5.3 PFC emission sources

### 5.3.1 Primary aluminium production

Primary aluminium production has been identified as a major emission source of the two PFCs tetrafluoromethane ($CF_4$) (with $GWP_{100}$ = 6,630) and hexafluoroethane ($C_2F_6$) (with $GWP_{100}$ = 11,100 times that of $CO_2$). During normal operating conditions, an electrolytic cell used to produce aluminium does not generate measurable amounts of PFC. PFC is only produced during brief upset conditions known as "anode effects". These conditions occur when the level of aluminium oxide drops too low and the electrolytic bath itself begins to undergo electrolysis. Since the aluminium oxide level in the electrolytic bath cannot be directly measured, surrogates such as cell electrical resistance or voltage are most often used in modern facilities to ensure that the aluminium in the electrolytic bath is maintained at the correct level.

For EU-27, the GAINS model uses the production volumes of primary aluminium as the activity driver for calculating emissions from this source. Primary aluminium production data for historical years (2005, 2010 and 2015) and projections are taken from the PRIMES (E3modelling, 2020) baseline projection for the EU. Four different types of activities are distinguished based on the technology used; point-feeder prebake (PFPB), side-worked prebake

(SWPB), vertical stud söderberg (VSS), and center-worked prebake (CWPB) technology. As per the information provided by the MS, there was a significant reduction of primary aluminium production in Slovenia due to the economic crisis in 2010 as the new electrolysis unit operated less than half of its capacity. According to most recent data submitted in Slovenia's National Inventory Report 2015, primary aluminium production amounted to 84 kt in 2013, which is close to maximal capacity of 85 kt per year of the existing plant. Therefore, we have used primary aluminium production of 84 kt per year in Slovenia for 2015. Shares of different primary aluminium production technologies were adopted from the aluminium industry website and from national communications to the UNFCCC (2019). The latter source is also used for final verification of emissions. Emission factors depend on the production technology and on a number of site-specific conditions and are taken from Harnisch and Hendricks (2000).

Conversion of SWPB, VSS or CWPB technology to PFPB technology removes over 90 percent of PFC emissions, while retrofitting of the three technologies would remove about a quarter of emissions (Harnisch and Hendricks 2000). Data on mitigation costs is taken from the same source. As emissions from the primary aluminium production is regulated under the EU-ETS system, control options with marginal costs falling below the expected ETS carbon price in the PRIMES energy scenario are assumed adopted in the GAINS non-$CO_2$ baseline scenario. This means that with the natural turn-over of capital, all EU member states will have phased-in PFPB technology by 2020.

The development of inert anodes is sometimes promoted as a promising mitigation option, which could eliminate emissions of PFCs from the electrolysis process (Bernstein et al., 2007). In the Energy Technology Perspective (ETP) 2010 by the International Energy Agency (IEA/OECD, 2010), deployment of inert anode technologies is expected to start in 2015-2020 with full commercialization by 2030. If realized, inert anode technology would have significant energy, cost, productivity, and environmental benefits for the aluminium industry worldwide (Inert Anode Roadmap*,* 1998; RUSAL, 2010). The technology is expected to eliminate PFC emissions from primary aluminium production altogether. However, the commercial aspects of inert anodes have not yet been proven (Kvande and Drabløs, 2014). Despite promising initial results, the technology still needs further development before it can be introduced as a viable alternative to PFPB technology. In GAINS, inert anode technology is assumed available as a mitigation option from 2035 onwards, however, no adoption in the baseline scenario is assumed.

Figure 20 shows PFC emissions from primary aluminium production in EU-27 as estimated by the GAINS model and in comparison, with emissions reported to UNFCCC for years 2005, 2010 and 2015. While the recent version of the GAINS estimates uses IPCC/AR5 $GWP_{100}$ values however, for the comparison with UNFCCC (2019) we have converted GAINS emission estimates for PFC emissions (in kt CO2eq) from primary aluminium (Al) production into IPCC/AR4 $GWP_{100}$ values.

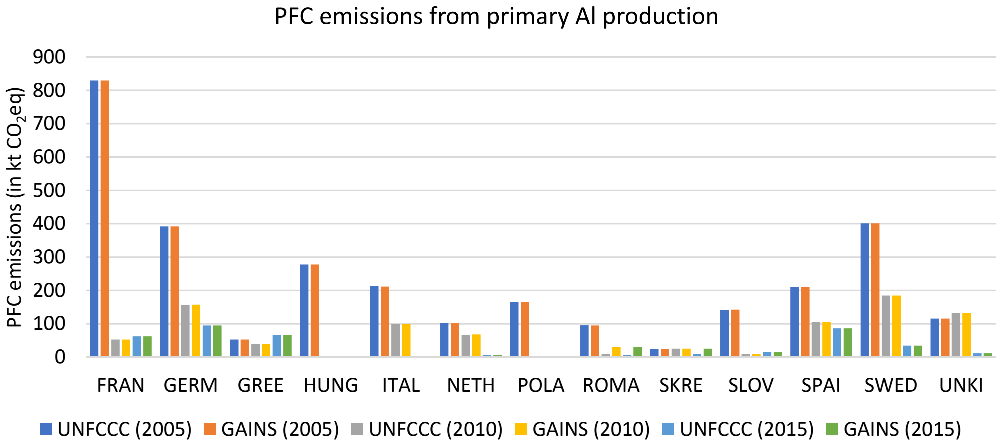

Figure 20: PFC emissions (in kt CO2eq) from primary aluminium (Al) production in the GAINS model and as reported to UNFCCC for years 2005, 2010 and 2015.

### 5.3.2 Semiconductor industry

The semiconductor industry uses HFC-23, $CF_4$, $C_2F_6$, octafluoropropane ($C_3F_8$), carbon tetrafluoride (c-$C_4F_8$), sulphur hexafluoride ($SF_6$) and nitrogen trifluoride ($NF_3$) in two production processes: plasma etching thin films (etch) and plasma cleaning chemical vapour deposition (CVD) tool chambers (IPCC, 2000a). Both the PFCs ($GWP_{100}$=6630) and $NF_3$ are potent greenhouse gases ($GWP_{100}$=16100). PFCs have been regulated under the Kyoto Protocol, while $NF_3$ is added to the 'basket' of six greenhouse gases (covered by the KP in the first commitment period) with effect from 2013 and the commencement of the second commitment period of the protocol. The semiconductor industry has been switching from PFCs to $NF_3$ as part of a voluntary agreement to reduce PFCs. The commitment has been to reduce PFC emissions to ten percent below the 1995 baseline level by 2010 (ESIA 2006; WSC 2008). In 2008 the industry was close to reaching this target (WSC 2008). Under well managed conditions and certainly in the EU and the US, a switch from PFC to $NF_3$ reduces the net global warming effect because about 98 percent of $NF_3$ is destroyed by industry in the process (UNFCCC 2012b). With a release of about 60 percent of the PFCs used in the industry, the switch to $NF_3$ contributes to a net reduction in greenhouse gas emissions by about 85 percent.

As PFC is only used by a few companies in a country and because the amount of PFC used allows deriving production volumes, data on the use is often confidential. The activity variable used in GAINS for this sector is the volume of PFC emissions as reported by member states to UNFCCC (2019). We use the reported emissions for the years 2005, 2010 and 2015 while future projections follow growth in value added for manufacturing industry.

The European semiconductor manufacturers have been part of the voluntary commitment to reduce PFC emissions from this source (ESIA, 2006). We assume that the reduction attained by the industry in 2005 will continue into the future. This corresponds to an application of control to 86 percent of the production from 2010 onwards. Costs for switching to $NF_3$ use were taken from Harnisch et al. (2000), Harnisch and Hendriks (2000) and Tohka (2005).

Figure 21 shows the GAINS model estimates of PFC emissions from semiconductor industry in EU member states. For the comparison with UNFCCC (2019) we have converted GAINS emissions estimates for PFC emissions from semiconductor industry into IPCC/AR4 $GWP_{100}$ values.

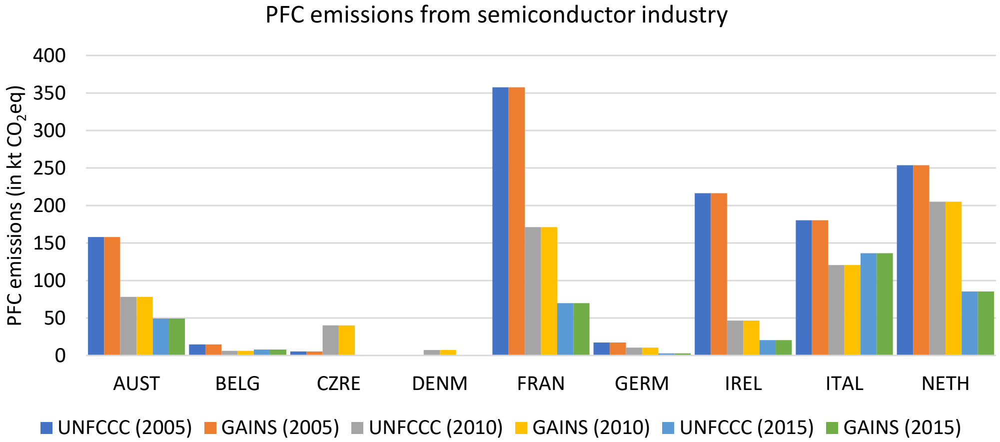

Figure 21: PFC emissions (in kt CO2eq) from semiconductor industry in the GAINS model and as reported to UNFCCC (2019) for years 2005, 2010 and 2015.

## 5.4 $SF_6$ emission sources

Sulphur hexafluoride ($SF_6$) emissions are released from high- and mid-voltage switches, magnesium production and casting and a variety of other applications e.g., soundproof windows or sports equipment. $SF_6$ has a very high global warming potential of 23,500 times that of $CO_2$ over a 100-year time horizon (IPCC, 2013).

### *5.4.1 High and mid voltage switches*

$SF_6$ is used as an electrical insulator in the transmission and distribution equipment of electric systems. Most of the $SF_6$ is stored in gas-insulated switchgears for high and mid-voltage electric networks. Emissions of $SF_6$ depend on the age of the gas insulated switchgear (GIS) since older models leak more than newer ones, as well as on the size of the transmission network and on recycling practises of the old equipment. The GAINS model uses electricity consumption as activity variable for this sector. The emission factor for $SF_6$ in electricity transmission per unit of electricity consumed is taken from the GHG inventory of California (CEPA, 2010) and applied in a consistent manner to all EU member states.

Suitable alternatives to $SF_6$ do not exist for these applications as the oil and compressed air systems, which were used previously, suffer from safety and reliability problems (AEAT, 2003). Emissions can be reduced through the adoption of recycling practices of used $SF_6$ switchgears. The EU F-gas Regulation requires end-of-life recollection and recycling from 2010 onwards. Full compliance with this regulation is assumed in GAINS to apply in all EU countries. Figure 22 shows GAINS emission estimates for this sector in comparison to emissions reported to UNFCCC (2019) for years 2005, 2010 and 2015. For the comparison with

UNFCCC (2019) we have converted GAINS $SF_6$ emission estimates from high and mid voltage electrical switches into IPCC/AR5 $GWP_{100}$ values.

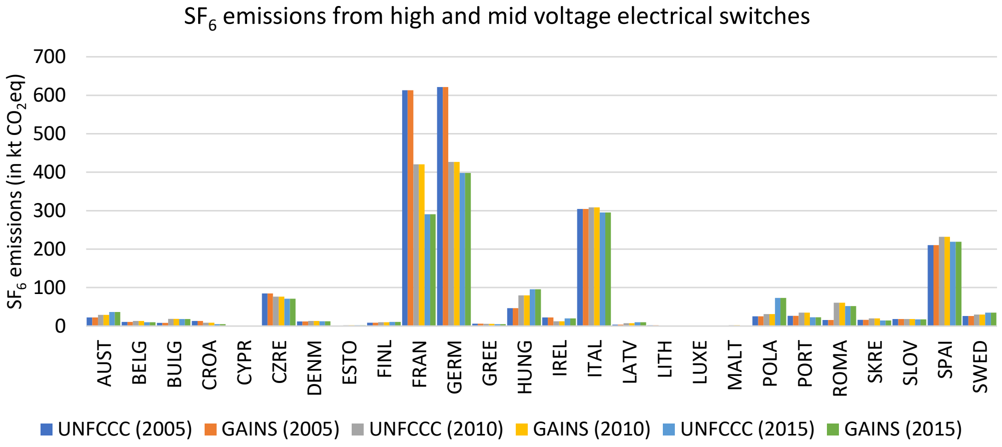

Figure 22: Historical $SF_6$ emissions from high and mid voltage electrical switches in the GAINS model and as reported to UNFCCC (2019).

### 5.4.2 Magnesium production and casting

Casting and production of primary and secondary magnesium are well known sources of atmospheric emissions of $SF_6$. The gas is used as a shielding gas in magnesium foundries to protect the molten magnesium from re-oxidizing whilst it is running to best casting ingots (IPCC, 2000b). GAINS use the reported $SF_6$ emissions for the years 2005, 2010 and 2015 while future projections follow growth in value added for manufacturing industry. Activity data on historic volumes of processed magnesium in the EU countries is taken from national communications to UNFCCC (2019). An emission factor of one kg $SF_6$ per ton processed metal is taken from Schwartz and Leisewitz (1999) and Tohka (2005). The EU's 2006 regulation only restricted use of $SF_6$ in die casting operations. The bans on the use of $SF_6$ in the recycling of magnesium die casting alloys are new to the 2014 regulation (DEFRA, 2014). $SF_6$ emissions in magnesium production and casting can be substituted using sulphur dioxide ($SO_2$) as alternative gas.

### 5.4.3 Soundproof windows

Some European countries used significant amounts of $SF_6$ in soundproof windows. From 2006, the F-gas Regulation (EC 842/2006) bans the use of $SF_6$ in soundproof windows. Soundproof windows have a relatively long life-time and it is therefore expected that the stock of $SF_6$ found in such windows in 2005 will be successively phased-out over a period of 25 years. The available stock of $SF_6$ in soundproof windows in 2005 in EU countries is estimated at 288 t $SF_6$ in Austria, 75 t $SF_6$ in Belgium, 86 t $SF_6$ in Denmark, 1764 t $SF_6$ in Germany, 1.78 t $SF_6$ in Slovenia, and 11.1 t $SF_6$ in Sweden. These estimates were verified in national communications between IIASA and country experts as part of review processes of baseline non-$CO_2$ GHGs organized by the European Commission in 2009 and 2012. With an assumed leakage/refill rate of 1 percent per year for windows still in use and a linear phase-out of emissions, annual emissions from this source until 2030 (when phase-out is completed) are estimated as:

$$E_t^{SF_6} = \frac{Stock_{2005}}{25} + Stock_t * 0.01, \tag{42}$$

where the first term represents the end-of-life emissions from soundproof windows scrapped in year t and the latter term represents the emission leakage from windows still in use.

No further mitigation options beyond the ban included in the F-gas Regulation are considered necessary to control emissions from soundproof windows.

### *5.4.4 Other $SF_6$ sources*

$SF_6$ has been used in tyres, sports equipment manufacturers in tennis balls and sport shoes. Activity data for these other sources of $SF_6$ emissions are taken from emissions reported by countries to the UNFCCC (2019). From 2006, the EU F-gas Regulation (EC 842/2006) bans the use of $SF_6$ in sports equipment and tyres. GAINS assumes that all EU countries adhere fully to this ban.

## 5.5 Costs of F-gas mitigation options

Costs of replacing HFCs with alternative cooling agents or through the use of alternative technological processes have recently been updated in the GAINS model as part of work to produce global mitigation scenarios for HFCs. The Supplement of Höglund-Isaksson et al. (2017) presents detailed information about HFC cost estimates and assumptions about cost parameter values. The principal source of information for HFC mitigation costs used here is Schwartz et al. (2011). Assumptions and references for mitigation costs for PFCs and $SF_6$ can be found in the Supplement of Purohit and Höglund-Isaksson (2017) and Höglund-Isaksson et al., (2017).

# 6 Comparison of GAINS bottom-up emission estimates to UNFCCC inventory

Table 26 presents the deviation of GAINS emission estimates for year 2015 in the EUCLIMIT5 Reference 2020 scenario from emissions reported by member states to the UNFCCC (December 2019). Note that for the comparison, emissions have been converted to kt $CO_2$eq applying Global warming potentials over 100 years ($GWP_{100}$) from IPCC AR5 (2013), while UNFCCC Common Reporting Format (CRF) apply $GWP_{100}$ from IPCC AR4 (2007) for the same conversion.

For EU-27, the GAINS estimate of total non-$CO_2$ GHG emissions in 2015 is 4% lower than the estimate reported to the UNFCCC, however, when excluding the impact from differences in $CH_4$ emissions estimates for the waste sector (as discussed in Section 3.4.1), the GAINS model estimate matches almost exactly emissions reported to the UNFCCC. For individual member states, deviations from reported emissions are larger, deviating within ten percent for all countries except Austria (-13%), Bulgaria (+14%) and Sweden (-11%). At the level of individual gases, estimated emissions may deviate by more than ten percent from reported emissions.

The principal difference between GAINS model estimates and those reported by member states to the UNFCCC is that GAINS applies a consistent methodology across all countries, whereas estimation methodologies applied by countries tend to differ in key assumptions. Another reason for differences is that member states sometimes report emissions for minor sources which the GAINS model structure does not capture, e.g., $CH_4$ emissions from thermal baths.

The GAINS emission estimates for $CH_4$ and $N_2O$ have been calibrated to total emissions by gas as reported by member states to the UNFCCC for years 2005, 2010 and 2015. The part of the $CH_4$ calibration factor for 2015 that is not reflecting the differences in emissions estimated for the waste sector, have been derived as calibration factor for future years. The derived calibration factor for 2015 is scaled proportionately with the future estimated development in total $CH_4$ emissions in order to avoid that a constant calibration factor becomes a dominant source in some countries as future emissions decline. Similarly for $N_2O$, the calibration factor is derived from the difference in the GAINS emission estimate for 2015 and total $N_2O$ emissions reported by member states to the UNFCCC, with a scaling for future years to the estimated development in future total $N_2O$ emissions. No calibration was conducted for F-gas emissions, because of large unexplained variations in leakage rates between countries and in the completeness of the sector emissions reported for F-gases.

Table 26: Deviation of GAINS EUCLIMIT5 (Reference 2020) from UNFCCC (2019).

| Country | Gas | UNFCCC (2019) GWP$_{100}$ IPCC AR4 year 2015 | UNFCCC (2019) GWP$_{100}$ IPCC AR5 year 2015 | GAINS REF2020 (before calibration) year 2015 | Difference GAINS vs UNFCCC for year 2015 | | Difference GAINS vs UNFCCC when excluding differences in CH4 from the waste sector | |
|---|---|---|---|---|---|---|---|---|
| | | kt CO2eq | kt CO2eq | kt CO2eq | kt CO2eq | % | kt CO2eq | % |
| Austria | CH4 | 6678 | 7479 | 7117 | 362 | 5% | -250 | -3% |
| | N2O | 3498 | 3111 | 3747 | -636 | -20% | -636 | -20% |
| | F-gases | 1988 | 1894 | 2691 | -702 | -37% | -702 | -37% |
| | **sum non-CO2** | **12164** | **12484** | **13555** | **-976** | **-8%** | **-1588** | **-13%** |
| Belgium | CH4 | 8099 | 9071 | 9522 | -451 | -5% | 116 | 1% |
| | N2O | 5991 | 5328 | 6196 | -868 | -16% | -868 | -16% |
| | F-gases | 3279 | 3171 | 3161 | 117 | 4% | 117 | 4% |
| | **sum non-CO2** | **17369** | **17569** | **18880** | **-1202** | **-7%** | **-635** | **-4%** |
| Bulgaria | CH4 | 7338 | 8219 | 8003 | 216 | 3% | 980 | 12% |
| | N2O | 5036 | 4478 | 4397 | 81 | 2% | 81 | 2% |
| | F-gases | 1240 | 1171 | 405 | 835 | 71% | 835 | 71% |
| | **sum non-CO2** | **13614** | **13868** | **12804** | **1133** | **8%** | **1897** | **14%** |
| Croatia | CH4 | 4103 | 4595 | 4336 | 259 | 6% | 203 | 4% |
| | N2O | 1816 | 1615 | 1636 | -21 | -1% | -21 | -1% |
| | F-gases | 488 | 466 | 316 | 172 | 37% | 172 | 37% |
| | **sum non-CO2** | **6407** | **6677** | **6287** | **410** | **6%** | **354** | **5%** |
| Cyprus | CH4 | 816 | 913 | 919 | -5 | -1% | -17 | -2% |
| | N2O | 280 | 249 | 238 | 11 | 4% | 11 | 4% |
| | F-gases | 251 | 233 | 145 | 106 | 45% | 106 | 45% |
| | **sum non-CO2** | **1347** | **1396** | **1302** | **111** | **8%** | **99** | **7%** |
| Czech Republic | CH4 | 13927 | 15598 | 13647 | 1951 | 13% | 977 | 6% |
| | N2O | 5665 | 5037 | 5067 | -29 | -1% | -29 | -1% |
| | F-gases | 3400 | 3193 | 2054 | 1346 | 42% | 1346 | 42% |
| | **sum non-CO2** | **22991** | **23828** | **20768** | **3268** | **14%** | **2294** | **10%** |
| Denmark | CH4 | 6857 | 7680 | 7764 | -84 | -1% | -85 | -1% |
| | N2O | 5247 | 4666 | 5107 | -441 | -9% | -441 | -9% |
| | F-gases | 635 | 610 | 1069 | -433 | -71% | -433 | -71% |
| | **sum non-CO2** | **12739** | **12956** | **13940** | **-958** | **-7%** | **-959** | **-7%** |
| Estonia | CH4 | 1089 | 1220 | 1691 | -471 | -39% | 2 | 0% |
| | N2O | 921 | 819 | 918 | -99 | -12% | -99 | -12% |
| | F-gases | 225 | 220 | 174 | 52 | 24% | 52 | 24% |
| | **sum non-CO2** | **2236** | **2259** | **2783** | **-519** | **-23%** | **-46** | **-2%** |
| Finland | CH4 | 4870 | 5455 | 5315 | 140 | 3% | -6 | 0% |
| | N2O | 4698 | 4178 | 4626 | -448 | -11% | -448 | -11% |
| | F-gases | 1431 | 1397 | 1654 | -223 | -16% | -223 | -16% |
| | **sum non-CO2** | **10999** | **11030** | **11595** | **-531** | **-5%** | **-678** | **-6%** |
| France | CH4 | 57071 | 63919 | 67849 | -3929 | -6% | 382 | 1% |
| | N2O | 41599 | 36992 | 39144 | -2151 | -6% | -2151 | -6% |
| | F-gases | 20168 | 19185 | 15329 | 4839 | 25% | 4839 | 25% |
| | **sum non-CO2** | **118838** | **120097** | **122321** | **-1242** | **-1%** | **3070** | **3%** |
| Germany | CH4 | 57079 | 63929 | 56974 | 6955 | 11% | 4517 | 7% |
| | N2O | 38698 | 34413 | 37757 | -3344 | -10% | -3344 | -10% |
| | F-gases | 15473 | 14828 | 18551 | -3077 | -21% | -3077 | -21% |
| | **sum non-CO2** | **111251** | **113169** | **113282** | **533** | **0%** | **-1904** | **-2%** |
| Greece | CH4 | 10003 | 11204 | 13818 | -2614 | -23% | -706 | -6% |
| | N2O | 4244 | 3774 | 3712 | 62 | 2% | 62 | 2% |
| | F-gases | 6124 | 5821 | 3570 | 2555 | 44% | 2555 | 44% |
| | **sum non-CO2** | **20371** | **20798** | **21100** | **2** | **0%** | **1911** | **9%** |
| Hungary | CH4 | 7407 | 8296 | 9945 | -1649 | -20% | -47 | -1% |
| | N2O | 4366 | 3882 | 5093 | -1211 | -31% | -1211 | -31% |
| | F-gases | 2317 | 2227 | 1308 | 1009 | 45% | 1009 | 45% |
| | **sum non-CO2** | **14090** | **14406** | **16346** | **-1851** | **-13%** | **-248** | **-2%** |
| Ireland | CH4 | 13321 | 14920 | 16441 | -1522 | -10% | 45 | 0% |
| | N2O | 6359 | 5655 | 7729 | -2074 | -37% | -2074 | -37% |
| | F-gases | 1102 | 1039 | 1211 | -109 | -11% | -109 | -11% |
| | **sum non-CO2** | **20782** | **21614** | **25382** | **-3705** | **-17%** | **-2138** | **-10%** |
| Italy | CH4 | 43801 | 49057 | 53402 | -4346 | -9% | 940 | 2% |
| | N2O | 17547 | 15604 | 15502 | 102 | 1% | 102 | 1% |
| | F-gases | 16912 | 16235 | 24391 | -7480 | -46% | -7480 | -46% |
| | **sum non-CO2** | **78259** | **80896** | **93295** | **-11723** | **-14%** | **-6438** | **-8%** |
| Latvia | CH4 | 1728 | 1935 | 2029 | -93 | -5% | 62 | 3% |
| | N2O | 2019 | 1796 | 1688 | 108 | 6% | 108 | 6% |
| | F-gases | 230 | 219 | 157 | 72 | 33% | 72 | 33% |
| | **sum non-CO2** | **3977** | **3950** | **3874** | **87** | **2%** | **242** | **6%** |

Cont.Table 26: Deviation of GAINS EUCLIMIT5 (Reference 2020) from UNFCCC (2019).

| Country | Gas | UNFCCC (2019) GWP$_{100}$ IPCC AR4 year 2015 | UNFCCC (2019) GWP$_{100}$ IPCC AR5 year 2015 | GAINS REF2020 (before calibration) | Difference GAINS vs UNFCCC for year 2015 | | Difference GAINS vs UNFCCC when excluding differences in CH4 from the | |
|---|---|---|---|---|---|---|---|---|
| | | kt CO2eq | kt CO2eq | kt CO2eq | kt CO2eq | % | kt CO2eq | % |
| Lithuania | CH4 | 3426 | 3837 | 4034 | -197 | -5% | 21 | 1% |
| | N2O | 3145 | 2796 | 2902 | -105 | -4% | -105 | -4% |
| | F-gases | 573 | 557 | 403 | 170 | 31% | 170 | 31% |
| | **sum non-CO2** | **7144** | **7190** | **7339** | **-133** | **-2%** | **85** | **1%** |
| Luxembourg | CH4 | 581 | 651 | 589 | 62 | 10% | -2 | 0% |
| | N2O | 306 | 272 | 318 | -45 | -17% | -45 | -17% |
| | F-gases | 76 | 71 | 94 | -18 | -25% | -18 | -25% |
| | **sum non-CO2** | **964** | **995** | **1001** | **-1** | **0%** | **-65** | **-7%** |
| Malta | CH4 | 182 | 203 | 316 | -113 | -56% | -23 | -11% |
| | N2O | 42 | 38 | 49 | -11 | -30% | -11 | -30% |
| | F-gases | 247 | 236 | 198 | 49 | 21% | 49 | 21% |
| | **sum non-CO2** | **471** | **477** | **563** | **-75** | **-16%** | **15** | **3%** |
| Netherlands | CH4 | 18217 | 20403 | 19218 | 1184 | 6% | 1797 | 9% |
| | N2O | 8816 | 7839 | 10298 | -2458 | -31% | -2458 | -31% |
| | F-gases | 2077 | 1976 | 3938 | -1861 | -94% | -1861 | -94% |
| | **sum non-CO2** | **29110** | **30219** | **33455** | **-3135** | **-10%** | **-2523** | **-8%** |
| Poland | CH4 | 50016 | 56018 | 55826 | 192 | 0% | 4916 | 9% |
| | N2O | 19031 | 16923 | 18927 | -2004 | -12% | -2004 | -12% |
| | F-gases | 9077 | 8955 | 8878 | 199 | 2% | 199 | 2% |
| | **sum non-CO2** | **78124** | **81897** | **83631** | **-1613** | **-2%** | **3111** | **4%** |
| Portugal | CH4 | 9491 | 10630 | 13158 | -2528 | -24% | -1465 | -14% |
| | N2O | 3132 | 2785 | 3300 | -515 | -18% | -515 | -18% |
| | F-gases | 2944 | 2846 | 1604 | 1339 | 47% | 1339 | 47% |
| | **sum non-CO2** | **15566** | **16261** | **18062** | **-1704** | **-10%** | **-640** | **-4%** |
| Romania | CH4 | 29544 | 33090 | 35121 | -2031 | -6% | 879 | 3% |
| | N2O | 7452 | 6626 | 7538 | -911 | -14% | -911 | -14% |
| | F-gases | 1696 | 1618 | 908 | 788 | 49% | 788 | 49% |
| | **sum non-CO2** | **38692** | **41334** | **43566** | **-2154** | **-5%** | **756** | **2%** |
| Slovakia | CH4 | 4503 | 5043 | 4641 | 402 | 8% | 207 | 4% |
| | N2O | 2055 | 1828 | 1986 | -159 | -9% | -159 | -9% |
| | F-gases | 758 | 725 | 543 | 215 | 30% | 215 | 30% |
| | **sum non-CO2** | **7315** | **7595** | **7170** | **459** | **6%** | **263** | **3%** |
| Slovenia | CH4 | 2113 | 2366 | 2989 | -622 | -26% | -120 | -5% |
| | N2O | 720 | 640 | 721 | -81 | -13% | -81 | -13% |
| | F-gases | 375 | 357 | 297 | 78 | 22% | 78 | 22% |
| | **sum non-CO2** | **3208** | **3363** | **4006** | **-625** | **-19%** | **-122** | **-4%** |
| Spain | CH4 | 39428 | 44160 | 47040 | -2880 | -7% | 1220 | 3% |
| | N2O | 17834 | 15859 | 17035 | -1176 | -7% | -1176 | -7% |
| | F-gases | 9237 | 8624 | 6278 | 2958 | 34% | 2958 | 34% |
| | **sum non-CO2** | **66499** | **68643** | **70353** | **-1098** | **-2%** | **3002** | **4%** |
| Sweden | CH4 | 4631 | 5187 | 5342 | -155 | -3% | -268 | -5% |
| | N2O | 4552 | 4048 | 4513 | -465 | -11% | -465 | -11% |
| | F-gases | 1221 | 1157 | 1618 | -397 | -34% | -397 | -34% |
| | **sum non-CO2** | **10404** | **10392** | **11473** | **-1018** | **-10%** | **-1131** | **-11%** |
| EU-27 | CH4 | 406318 | 455076 | 467042 | -11966 | -3% | 14276 | 3% |
| | N2O | 215068 | 191252 | 210145 | -18892 | -10% | -18892 | -10% |
| | F-gases | 103545 | 99033 | 100946 | 2599 | 3% | 2599 | 3% |
| | **sum non-CO2** | **724931** | **745361** | **778132** | **-28259** | **-4%** | **-2017** | **-0.3%** |
| United Kingdom | CH4 | 53053 | 59420 | 70969 | -11549 | -19% | -129 | 0% |
| | N2O | 18980 | 16878 | 23380 | -6502 | -39% | -6502 | -39% |
| | F-gases | 16760 | 16088 | 15473 | 1287 | 8% | 1287 | 8% |
| | **sum non-CO2** | **88793** | **92386** | **109822** | **-16764** | **-18%** | **-5344** | **-6%** |

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