Links between production, the environment and environmental policy

Summary report
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1 Introduction

The aim of this study was to explore the linkages between economic production, the environment and environmental policy. This aim was motivated by the desire to move the EU towards being a resource-efficient, low environmental impact economy; the second of the three priority areas in the 7th Environmental Action Programme (EAP) published in June 2016. This transition is also key to the EU’s commitment to the United Nations’ Sustainable Development Goals (SDGs).

The sectors analysed in this study cover the largest of Europe’s manufacturing sectors (such as food, drink & tobacco and motor vehicles) as well as those with the largest environmental impacts, both in manufacture and use-phases (e.g. motor vehicles). They also all have substantial scope to reduce their impact upon the natural environment.

- **Food and drink**: both as a manufacturing sector and as a source of environmental challenges, this sector is relevant to all the UN SDGs. Each step in the sector’s value chain has some impact on the environment, from manufacture, packaging, trade and distribution, and waste. However, substantial mitigation opportunities exist: there is evidence that demand is already shifting in response to increased public environmental awareness. Packaging and food waste are of particular interest to this study.

- **Plastics**: a large and economically important manufacturing sector, which generates environmental impacts through the use of materials and energy, as well as the generation of waste and emissions. It is closely coupled to fossil fuel production, which provides the sector’s primary feedstock. Several SDG targets are closely linked to the plastics sector.

- **Motor vehicles**: links between the sector and the environment are numerous and significant: production is important in the scale of raw material and energy use; product design is important from the perspective of the circular economy with regards to end-of-life vehicle waste; and product use is crucial because of localised noise, air pollution and emissions of greenhouse gases (GHGs).

- **Water**: links with other sectors with substantial environmental impacts. Current investment in water supply and treatment are sub-optimal relative to the economic, social, and environmental benefits, which would accrue from additional investment in the sector. This is well recognised by EU policy makers and poses the central question as to how public institutions in Europe work to increase investment levels.

- **Waste**: the sewerage and waste management sector is an environmental services sector with a heavy dependence on ecosystem services. The collection and transportation of waste from the point of generation to the point of reuse or treatment may generate significant GHG and NOx emissions, as well as resulting in significant fossil resource depletion and traffic/congestion. Sewage sludge, the by-product of waste water treatment processes, is mainly disposed of through incineration, landfilling and application to land. On average, only 40% of solid waste is prepared for re-use or recycled, although some individual Member States achieve a rate of 70%, demonstrating how waste could be used as one of the EU’s key resources. However, many Member States still landfill over 75% of their municipal waste. In the EU, 70% of the phosphorous concentrated in sewage sludge and solid waste is not recovered, suggesting untapped potential for the recovery of nutrients which could be looped back into the soils rather than being discharged.

For each sector, the current links with the environment have been explored, ongoing and potential future changes in these links have been set out, and the ways in which these links can be altered through environmental policy explored. In addition, cross-cutting themes have been examined, including the impacts of resource efficiency across the economy and some of the key challenges to the successful implementation of environmental policy.

The rest of this summary report sets out the key messages, the methodologies applied, and the results obtained from the sector studies.
2 Key messages

This analysis highlights that comprehensive action across sectors is required to achieve the desired environmental policy goals. The identification of future policy priorities in each of the sector studies shows that the action required in each sector is distinct.

What are the bilateral links between production and the environment?
Almost all economic sectors have some impact on the environment; either directly, through the activities carried out within that sector, e.g. fuel extraction, or indirectly through supply chains, e.g. through the consumption of electricity. Besides bilateral links, the production within these sectors and with the natural environment is also inter-related.

In terms of the agricultural sector, most EU agricultural land is used for food production either directly for human consumption, or as animal feed. Environmental impacts from the production of food include soil erosion with consequent loss of carbon sinks; the removal of trees and other landscape features to increase room for agriculture with impacts both on carbon sinks and biodiversity; loss of farmland and soil biodiversity due to the harmful effects of pesticides, and emissions of greenhouse gases and pollutants to air and water from livestock and arable farming. In addition, large amounts of animal feed are imported from outside the EU, where adverse environmental consequences such as deforestation are sometimes associated with their production. Energy use in food processing is a major part of the environmental impact. Water use also contributes to the environmental footprint of the sector; food processing uses significant amounts of water, and also generates high volumes of wastewater.

More than a third of Europe’s total demand for plastics is accounted for by packaging, of which a majority is only used once. Less than a third of plastic packaging is collected for recycling, and landfilling and incineration are the dominant approaches to manage post-consumer plastic waste. Littering and environmental leakage of plastic waste impose significant socioeconomic costs.

What is the role played by consumers?
Consumer behaviour is key to determining what the ‘representative product’ of each sector is and will be; including whether products with the greatest negative environmental impacts are replaced, bringing for example reduced energy consumption and demands on natural resources. In the motor vehicles sector, consumers will ultimately determine which powertrains (electric or internal combustion engine) become dominant and the mix of different sizes of a cars; consumer behaviour also determines use of public transport. Issues related to plastics, such as marine and terrestrial litter, as well as toxic additives, are contributing to making consumers increasingly aware of the environmental consequences of plastic products.

Current consumer-driven market changes towards more eco-friendly consumption patterns generated by increasing consumers’ education, awareness and demand on reliable eco-labelling are expected to continue; and will continue to encourage producers to invest in more sustainable practices to gain competitive advantage.

What is the role of investors?
The electrification of motor vehicles presents an opportunity to reduce the environmental impacts of motorised transport. Investors have a significant role to play in financing the infrastructure required for the transformation of the vehicle stock; electric charging networks, electric highways, increased power generation capacity, hydrogen supply infrastructure etc.

Clean water is an essential component in ensuring that the European population can live healthy lives. European legislation already aims to provide such a supply, adhering to minimum standards. However, while investment in water supply and use is essential on social and environmental grounds, these arguments do not necessarily translate into a compelling financial case for investment. Water utility infrastructure is financed in a variety of ways, including private, public, national and international sources of finance – often in combination – and through a range of different financial instruments. There is evidence that current investment levels are sub-optimal relative to the economic, social, and environmental benefits that would accrue from additional sector investment. The analysis in this report considers (1) increasing the extent of blended finance to further leverage contributions from financial markets, (2) using green bonds as
an investment vehicle and (3) investment in water technology. Investors must address the challenges faced by the water sector in terms of financing infrastructure.

**How do links vary across business models and countries?**

In all cases, consumer preferences are shifting towards business models and products that are less environmentally damaging; in plastics, and in the food and drink sector, increased awareness of damaging practices, such as the widespread use of single-use plastics, are impacting on consumer and producer decisions. In addition, clean technologies, most notably in motor vehicles, but also in waste management & sewerage, are offering the potential of cost and efficiency benefits to consumers, challenging the narrative that environmentally-friendly options are more expensive. Such developments show that these sectors are clearly on a path to reducing their environmental impacts.

As with many sectors of the economy, the increasing digitalisation of one or more aspects of the motor vehicle sector has the potential to significantly disrupt business activities and, in some cases, reduce several environmental impacts.

**Future policy developments**

It is also clear that there is a role for policy in encouraging and accelerating these trends. The policies considered are primarily market-based instruments, to discourage take-up of environmentally-damaging options, such as taxes on plastics to discourage the use of single-use plastics and encourage the take-up of alternative technologies. In addition, there is a role on the supply side to encourage investment in environmentally beneficial infrastructure, such as in the water supply sector, or waste management & sewerage, and rules around public procurement.

Negative effects of environmental policy are likely to be concentrated in sectors of resource extraction. Policy is required to manage a ‘Just Transition’. Socio-economic risks are most evident where there are geographical concentrations of these sectors.

Other than the switch to electric vehicles (EVs) as a result of incentives and air pollution awareness, environmental preferences may not have a significant impact on the total demand for vehicle. The shift in ownership due to socioeconomic factors and thanks to the sharing economy could potentially improve the efficiency of motor vehicles during their use phase, while increased infrastructure and technological advances may better integrate EVs into our energy systems. These need to be in the focus of future policymaking.

There are several factors and sector-specific characteristics that contribute to the underinvestment in the water sector. These include the long-term nature of investment projects, poor management of existing stock, emerging challenges linked to climate change, the risk of low returns on investment, and complex site-specific and legislative requirements, which need to be addressed by future policymaking.

**Long-term trends**

Future trends in the plastics sector include recycling, eco-design and bioplastics in the EU. Labour demand is expected to increase as a response to changes in the chemical and manufacturing processes of plastics, as redesigning requires new skills and technologies.

Shared mobility and autonomous vehicles are important trends, which could significantly impact the vehicle market both in terms of vehicle types and demand volumes. It is not yet clear how automated vehicles will affect environmental impacts from motor vehicles in the future. Analysis of the travel, energy and carbon use impacts suggests that vehicle automation could reduce energy consumption and carbon emissions; however, there is also a risk of a rebound effect, whereby efficiencies result in more travel overall, and thereby offsetting any net benefits.

The transition to a circular economy requires substantial changes to happen at all stage of product value chains. On the consumer side, growing trends towards circular consumption approaches are leading to an increase in re-use and recycling. Changes in cultural norms have been achieved as a result of policy interventions, such as the integration of taxes/charges and deposit refund schemes. While the impact of consumers’ behaviour strongly depends on the different materials and products, an increase in consumers’ awareness on the impacts of an improper management of waste and better knowledge of recycling processes can lead to greater engagement in waste sorting and recycling activities.
3 Our approach

3.1 Methodologies applied
Across the five sectors selected, the following methods have been applied to provide a rounded assessment of:

- the current environmental footprint of each sector
- how that footprint is expected to change in the future, given changing consumer preferences and business models
- policy recommendations for reducing the impact of the sector going forwards

Literature review
The purpose of the literature review was to address the questions and sector-specific priority areas identified by DG ENV. The review also informed the other methods: guiding data collection and modelling, providing insights on the links between the economy and the environment.

Stakeholder engagement
The stakeholder engagement process supported the narratives developed. Stakeholders were consulted on key policy priorities, and how different business models could be expected to influence environmental impacts.

Life-cycle analysis
Life cycle analysis (LCA) was used to better understand the environmental impacts of a product over its entire life cycle within the five sectors of this project. The analysis includes information regarding material and energy use, waste flows and the associated environmental impacts.

Scenario modelling
Scenario modelling was used to investigate a range of future policy and technology developments. Consideration of future policies, technologies, and investment, informed by the literature review and LCA work, was then mapped into Cambridge Econometrics' E3ME model1 to evaluate economic and environmental developments. A few scenarios were constructed and modelled for each sector, including sensitivity analyses2.

The modelling with E3ME is key for assessing quantitatively the bilateral links between a given sector and the environment; both in terms of linking economic and physical flow units and in terms of linking sectors to specific policy impacts. The modelling is important for considering the impacts of future trends: scenario modelling was used to investigate the potential impact of environmental policy on links between the sector and the environment; that is, how environmental policy affects resource and energy use and how it affects the sector’s growth, employment, and investment.

3.2 Results obtained
Applying a combination of these approaches gave the opportunity to extend the analysis of the macro-projections resulting from the E3ME model with complementary micro-level data from the lifecycle analysis; for example, supply chain data from LCA work could be used to adjust input-output coefficients within the E3ME model.

Cross-economy improvements in resource efficiency
Introducing measures to bring about, at least for a period of time, an absolute decoupling of economic growth and resource use can be achieved while delivering positive economic impacts within Europe. Our modelling, based upon economy-wide resource efficiency shows positive economic impacts from delivering improvements in resource efficiency, based upon raw material consumption, of up to 2% yearly. The impacts on GDP and employment clearly demonstrate that there is an upper limit to the economic benefits

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1 For more detail, please visit www.e3me.com
2 Sensitivity analysis is key to assessing the robustness of evidence developed and understanding the conditions under which relationships modelled hold.
that can be accrued, however; while a 1.5% yearly improvement in resource efficiency leads to GDP levels 1% higher in 2030 than in 2018, when the level of ambition increases to 2% improvement per year, GDP in 2030 is only 0.7% higher than in 2018. This reflects the higher costs of more ambitious measures. Carefully targeting the measures (for example minimising impacts on food and agricultural produce prices) would present an opportunity to achieve higher levels of ambition while minimising the negative economic effects (particularly regressive outcomes). Overall, the scenarios show environmental benefits from improved resource efficiency across impacts where combustion or agricultural emissions are prime drivers (global warming potential, acidification and eutrophication potentials), as well as for resource use apart from food use.

**Modelling a carbon tax on agricultural produce**

Modelling a tax on inputs from agriculture would incentivise low-carbon production methods and ultimately reduce the indirect environmental impact of food. The study results show that such a tax is not an effective tool in this case to influence behaviour as the first order effect of such a tax on purchases by the food & drink sector is an increase in the costs, leading to a slight reduction in output (-0.22% by 2030) and a price increase by 1%. Net economic impacts are positive, with GDP 0.15% higher and employment 0.04% higher than in 2018, as a result of the revenue balancing assumption – that increased tax take by government is balanced by a reduction in income tax rates of the equivalent monetary value. The environmental impacts are limited due to the rebound effects; the analysis also shows that there will not be substantial impacts unless diets change to increase demand for less emissions-intensive food.

**Modelling a food waste tax**

Similarly, where food waste taxes result in reductions of food waste, small positive GDP and employment effects are accompanied by a contraction of the food & drink and agricultural sectors. But, implementation of this policy extending producer responsibility for food and drink producers would be difficult, not least because it has potentially regressive impacts, as food prices are increased to the detriment of poorer households who spend a larger proportion of their income on food.

**Modelling variable charges for household waste**

When variable charges for household waste were modelled, such policies led to modest increases in GDP and employment in the EU (of 0.08% and 0.06% respectively in 2030).

**Modelling a plastic tax**

The modelling of a European wide plastics tax focused on a substitution to recycled materials, rather than an absolute reduction in material use. We found positive GDP and employment effects across several different scenarios, although these effects are likely to be minimal. In these scenarios there is a contraction of the plastics sector, but demand is shifted to the waste management sector and the recovery of materials: key sectors to the circular economy. This suggests that there is no significant net cost of taxation designed to reduce the use of plastics.

In the scenario which focussed on a tax on the production of plastic, an important impact is production ‘leakage’, the displacement of production to countries in which the tax is not levied. Exports from the EU contract by €667 million by 2030 in the first production tax scenario (compared to baseline). Production outside of the EU increases, with the largest absolute increase in production being in China. When border taxes are introduced on plastics, EU domestic output increases, replacing otherwise imported products.

The macroeconomic effects of the tax and measures to encourage higher rates of recycling and reuse are typically small; however the small positive impacts (particularly on employment) are dependent upon how the accrued tax revenues are re-distributed across the economy. The results for environmental impacts show a weak overall response to the policy instruments modelled but rather reflect overall macroeconomic outcomes.

**LCA analysis of the environmental impact of alternative fuel technologies**

LCA analysis demonstrates that battery electric vehicles outperform other fuel technologies in terms of GHGs but with the electricity grid GHG intensity being a major determinant and with off-peak charging being important. This outperforming is most pronounced in an urban (and somewhat so in a suburban) setting and is eroded on highways due to the limited potential for regenerative breaking and no idling of ICEs. The battery pack & related components are the prime source of non-GHG impacts in BEVs, with end-
of-life management of batteries being important for mitigating these effects. This finding suggests that a shift to e-mobility could reduce the environmental footprint of the sector. Another environmental impact of EVs is the future increased extraction of strategic materials for batteries.

**Socioeconomic Impact assessment of decarbonising road transport**

In addition, economic analysis has demonstrated that the shift to e-mobility is also beneficial for Europe’s economy. In this sector, the modelling focused on the socioeconomic outcomes of successful policy implementation, i.e. more rapid deployment of e-mobility in Europe. Two scenarios have been modelled; one in which fuel-efficient technologies have been deployed with no shift in powertrains (i.e. no increase in sales of battery electric and plug-in hybrid vehicles), and another in which this technology deployment is accompanied by a rollout of advanced powertrains. Using Germany as an example Member State with a substantial domestic motor vehicle industry, and therefore more acutely affected by the transition, GDP is estimated to be around 0.8% higher in 2050 as a result of a substantial shift towards advanced powertrains. In terms of employment, there is a net increase in jobs in both scenarios. Emissions, both in terms of CO$_2$ and other pollutants (NO$_x$ and PM$_{10}$) are reduced by around 90% in 2050 compared to the baseline (of no change in the fuel efficiency or powertrains in vehicles being sold).

The modelling shows that the location of battery cell production could have a substantial impact on the overall economic effect of a shift to advanced powertrains. Scenario variants were explored where: Germany produces all battery cells domestically; imports all battery cells; and produces 50% and imports 50. Even in the most pessimistic case where all battery cells are imported, the overall impact of the transition is positive for the German economy, as the country is dependent on imports of crude oil and manufactured fuel to meet the demands for road transport. So, a reduction in demand for oil in the German economy leads to an improvement in the balance of trade and to an increase in spending on goods and services.

**Quantitative analysis of bridging the water investment gap**

As was the case with motor vehicles, the modelling tools available in this project are better able to evaluate the socioeconomic impacts of the successful implementation of these policies, rather than the marginal impact of a single policy. The quantitative analysis of policy therefore focussed on measuring the impact of bridging the water investment gap; namely, additional annual investment of €58bn (2016 prices), compared to recent historical expenditure, over the period 2019 to 2025.

The principal effect in the short term is a significant investment stimulus across the EU. The investment stimulus results in EU GDP being around 0.5% higher in 2025 than in the baseline. The increase in economic activity from 2019 to 2025 is greater than the investment stimulus itself, given the indirect and induced effects of the water sector investment. After the investment period, the cost of the investment reduces growth in economic activity, compared to the baseline. Following full repayment of the debt, EU economic growth recovers. From 2046 onwards, EU GDP is at a level very similar to the baseline.

**Socioeconomic impacts of policies targeting waste and wastewater**

Implementing policies aimed at increased reuse of waste and wastewater and more use of sewage-to-energy processes could lead to substantial positive environmental benefits, through improved resource efficiency. However, the macroeconomic impacts are likely to be relatively small, and primarily redistributive across the economy – away from material extraction and towards recycling and re-use, in line with the typical impacts of circular economy measures. Increased use of sewage for energy generation has similar impacts; given the overall scale of additional generation compared to EU-wide energy demand, no impact on overall energy prices would be expected to result from such a shift.

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1 This analysis is backed-up by analysis at a European level, such as in Harrison (2018), where, at the level of the EU28, a similar decarbonisation scenario delivers a GDP impact of +0.5% by 2050 while reducing tailpipe CO$_2$ emissions from passenger cars by around 90%.
Conclusions

The identification of future policy priorities in each of the sector studies shows that the action required in each sector is distinct. Given the distinct nature of policy interventions across sectors, the macroeconomic effects are expected to be largely additive.

1. Consumer preferences have a substantial role to play in reducing the environmental impact of the food & drink sector; a key challenge for the sector is meeting demand for new products (such as meat substitutes). In all cases, the positive macroeconomic impacts are expected to be small and largely rely on the use of any tax revenues from market-based instruments. However, there is the potential for adverse distributional effects, as many of these policies would be expected to push up the price of food & drink products, which would more adversely affect poorer households who spend a larger proportion of their income on these products.

2. Recently, the environmental impact of the plastics sector has received substantial attention from both the general public and policy makers. This is likely to result in rapid changes in consumer preferences and behaviour, as well as substantial policy, to reduce consumption of (in particular) single use plastics. Three key policies for closing the loop on plastic production were analysed, with the potential to dramatically reduce the emissions of the sector. This analysis suggests that there is no significant net cost of taxation designed to reduce the use of plastics. Changing sector priorities away from virgin inputs towards the management and conversion of plastic waste has the potential to generate employment as these activities are foreseen to be more labour intensive. Increasing recycled content in the manufacturing of plastics products can reduce emissions as less energy is needed per unit of output.

3. Transport is a major contributor to climate change and has substantial environmental impacts, through GHG emissions, releases of other pollutants (such as PMs), as well as noise and land use changes. However, new technologies, in terms of electrification of powertrains and the potential for autonomous vehicles, offer potential ways of reducing these impacts. Priorities for future policy in the production and use of motor vehicles should be designed with two broad aims in mind: (1) Shifting demand towards low-carbon mobility options and (2) Managing overall demand for road transport. European policy has typically focussed on the former; CO₂ standards will continue to reduce emissions from new vehicles in the future. In addition, market-based instruments can be used to reduce the environmental impacts and dependencies of motor vehicles across their product life cycles. Macroeconomic modelling has demonstrated that the shift to e-mobility is also beneficial for Europe’s economy. However, policymakers should be aware of the potential for rebound effects to reduce these benefits, while changes to taxation should also consider non-tailpipe environmental impacts, including PMs from tyre wear.

4. Planned investment in the water supply sector is insufficient to meet the requirements of the existing water legislation. A gap of around €58bn has been identified; and there is a role for policy to help to bridge this gap. Policy options focus on encouraging investment from the private sector, which could have notable macroeconomic impacts. In the macroeconomic modelling undertaken, the investment is funded through public sector expenditure, which requires higher taxation rates in order to generate the required government revenues. However, if the investment can be funded through private investment, these tax increases are not required, which is likely to give a more favourable economic outcome from the investment.

5. Well-targeted policy can reduce the substantial environmental impacts of waste management and sewerage. Implementing these policies could also lead to a reduction in environmental impacts in other sectors, for example where waste can be used for energy generation, or by reducing waste and therefore encouraging greater resource utility. The clearest opportunities for reducing the environmental impact of this sector are from increased re-use and recycling of waste and wastewater; and while some measures require substantial investment, such as for improved waste sorting, others can be achieved at relatively low cost, such as standards for wastewater use.
6. **Economy-wide resource efficiency** presents an opportunity to improve environmental outcomes in Europe while also providing a boost to economic outcomes; introducing policies to bring about no change in absolute resource use in Europe between now and 2040 (i.e. by achieving resource efficiency equivalent to output growth) leads to GDP 1.6% being higher, and employment 0.9% being higher, than in a baseline where no such measures are introduced. However higher levels of ambition start to reduce the economic benefits (ultimately leading to economic costs), even though the environmental benefits continue to increase.

7. There are a number of challenges to the **successful implementation of environmental policy**:

   a. **Rebound effects** – environmental policy (particularly efficiency policy) can lead to additional consumption of resources because of increased incomes. Rebound effects need to be accounted for when determining the true benefits of environmental policy.

   b. **The role of technology** – technology plays a crucial part in the pathway to improved resource efficiency. However, new technologies face many barriers in their development and implementation. Government intervention is needed to ensure that the true economic benefit of these technologies is realised, and firms are incentivised to innovate.

   c. **Financing of investment** – a key barrier faced by new technologies is inadequate financing from the private sector. Government intervention is required to encourage private sector investment in new resource efficient technology developments.

   d. **Crowding out and capacity constraints** – low-carbon and circular economy transitions necessitate a significant reallocation of capital and labour resources in the economy, and effective environmental policy should therefore be designed to consider current and anticipated future capacity constraints.

Each of these factors needs to be considered when designing environmental policy, to ensure that a policy framework effectively meets the environmental objectives of the 7th EAP, and at the same time facilitates economic growth and the creation of jobs.
Links between production, the environment and environmental policy

Final report
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1 Introduction

This report brings together the analysis carried out under contract number ENV.F.1/FRA/2014/0063, titled “Links between production, the environment and environmental policy”. This report is the final deliverable from the project. It summarises the main findings and conclusions from the analysis that has been undertaken, drawing on seven Annex documents (five sector studies, an assessment of environmental protection expenditure data across the sectors, and a cross-cutting piece of analysis examining the potential scale of the impact of cross-theme environmental policy action).

The aim of this project was to explore the linkages between economic production, the environment and environmental policy. This aim was motivated by the desire to move the EU towards being a resource-efficient, low environmental impact economy; the second of the three priority areas in the 7th Environmental Action Programme (EAP) published in June 2016. This transition is also key to the EU’s commitment to the United Nations’ Sustainable Development Goals (SDGs).

To effect a change in the resource usage of the economy, it is necessary to understand both the historical interlinkages between the economy and the environment, and the way those linkages are likely to change in the future. The analysis in this study seeks to understand how changing trends from the perspective of consumers, firms, the sector and the wider economy impact upon the environmental footprint of industry.

Specifically, this project studies five key sectors; to give clear insight into the business models used in these sectors, their impacts upon the environment (as a source of inputs, and as a sink for emissions and waste products), and how those impacts might develop in the future.

Policy has an explicit role to play in shaping the nature of future linkages. The evidence base developed in this project will inform the evaluation of current policy and the design of future policy, with the aim of informing the development of the 8th EAP.

This chapter describes the methodologies applied, and the sector selection for the analysis. Chapter 2 assesses the potential for cross-cutting environmental policy, and the opportunities and challenges that this presents, also covering the evolution of environmental protection expenditure (EPE) across the EU. Chapters 3-7 report the key findings from each of the sector studies. Chapter 8 concludes. The underlying studies for this Final Report are provided as separate annex files, Appendix B presents Cambridge Econometrics’ E3ME model. Appendix C lists the references used in the study.

1.1 Methods applied in this study

This study applies a variety of methods and information sources to construct a clear assessment of the current and future environmental impact of sectors, as well as an evaluation of relevant future policy priorities and how they might impact, both in terms of environmental footprint and socioeconomic outcomes. There are four primary research methods applied:

- **Literature review** – the purpose of the literature review is to address qualitatively the aims of the study, through drawing on existing evidence. The review also informs the other methods, i.e. guides methodology, data collection and modelling, providing insights on the links between the economy and the environment.

- **Stakeholder engagement** - the aim of the stakeholder engagement is to support the narratives developed in other methods. Stakeholders have been consulted on key policy priorities, and how different business models can be expected to influence environmental impacts.

- **Life cycle analysis** (LCA) - Life cycle approaches consist of a wide range of quantitative methodologies that allow users to better understand the environmental impacts of a product over its entire life cycle. This includes information regarding material and energy use, waste flows and the associated environmental impacts (e.g. water use, climate change, land use, acidification, etc.). These have been applied as relevant to each of the sector studies to assess the specific environmental impact of example products.
• **Macroeconomic modelling** – Cambridge Econometrics’ E3ME model\(^4\) has been used to apply scenario modelling techniques to investigate the potential impact of environmental policy on links between the sector and the environment; that is, how does environmental policy affect resource and energy use and how does it affect the sector’s growth, employment, and investment.

Across the five studies, these methods have been applied to provide a rounded assessment of the current environmental footprint of each sector, how that footprint is expected to change in the future given changing consumer preferences and business models and set out clear policy options for reducing the impact going forward. Finally, the potential environmental and/or socioeconomic impact of these policies has been explored.

### 1.2 Identification of sectors for analysis

The first task within this project was to identify the sectors to be the focus of subsequent evidence gathering and modelling work in the remainder of the study. The choice of sectors is crucial as it determines the content and focus of the study as a whole and the usefulness of the study outputs. This section presents a summary of the process and sectors identified.

**Factors to consider in choice of sectors**

Selection of sectors needs to take account of the strength of their linkages to the environment, as well as the potential policy opportunities that enhanced evidence may provide for DG Environment and the Member States. These opportunities may relate to the scope to revise or formulate new policies, designed to encourage greater business responsibility for the consumption of environmental resources (raw materials and pollution) associated with the production of sector outputs (and consumption, to the extent that producers influence through design and operational guidance the environmental impacts from the use of their products). The potential to influence stakeholder groups such as investors, regulators, consumers and local communities is also an important characteristic. The structure of the sector – and the numbers and sizes of firms which are the target of awareness raising and policy development – is also a relevant factor.

The objectives of the study are wide-ranging and suggest that a broad range of sectors across the economy (including primary, manufacturing and service sectors) should be considered; however, the work should avoid sectors which are subject to their own detailed policy reviews and evaluations (such as chemicals, agriculture, energy and construction).

Assessing sectors’ use of raw materials will be an important aspect of the analysis, both in understanding dependencies on the environment and assessing sectoral environmental impacts. However, the analysis of environmental dependencies could usefully go beyond examination of raw material use to examine sectors’ broader dependencies on ecosystem services. Examples include the dependence of the tourism and recreation sector on cultural ecosystem services, and the water industry on regulating services. This suggests that the analysis might need to look, as far as possible, beyond the total material consumption indicator in the accounts, while recognising that such linkages may be difficult to quantify.

Furthermore, the selection criteria also need to take account of how amenable the sectors are to the subsequent modelling and research work, and in particular how well the E3ME model and LCA approaches are likely to be able to quantify and explain sectoral impacts and dependencies.

**Sector Definitions**

To facilitate the modelling work, the list of sectors to be included has been drawn from the list of 70 sectors in the E3ME model. These broadly follow the two-digit sector codes within the standard NACE industrial classification adopted by Eurostat.

**Selection Criteria**

Firstly, the following sectors were excluded because they are already subject to policy initiatives and reviews, the scope for further policy influence is limited, and there is limited value in collecting additional evidence regarding environmental linkages:

\(^4\) e3me.com
Agriculture
Fishing
Other chemicals
Electricity
Construction

The remaining 65 sectors have been assessed against a set of specified selection criteria, relating to:
- The strength of the linkages of the sector to the environment;
- The economic significance of the sector;
- The policy relevance or potential policy influence of the sector; and
- The amenability of the sector to modelling and literature review.

More specific criteria are defined in each of these categories, giving a total of 15 criteria.

Each sector has been scored against each of the criteria, either on the basis of quantitative data or a qualitative judgement. Based on these data and judgements, each sector has been assigned a score of:
1. Low significance/relevance
2. Medium significance/relevance
3. High significance/relevance.

These scores are summed to give a score out of 45 across the 15 criteria.

Long List of Sectors

Scoring the 70 industrial sectors against the criteria specified gave us a long list of sectors with a score of 33 or more out of 45 (Table 1.1).

<table>
<thead>
<tr>
<th>Rank</th>
<th>Sector</th>
<th>Score (out of 45)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Food, drink &amp; tobacco</td>
<td>42</td>
</tr>
<tr>
<td>2</td>
<td>Wood &amp; wood products</td>
<td>36</td>
</tr>
<tr>
<td>3</td>
<td>Motor vehicles</td>
<td>35</td>
</tr>
<tr>
<td>4</td>
<td>Non-metallic mineral products</td>
<td>35</td>
</tr>
<tr>
<td>5</td>
<td>Other mining</td>
<td>35</td>
</tr>
<tr>
<td>6</td>
<td>Paper &amp; paper products</td>
<td>35</td>
</tr>
<tr>
<td>7</td>
<td>Pharmaceuticals</td>
<td>35</td>
</tr>
<tr>
<td>8</td>
<td>Sewerage &amp; waste management</td>
<td>35</td>
</tr>
<tr>
<td>9</td>
<td>Financial Services</td>
<td>35</td>
</tr>
<tr>
<td>10</td>
<td>Accommodation &amp; food services</td>
<td>34</td>
</tr>
<tr>
<td>11</td>
<td>Basic metals</td>
<td>34</td>
</tr>
<tr>
<td>12</td>
<td>Fabricated metal products</td>
<td>34</td>
</tr>
<tr>
<td>13</td>
<td>Land transport, pipelines</td>
<td>34</td>
</tr>
<tr>
<td>14</td>
<td>Water, treatment &amp; supply</td>
<td>34</td>
</tr>
<tr>
<td>15</td>
<td>Human health activities</td>
<td>33</td>
</tr>
<tr>
<td>16</td>
<td>Rubber &amp; plastic products</td>
<td>33</td>
</tr>
</tbody>
</table>

Sector Selection

The long list of 16 sectors provided a good starting position for selecting the 5 sectors on which the rest of the study focused. The 5 sectors were chosen as these:
- Span a range of sectors to cover a breadth of economic activity;
- Include a range of environmental linkages and policy issues (e.g. SDGs, circular economy issues, opportunities for fiscal instruments); and
- Are amenable to modelling through E3ME and life cycle analysis.

On this basis, we proposed that the 5 sectors are drawn from the following list of 10.

<table>
<thead>
<tr>
<th>Broad sector</th>
<th>Detailed sectors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manufacturing</td>
<td>Food, drink and tobacco</td>
</tr>
<tr>
<td></td>
<td>Wood and wood products</td>
</tr>
</tbody>
</table>
The following sectors were excluded.

**Table 1.3 Sectors excluded**

<table>
<thead>
<tr>
<th>Sector to be excluded</th>
<th>Explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Non-metallic mineral products</td>
<td>These sectors all have significant environmental impacts which can be modelled. However, they have low growth prospects, and environmental impacts are generally well understood and give rise to less interesting environmental policy challenges than the sectors listed above.</td>
</tr>
<tr>
<td>Paper &amp; paper products</td>
<td></td>
</tr>
<tr>
<td>Basic metals</td>
<td></td>
</tr>
<tr>
<td>Fabricated metal products</td>
<td></td>
</tr>
<tr>
<td>Financial services</td>
<td>This is an economically important and growing sector, and is interesting with respect to its engagement with other sectors. However, because direct environmental impacts are less significant, it is not amenable to modelling of environmental impacts or dependencies.</td>
</tr>
<tr>
<td>Land transport, pipelines</td>
<td>Environmental impacts of transport are significant and policy relevant; however, our preference would be to examine the motor vehicles sector which gives rise to a range of issues and linkages in production as well as use.</td>
</tr>
</tbody>
</table>

Following discussion with the Commission, these five sectors were selected:

**Table 1.4 Sectors selected**

<table>
<thead>
<tr>
<th>Sectors selected</th>
<th>Explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Food, drink and tobacco</td>
<td>A large, economically important and growing sector subject to a range of interesting issues and challenges relating to ecosystem dependencies/ raw materials and links to land management, environmental impacts of production, packaging and circular economy, as well as food waste.</td>
</tr>
<tr>
<td>Motor vehicles</td>
<td>An economically important and politically significant manufacturing sector which raises a number of issues with respect to raw materials, production impacts, circular economy as well as product design and use.</td>
</tr>
<tr>
<td>Rubber and plastics</td>
<td>Another manufacturing sector which plays a key role in supply chains, and generates environmental impacts through use of materials and energy and generation of wastes and emissions, and especially through product life cycle impacts. The sector presents a number of significant issues around circular economy and recyclability/ reusability of products, as well as impacts of marine litter. Amenable to LCA modelling.</td>
</tr>
<tr>
<td>Sewerage and waste management</td>
<td>an environmental services sector, with a strong influence on other sectors, dependence on ecosystem services, environmental policy challenges relating to wastes and emissions, and policy influence through links to SDGs (sanitation). Opportunities exist in this sector in the recovery of materials and biomass.</td>
</tr>
<tr>
<td>Water treatment and supply</td>
<td>an environmental services sector, with a strong influence on other sectors and with strong links to SDGs, strong dependence on ecosystem services, and environmental policy challenges relating to water abstraction. The sector is amenable to modelling and presents interesting issues around scarcity and the role of pricing, as well as finance and investment.</td>
</tr>
</tbody>
</table>
2 Cross-cutting analysis and the environmental protection expenditure

2.1 Introduction

The 7th Environmental Action Programme (EAP) lists nine priority objectives and sets out how they can be achieved by 2020 (EC, undated). There are three specific key objectives;

1. to protect, conserve and enhance the Union’s natural capital
2. to turn the Union into a resource-efficient, green, and competitive low-carbon economy
3. to safeguard the Union’s citizens from environment-related pressures and risks to health and wellbeing.

These are then supported by four “enablers”;

4. to maximise the benefits of the Union’s environment legislation by improving implementation
5. to increase knowledge about the environment and widen the evidence base for policy
6. to secure investment for environment and climate policy and account for the environmental costs of any societal activities
7. to better integrate environmental concerns into other policy areas and ensure coherence when creating new policy with two “horizontal” priority objectives;

8. to make the Union’s cities more sustainable
9. to help the Union address international environmental and climate challenges more effectively.

The sector studies can best be mapped to the 7th EAP in terms of the thematic priorities from the EAP (see Table 2.1).

Table 2.1 Mapping priorities outlined in the 7th EAP to Sector Studies

<table>
<thead>
<tr>
<th>Natural capital: air</th>
<th>Food, drink, and tobacco</th>
<th>Plastics</th>
<th>Motor vehicles</th>
<th>Water treatment and supply</th>
<th>Sewerage and waste management</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural capital: water</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Natural capital: land</td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Resource efficiency, waste, and the circular economy</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Biodiversity</td>
<td>X</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low-carbon and climate change</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Human health and well-being</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
</tbody>
</table>

The mapping highlights the fact that for most objectives, comprehensive action across sectors is required. The identification of future policy priorities in each of the sector studies show that the action required in each sector is distinct. Given the distinct nature of policy interventions across sectors, the macroeconomic effects are expected to be largely additive – that is, the macroeconomic impacts from each policy are distinct, and not inter-related; so introducing all policies across all themes simultaneously would lead to an impact on GDP equivalent (approximately) to the sum of the individual impacts.
2.2 The macroeconomic impacts of resource efficiency

Resource efficiency is a key component of the EC’s 7th EAP; managing waste flows and sustainable use of natural resources are needed to live within the means of our planet. This section addresses the recurring research question as to whether pursuing environmental policy, and particularly resource efficiency, is also economically beneficial. Identifying potential economic co-benefits of environmental policy can be used to engage stakeholders, where they would not take action motivated purely by environmental concerns.

Resource productivity at the economy level can be measured by calculating GDP per unit of raw material consumption. Cambridge Econometrics (2014) argues that whilst resource productivity in the EU28 has increased substantially in recent years, 19.6% from 2001 to 2011, raw material consumption has remained relatively coupled with economic growth; before the financial crisis, from 2001 to 2007, average annual resource productivity was +1.2% but GDP growth was on average +2.3%. The analysis from Cambridge Econometrics (2014) has been updated for this study; the 2011-2018 trend is similar, resource productivity increasing by 1.9% per year.

In the updated analysis, resource productivity improvements are assumed in the baseline: the average resource productivity growth rate is -0.01% from 2018 to 2040. The introduction of resource efficiency leads to positive economic impacts; to the extent that an absolute decoupling of GDP growth and resource consumption can be achieved while increasing GDP and employment. When resource productivity improvements of 1.5% per annum are modelled to 2040, leading to very little change in overall material consumption over 2018-40, GDP is 1.0% higher and employment 0.5% higher in 2030 compared to the baseline. However, a higher level of ambition, represented by a 2% per annum improvement in resource efficiency, reduces these gains to 0.7% in GDP and 0.3% employment, suggesting that socioeconomic gains peak and then start to decline at more ambitious levels of resource efficiency. Such a finding is consistent with Cambridge Econometrics (2014). Table 2.2 details macro-economic impacts in 2030, compared to the baseline.

Table 2.2 Socioeconomic impacts in 2030, percentage difference from baseline

<table>
<thead>
<tr>
<th>2030</th>
<th>1% productivity improvement</th>
<th>resource per year</th>
<th>1.5% productivity improvement</th>
<th>resource per year</th>
<th>2% productivity improvement</th>
<th>resource per year</th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP</td>
<td>0.9%</td>
<td></td>
<td>1.0%</td>
<td></td>
<td>0.7%</td>
<td></td>
</tr>
<tr>
<td>Employment</td>
<td>0.5%</td>
<td></td>
<td>0.5%</td>
<td></td>
<td>0.3%</td>
<td></td>
</tr>
<tr>
<td>Consumer spending</td>
<td>1.2%</td>
<td></td>
<td>1.1%</td>
<td></td>
<td>0.1%</td>
<td></td>
</tr>
<tr>
<td>Investment</td>
<td>1.3%</td>
<td></td>
<td>2.0%</td>
<td></td>
<td>2.5%</td>
<td></td>
</tr>
<tr>
<td>Imports (extra-EU)</td>
<td>-0.2%</td>
<td></td>
<td>-0.1%</td>
<td></td>
<td>-0.4%</td>
<td></td>
</tr>
<tr>
<td>Exports (extra-EU)</td>
<td>-0.3%</td>
<td></td>
<td>-0.3%</td>
<td></td>
<td>-0.6%</td>
<td></td>
</tr>
</tbody>
</table>

There are a number of competing factors which affect the macroeconomic impacts of increasing resource efficiency. Table 2.3 lists the positive and negative effects of resource efficiency policy, identified by Cambridge Econometrics (2014). Policy to increase resource productivity imposes costs on industry; market-based instruments would directly increase the cost of material use, and regulation would require industry to invest in abatement measures. Investment in resource efficiency is the key positive economic driver, particularly in the short to medium term.

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5 Resource productivity ‘improvement per year’ values are defined by the compound average growth rate in resource productivity from 2018 to 2040 (not calculated as a difference from the baseline).

6 There may be some zero cost improvements in resource efficiency, associated with waste reduction and information improvement (EC, 2014). In this case, industry costs would fall in this part of the abatement cost curve.
Table 2.3 Key macroeconomic impacts of resource efficiency policy

<table>
<thead>
<tr>
<th>Positive</th>
<th>Negative</th>
</tr>
</thead>
<tbody>
<tr>
<td>Investment in resource efficiency</td>
<td>Higher costs of resource use, higher domestic prices</td>
</tr>
<tr>
<td>Revenue recycling of market-based instrument revenue (dependent on policy choice)</td>
<td>Lower exports; reduced competitiveness of domestic industry (resource use equivalent of carbon leakage)</td>
</tr>
<tr>
<td>Imports; decrease in imports of resources</td>
<td></td>
</tr>
</tbody>
</table>

Sectoral effects of resource productivity should also be considered. Intermediate sectors which sell raw materials face significant contraction of output. Material resource intensive sectors face higher proportional increases in costs. However, sectors such as manufacturing and construction, which are material-intensive benefit from demand stimulus from investment in resource efficiency. Effects across Member States are likely to be driven by relative domestic importance of resource extraction and processing, manufacturing, and construction industries.

Sectoral-level scenario analysis in this project provides complementary evidence to these studies of economy-wide resource use. Modelling of a plastics tax focused on examining substitution to recycled, rather than absolute reduction of material use: positive GDP and employment effects are found across tax design scenarios, accompanied by a contraction of the plastics sector. Similarly, modelling reductions of food waste, positive GDP and employment effects are accompanied by a contraction of the food, drinks and tobacco and agricultural sectors. Positive macroeconomic effects in these scenarios largely rely on the revenue recycling mechanism of any tax revenues.

Negative effects of environmental policy are likely to be concentrated in targeted sectors, namely sectors of resource extraction. Policy is required to manage a ‘Just Transition’. Socio-economic risks are most evident where there are geographical concentrations of these sectors.

2.2.1 Environmental implications

The modelling of cross-economy resource efficiency carried out in this study also considered the environmental implications of resource efficiency policy. Modelling of CO₂ emissions and resource use was carried out using the macroeconomic model E3ME, with additional impacts considered via coupling with the EXIOBASE environmentally-extended input/output database. The same resource efficiency scenarios have been used, taking into account the following impacts:

Table 2.4 Environmental impacts considered in the modelling of cross-economy resource efficiency

<table>
<thead>
<tr>
<th>Impact considered</th>
<th>Unit of measurement</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acidification potential</td>
<td>Thousand tonnes SO₂ equivalent</td>
</tr>
<tr>
<td>Eutrophication potential</td>
<td>Thousand tonnes PO₄equivalent</td>
</tr>
<tr>
<td>Photochemical oxidation potential</td>
<td>Thousand tonnes ethylene equivalent</td>
</tr>
<tr>
<td>Human toxicity potential</td>
<td>Thousand tonnes 1,4-DCB equivalent</td>
</tr>
<tr>
<td>Global warming potential</td>
<td>Thousand tonnes CO₂ equivalent</td>
</tr>
<tr>
<td>Land use</td>
<td>Square kilometres</td>
</tr>
<tr>
<td>Food use</td>
<td>Thousand tonnes</td>
</tr>
<tr>
<td>Feed use</td>
<td>Thousand tonnes</td>
</tr>
<tr>
<td>Forestry use</td>
<td>Thousand tonnes</td>
</tr>
<tr>
<td>Construction Minerals use</td>
<td>Thousand tonnes</td>
</tr>
</tbody>
</table>

7 Plastics sector output does not contract under a border tax scenario with intermediate demand characterised by perfect inelasticity.
Across the environmental impacts considered, some similarities emerge toward 2030. Acidification and eutrophication potentials decrease by approximately 13% under a 1% and 1.5% resource efficiency scenario with little difference between scenarios. Under a 2% resource efficiency scenario, impacts decrease by approximately 16% and 18% respectively. Both decreases are primarily owed to the falling output of the agricultural sector and decreases in NH3 emissions to air and (in the case of eutrophication) N emissions to water. In both cases, falling combustible fuel use leads to decrease in NOx and (in the case of acidification) SOx emissions, which also contributes to the overall decreases across resource efficiency scenarios. The decrease in combustible fuel use with increasing resource efficiency also leads to falling GHG emissions (Global warming potential) of approx. 26% in the modelled annual resource efficiency increase of 1% and 1.5%, and up to 27% in the 2% scenario.

In contrast to the contraction of the agricultural sector, the projected increased output of the industrial sector leads to increases in Photochemical oxidation (summer smog) and Human toxicity potentials. Summer smog potential rises by approx. 4 to 4.4% with 1% and 1.5% annual resource efficiency increase and 3.6% with the 2% annual increase scenario. Human toxicity potential rises by approx. 10% regardless of scenario. In both cases, rising impacts are owed to increases in non-combustion emissions, which in the modelled scenarios are assessed via EXIOBASE by linking to sectoral gross output.

Finally, resource use itself (construction minerals, industrial minerals, ferrous ores and non-ferrous ores) is reduced by at least 25% compared to the baseline in 2030, while food use rises by approx. 70% due to the inelasticity of food demand.

The implications of the modelling exercise are heavily dependent upon key aspects of the scenario design; for example, the scenarios consider cross-economy resource efficiency, rather than allocating resource efficiency on a ‘least cost’ basis across the economy; the biggest environmental impacts therefore come from changes to the agricultural and food manufacturing sectors, although the distributional impacts of the socioeconomic outcomes from these changes are regressive (due to the inelasticity of food demand to changes in price). The linkage of non-combustion emissions to sectoral gross output means that not all expected aspects of resource efficiency are necessarily captured in the environmental measures. This is not necessarily true for the agricultural sector, where a correspondence between sectoral output and fertiliser use (and hence nitrogen emissions) can be expected. However, in the case of industrial non-combustion emissions, these are strongly related to technological aspects, and in the absence of clear data on altering future production technologies, linkage to gross output is necessitated. As EXIOBASE technological coefficients are held constant over time, this also implies that environmental impacts may be overestimated.

Overall, the scenarios show environmental benefits from improved resource efficiency across impacts where combustion or agricultural emissions are prime drivers (global warming potential, acidification and eutrophication potentials), as well as for resource use apart from food use. It is worth noting that the differences in projected impacts between resource use scenarios is somewhat limited apart from direct resource use indicators (ores and minerals use). The environmental results necessarily demand consideration alongside projected economic impacts, as though environmental benefits are seen with increased resource use efficiency, exceeding certain efficiency limits ultimately lead to economic costs (Table 2.2).

2.3 The challenges associated with the implementation of environmental policy

Much of the analysis in this report (including in the previous section) focuses on the potential benefits from the implementation of specific pieces of environmental policy. However, the LCA and modelling analysis carried out in this project, as well as a large number of previous exercises which have assessed the impact of environmental policy, also demonstrate some of the challenges associated with such policy; specifically,
barriers that can prevent the successful realisation of these socioeconomic and environmental benefits, and ultimately of achieving the goals of the 7th EAP. These can be summarised as:

- **Rebound effects** – environmental policy (particularly energy efficiency policy) can lead to additional consumption of energy because of increased incomes. Rebound effects need to be taken into account when determining the true benefits of environmental policy.
- **The role of technology** – technology plays a crucial part in the pathway to a decarbonised economy. However, low-carbon technologies face many barriers in their development and implementation. Government intervention is needed to ensure the true economic benefit of these technologies is realised, and firms are incentivised to innovate.
- **Financing of investment** – a key barrier faced by low-carbon technologies is inadequate financing from the private sector. Government intervention is required to encourage private sector investment in new clean technology developments.
- **Crowding out and capacity constraints** - low-carbon and circular economy transitions necessitate a significant reallocation of capital and labour resources in the economy, and effective environmental policy should therefore be designed considering current and anticipated future capacity constraints.

Each of these factors need to be considered when designing environmental policy, to ensure that a policy framework effectively meets the environmental objectives of the 7th EAP, and at the same time facilitates economic growth and the creation of jobs.

### 2.3.1 Rebound effects

One of the main objectives of the EAP is to ‘turn the Union into a resource-efficient, green, and competitive low-carbon economy’ and improving energy-efficiency is a key policy consideration in meeting this objective. The potential energy savings and emissions reductions to be realised from energy efficiency measures and improvements in technology are often estimated using complex models and analysis, but in reality, evidence suggests the actual energy savings and emissions reductions achieved by policy are lower than expected.

Rebound effects can be both direct and indirect. Direct rebound effects occur when households and/or firms use this additional income to consume a technology more. For example, if household heating systems are more energy efficient, we may use the money saved on bills to simply keep our heating on for longer, or in the case of firms, a reduction in energy costs leads to greater profitability, and potentially greater investment and levels of production.

There are two types of indirect rebound effects. First, indirect rebound effects occur if households instead spend the additional income that results from lower energy costs or from greater income earned through collaborative economic activities on other goods and services in the economy, creating increased demand for energy, and increased emissions in other sectors. For example, households may spend their additional income on an overseas holiday, which includes a flight with a high carbon footprint. Second, an overall reduction in energy use will mean less demand for fossil fuels, but as a result, fossil fuel prices are lowered and greater consumption of these fuels is encouraged.

**Implications for environmental policy**

There are clear economic benefits of rebound effects from energy efficiency, resource efficiency and other environmental policy measures. Assuming that savings made on energy or increased efficiency of a resource are instead spent on consuming more of the good or service, or on other goods and services in the economy, additional economic activity is generated, boosting GDP and employment. However, the economic benefits of the rebound effect may be somewhat detrimental to the environmental targets the policy first set out to achieve. As with many environmental policy measures, policy-makers are faced with the challenge of ensuring that the policy achieves the double-dividend of both reducing emissions, energy and resource use, while achieving economic growth. In the case of energy efficiency measures, little evidence exists that rebound effects would lead to overall increases in energy use that exceed the original reduction in energy use achieved (i.e. ‘backfire’). Therefore, energy efficiency policy is still effective at reducing energy use and emissions, while also leading to greater expenditure and economic growth.
Likewise, the evidence of rebound effects in resource efficiency policy can also lead to increased consumption and economic growth.

2.3.2 The role of technology

Improvements in technology are a crucial part of the pathway to a resource efficient economy for the EU, helping to meet the targets of policies set forth by the 7th EAP. As well as playing a major role in the improved resource efficiency of the EU economy, innovation and investment in new technologies also benefit the EU economy through increased economic growth and jobs, contributing towards the EU’s innovation, jobs and growth agenda. Policies and initiatives aimed at increasing investment in new technologies therefore satisfy the policy goal of achieving environmentally sustainable economic growth.

Technologies for a low-carbon future can be categorised in to four key sectors; power, transport, buildings and industry, and many of these are also relevant when considering key options for improving resource efficiency. In the power sector, renewable energy technologies that generate electricity through wind, solar, hydro or ocean, geothermal or bioenergy, instead of through fossil fuels, are playing a substantial role in the pathway to a greener EU economy. In transport, developments in electric vehicles and the use of alternative fuels such as biofuel in aviation are ongoing, helping to both decarbonise and improve overall resource efficiency. Advancements in heating and cooling systems and greater energy efficiency in appliances will be essential for the buildings sector to reduce its resource efficiency. Finally, the development of new technologies in industry offers the potential to improve both energy and resource efficiency. As previously cited within this report, countries should have a strong policy agenda across three policy response pillars – standards and regulation, markets and prices and strategic investment in clean and renewable technologies (Grubb 2014).

Implications for environmental policy

Together, the barriers to investment discussed above can lead to resource efficient technologies becoming trapped in what is known as the technology ‘valley of death’ whereby high costs and low volumes prevent the technology becoming commercially viable, and many technologies can fail to reach the marketplace. Aiding the innovation of such technologies is a key priority for policy-makers since they provide clear economic and environmental benefits. Alongside market-based instruments to internalise external costs, and potentially strong property rights, additional government support is required to pull technologies through this potential trap. Government support may come in the form of public funding programmes (such as the examples as outlined above), subsidies, supporting better access to finance, through regulations and standards.

2.3.3 Financing of investment

Cambridge Econometrics et al (2017c) highlights that returns to low-carbon investments are typically lower than the returns from fossil fuel projects, and that returns are slower to materialise. Finance for low-carbon projects is difficult to secure since these projects usually take a long time to develop and in the initial stages of operation the technology may operate at a loss. Profitability may only develop in the medium-term, ‘beyond the usual grace period provided by banks’. Furthermore, the technology may require substantial supporting infrastructure, which requires additional investment from investors, but also entails greater risk for investors. The same is true of investments in resource efficient technologies.

Cambridge Econometrics et al (2017c) identifies various barriers to investment that relate explicitly to the availability of finance. The barriers can be separated in to two distinct categories – barriers that arise from the nature of the low-carbon project, and barriers that arise from the decision-making framework of lenders. Table 2.3 below summarises the identified barriers to investment within each of these two categories.

---

8 https://ec.europa.eu/clima/policies/lowcarbon_en
| Financial barriers typical for innovative clean energy technologies/projects |
|-----------------|--------------------------------------------------|
| **Barrier** | **Specifics** |
| Lack of a clear market opportunity | Low-carbon R&D is driven by environmental policy. For a market opportunity to exist, environmental policy must require action in the short-medium term. |
| R&D uncertainty | As with all technology development, the development of low-carbon technologies is subject to the same uncertainties in terms of costs, and the time it will take to become a commercially-viable and profitable technology. Furthermore, in that time, market conditions and policy setting can change. |
| High capital costs and long-lived assets | Renewable technologies have relatively high up-front capital costs and low operating costs compared to conventional technologies, requiring long-term finance. Investment decisions may be biased towards conventional technologies. |
| Availability and volume | Financial instruments/investment products (e.g. green bonds), are not readily available for low-carbon technologies. |
| Risk-return related to regulatory (political) stability | Historically, clean energy investments have relied upon policy support to attract investment. This has led to a perception of lower returns and higher risks in such projects, compared to carbon-intensive projects in more established sectors. Such projects can be at risk of future changes to policy; however, as the cost of low carbon options falls (through learning effects), such projects are becoming competitive without subsidy, and under such conditions the rate of return on these projects would be expected to be the same as their carbon-intensive equivalents. |
| The high-risk nature of First-Of-A-Kind projects | First-Of-A-Kind projects can be considered too risky since they involve unproven technologies. The risks of these projects are difficult to determine and quantify, and institutional investors and most commercial banks generally do not have the internal expertise to address these risks. |
| Lack of coordination and complementarity | Complications arise from the lack of coordination and complementarity between financing instruments from the EU, Member States, and technology promoters. |
| Lack of financial and technical advice | Low-carbon technology developers have a lack of financial advice and investors in these technologies have a lack of technical advice. |
| Capital intensive | First-Of-A-Kind low-carbon energy projects are often too capital-intensive for venture capital investments and too risky for private equity financing. Furthermore, the lack of historical data in the sector prevents the insurance industry from designing products which could contribute to the de-risking of such projects. |
| High transaction costs | Low-carbon technology projects are often small in nature, and transaction costs are higher when investing in smaller assets. |

| Financial barriers arising from investors’ decision-making framework |
|-----------------|--------------------------------------------------|
| **Barrier** | **Specifics** |
| Time horizon of decision-making: | Most fund managers have a decision-making time horizon of maximum three years (and rely on the liquidity of their investments to adjust their portfolio to meet these short-term targets). |
| Lack of integration of climate risks into fiduciary duty and engagement practices: | The relationship between climate-related risks and benefits and institutional investors’ fiduciary duty is not clearly established. Additionally, assessments of investment managers’ engagement practices with companies do not sufficiently include climate-related concerns. |
| Lack of relevant climate- | Current climate risks (physical impacts) and carbon risks (structural policy |
### Financial barriers typical for innovative clean energy technologies/projects

| related risk and performance methodologies: | changes) face methodological shortcomings. The assessments done to date are not easily integrated into mainstream investment tools and practices. Investors currently cannot easily measure the climate and carbon performance of their portfolios. |

Source: Cambridge Econometrics et al (2017c)

### Implications for environmental policy

Many of these barriers and the government intervention to overcome them are related to inadequate financing of new technology developments. However, the government interventions may not be enough to alter the behaviour of investors. Further policy considerations may be needed to address the specific financial barriers outlined in the table above, through interventions in the credit market.

#### 2.3.4 Crowding out and capacity constraints

Capacity constraints in the economy are an important consideration when designing policy; the effect of a given policy may be dependent on the state of the economy within business cycle. A wider discussion of capacity constraints is relevant, rather than focusing only ‘crowding out’. Three key categories of supply constraints can be identified in this area of policy (Cambridge Econometrics and E3-Modelling, 2017a):

- financial capital markets
- labour markets
- product markets

Both the low-carbon and circular economy transitions necessitate a significant reallocation of resources in the economy (Cambridge Econometrics and E3-Modelling, 2017a):

- intertemporal, usually more up-front investments
- across sectors, with construction, engineering and waste management/recycling sectors usually benefiting
- between geographical regions, with fossil fuel and primary material producers and exporters losing out

Where policy increases demand in a market to the level of potential supply capacity, or beyond, displacement of resources will result. The typical mechanism of displacement is price effects.

### Implications for environmental policy

The policy options in this report should be considered in the context of the current European policy and economic environment, and its outlook. Given extant unemployment rates across Europe, it is unlikely that labour will be a capacity constraint at the macro level in the short term. Policy initiatives in green finance, crucially the European Regional Development Fund, alleviate the issue of financial capacity constraints in the transition: such funding mechanisms may ‘crowd-in’ private investment through signalling and confidence effects. Product market and sector-level skills capacity constraints, however, should be anticipated.

### 2.4 Trends in environmental protection expenditure across the EU

Almost all economic sectors have some impact upon the environment; either directly, through the activities carried out within that sector (e.g. fuel extraction), or indirectly through supply chains (e.g. through the consumption of electricity). However, these impacts (particularly the former) can be mitigated through expenditure on measures aimed at prevention, reduction and elimination of pollution or any other degradation of the environment. Such expenditure is commonly described as Environmental Protection Expenditure (EPE).

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10 Crowding out generally refers to a specific case of capacity constraints: when an increase in public sector activity results in a reduction in private sector activity, particularly investment activity.
Eurostat produces estimates of EPE by Member State and sector over time, although the comparability and coverage of the data varies somewhat (as is set out in more detail below). The EPE data provided by Eurostat has been analysed, with the aim of answering a number of key questions:

- How did total EPE change over time in the EU 28?
- How did the proportion of capital investments and operating expenses change within the total spending in the EU 28?
- Did certain key industries have a different spending pattern over time?
- Did certain regions, EU 15 and EU 13 countries, show different spending patterns?
- How did the spending of the largest EPE spender countries evolve?

Below we provide a summary of the methodology that was adopted to analyse this data and the key findings from the work; a more comprehensive analysis can be found in the Annex report *Analysis of environmental protection expenditures by industry, expenditure type and region*.

### 2.4.1 Methodology

#### Data

**For the sectoral EPE analysis** information was combined from two Eurostat datasets: 1. *Environmental protection expenditure - million euro (env_ac_exp1r2)* and 2. *Environmental protection expenditure (NACE Rev. 2, B-E) (sbs_env_dom_r2)*. Both datasets contain EPE spending at a sectoral level, across all environmental protection activities for EU 28 countries. The first data set reports capital and operating expenditure separately, but only covers the period 1995-2013. The second data set provides more recent information, from 2008-2016, but only reports capital expenditure. It is taken from the Structural business statistics (SBS) from Eurostat, which cover industry, construction, trade and services, presented according to the NACE activity classification.

To gain the fullest picture on EPE patterns, information from both sources was used.

**For the economy-wide EPE analysis** 3. *National expenditure on environmental protection by institutional sector (env_ac_epneis)* data set was used. This source covers the period 2006-2017 showing the most recent data on EPE. This data is not disaggregated by sector or expenditure type and has large gaps in its series. Thus, it is presented as an approximate picture on main and recent trends.

Additionally, data on sectoral gross value added (GVA) and overall economy GVA was used to show the relative importance of EPE: 4. *National accounts aggregates by industry (up to NACE A*64) (nama_10_a64)*.

<table>
<thead>
<tr>
<th>Years</th>
<th>Countries</th>
<th>Sectors</th>
<th>EPE Types</th>
<th>Units</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Environmental protection expenditure</td>
<td>1995-2013</td>
<td>EU 28</td>
<td>NACE-2 aggr.</td>
<td>Total, Capital Operating</td>
<td>Current prices, million €</td>
</tr>
</tbody>
</table>

---

11 Note: Production of environmental protection services of corporations other than specialist producers by economic characteristics and NACE Rev. 2 activity (env_ac_pepsnsp) was also considered as a source. However, aggregating “sectoral” EPE to total was not straightforward, therefore we decided not to use it.
### Table 2.5 Summary of Data Sources

<table>
<thead>
<tr>
<th>National expenditure on environmental protection by institutional sector</th>
<th>2006-2017</th>
<th>EU 28 missing: GR, CY, FI, HU, LT, RO</th>
<th>National</th>
<th>Total</th>
<th>Current prices, million €</th>
<th>env_ac_epneis</th>
</tr>
</thead>
<tbody>
<tr>
<td>4. National accounts aggregates by industry (up to NACE A*64)</td>
<td>1975-2017</td>
<td>EU 28</td>
<td>NACE-2</td>
<td>-</td>
<td>Current prices, million €</td>
<td>nama_10_a64</td>
</tr>
</tbody>
</table>

**Variables used**

This section describes the key variables across the different data sources:

- **Classification of Environmental Protection Activities (CEPA):** For most sectoral analysis the total spending across all environmental activities is used which aggregates environmental spending across different domains (TOTCEPA). For analysing waste and wastewater management, we show the whole industry’s spending on these specific activities: CEPA2 and CEPA3.

- **Industry (NACE_R2):** Both sectoral datasets are in principle disaggregated at the NACE-2 level and cover the whole industry sector. However, the first source groups some of the industries together and the second does not include all sectors in the first. To combine information from both sources, most analysis uses the higher level of sectoral aggregation in the EPE spending based on the first source. Whenever the more disaggregated series can be integrated to the analysis those are also presented. The sectors are consistent with those used in the rest of this report:

1. **Food, Drink and Tobacco:** C10-C12 Manufacture of food products; beverages and tobacco products
2. **Rubber and Plastics:** C20-C22 Manufacture of chemical, pharmaceutical, rubber and plastic products and C22 Manufacture of rubber and plastic products.
3. **Motor Vehicles:** C25-C33 Manufacture of metal products and other equipment and C29 Manufacture of motor vehicles, trailers and semi-trailers
4. **Electricity, Gas, Water Supply & Treatment:** D35-E36 Electricity, gas, steam and air conditioning supply; water collection, treatment and supply and E36 Water collection, treatment and supply
5. **Waste management and Sewerage:** CEPA2 Waste and CEPA3 Wastewater management expenditure across all industries present in the data (NACE B-E)
6. **Manufacturing:** C Manufacturing

- **Total Spending, Capital Expenditure, Operating Expenses:**

1. From the first sectoral dataset (env_ac_exp1r2), *Environmental protection expenditure (EE1000)* is used as total EPE. *Total environmental investments (EE1100)* is used as capital expenditure and *Total environmental current expenditure (EE1200)* as operating expenses.
2. In the second sectoral data set (sbs_env_dom_r2) there is no reported information on the operating expenses. The sum of *Investment in equipment and plant for pollution control (V21110)* and *Investment in equipment and plant linked to cleaner technology (V21120)* is used as a capital expenditure measure.
3. In the economy wide dataset (env_ac_epneis) there is no separation of different expenditure types. Therefore, we treat it as a total expenditure measure.

- **Years:** The first sectoral dataset (env_ac_exp1r2) shows the period 1995-2013, the second (sbs_env_dom_r2) covers 2008-2016. The economy-wide dataset has data from 2006-17 (env_ac_epneis). However, all series have many missing values, gaps and the first/last years are especially affected.

- **Countries:** In principle all datasets should include all EU-28 countries. However, the second sectoral dataset (sbs_env_dom_r2) has no data on Luxembourg and Latvia. The economy-wide dataset (env_ac_epneis) does not include Greece, Cyprus, Hungary, Lithuania, Finland and Romania. Therefore, caution should be applied when aggregate series from different sources are compared. Note, that even if a country is included in the dataset, it may have gaps in the EPE series.

- **Unit of Measurement:** Millions of Euros, current prices.

### 2.4.2 Economy-wide EPE

EPE across the economy as a whole has been relatively static over the period from the mid-2000s; representing around 2% of total value-added across the economy. EPE is overwhelmingly dominated by expenditure from the EU15 (i.e. western Europe), as demonstrated by the fact that the EU15 and EU28 lines overlap substantially. However, EPE in the EU13 (i.e. central and eastern Europe) was higher than the EU15 (expressed as a percentage of value added), at around 2.5% in 2008, rising to 3% in 2015. Over the same period, nominal expenditure across the EU28 rose substantially – but as this chart shows, that was only reflecting increased economic activity.

**Figure 2.1 National Environmental Protection Expenditure of EU 28, 25 and 13 countries, Normalized with Total GVA of Countries Providing Expenditure Data**

![Graph showing EPE across EU28, 25, and 13 countries normalized with total GVA of countries providing expenditure data.](image)

### 2.4.3 Sectoral EPE

**Manufacturing**
Overall trends in EPE in manufacturing show a steady decrease in expenditure, whether on operating expenditure (OPEX) or capital expenditure (CAPEX). OPEX fell from 1.7% of manufacturing value added in 1996 to 1.2% in 2013. CAPEX fell more slowly but consistently; from 0.6% of GVA in 1996 to 0.3% in 2016. As above, nominal EPE increased rapidly over this period, but as demonstrated by the chart this was at a slower rate than overall economic output from manufacturing.

Note: The two capital expenditure data are taken from two different Eurostat datasets covering different periods.

Food, drink & tobacco
Missing data provides a major challenge to interpretation of the data; even though the series presented in Figure 4.1 presents EPE as a percentage of GVA, and includes in both parts of the calculation only countries for which EPE data is available, it seems likely that intensity of EPE would not be expected to converge to the same rate in all Member States – therefore the inclusion (or not) of specific countries can serve to alter the perceived trend. This is true of all sectors (and is why missing data is specifically addressed within these charts in the light grey columns, measured against the secondary y axis) – however it is mentioned only here for the first sector studied.
From 2000 onwards, when more than half of EU Member States include data, EPE in the food, drink & tobacco sector stabilises at around 1.1-1.5% of industry value added. In addition, just less than 0.5% of industry value added is committed to investment in environmental capital every year. In nominal terms, while volatile, EPE shows an upward trend; however, this primarily reflects the inclusion of additional countries, and the steady expansion of the economic footprint of the sector.

Note: The two capital expenditure data are taken from two different Eurostat datasets covering different periods.

**Chemicals, pharmaceuticals, rubber & plastic products**

Rubber & plastics data is not specifically available for many years (it is in only one of the sources analysed); the analysis presented focusses instead on the broader sector, which includes chemicals and pharmaceuticals activity. In this broader sector, EPE has steadily fallen, both the OPEX and the CAPEX components. While these were around 6% and 2% of industry value added in 1996, by 2010 they had fallen to around 2% and 0.5% respectively. In nominal terms, EPE was very volatile over this period; but the primary driver of this is the country coverage; where large countries do not report data for specific years, they can heavily skew the nominal values (which is why such data is not included in this report, although it is in the Annex report).
Note: The two capital expenditure data are taken from two different Eurostat datasets covering different periods.

Metal products and other equipment
This sector includes motor vehicles (the focus of the relevant sector study considered in a subsequent chapter). Data on the OPEX component of EPE shows a slight decline over time, but the data is relatively volatile, and the trend could be overly influenced by a small number of data points. CAPEX shows a clearer and sustained downwards trend, however; from 0.4% of industry output in 1996 to less than 0.2% by 2016. In nominal terms, EPE grew relatively rapidly over the period from the mid-1990s; however, the contrasting trend in relative terms demonstrates that the expansion in EPE did not keep pace with overall growth in the industry.

EPE, both in CAPEX and OPEX terms, is substantially lower as a proportion of industry output than in most other sectors assessed in this analysis. However, the motor vehicles sector (C29, labelled in the chart below) shows a slightly higher concentrated of CAPEX than the broader sector, maintained at around 0.2% of industry value added in the period from 2008-16.
Electricity, Gas, Water Supply & Treatment

This sector includes the water supply sector which has a dedicated chapter later in this report. Data is more readily available for this sector, probably reflecting the stronger and more explicit environmental links in this sector; there are substantially fewer missing data points than for other sectors examined.

EPE across the broad sector is a consistent share of industry value added over the period assessed; operating expenses on EPE are around 4% of GVA, while capital expenditure shows a small decline in the early 2000s and a recovery afterwards, is above 2% of GVA for the majority of the time period.

The water supply sector, for which data is available over 2008-16, has a much higher concentration of EPE capital expenditure than the broader sector – it varies between 8 and 10% over the period available.

In nominal terms, EPE increases substantially over the period to 2010; but this reflects primarily the increasing value added of the sector; it falls substantially over 2011-13, reflecting largely a decrease in the country-coverage of the data.
In the area of waste and wastewater management, the EPE is well-aligned with the other aspects of the study; it is therefore possible to present detailed data for this specific sector covering the whole time period 1995-2016. As was the case with the water supply sector, there is also much less missing data for this sector, reflecting the importance of environmental impacts (and their mitigation) in the sector.

Despite this, overall levels of EPE are relatively small compared to other sectors studied. Operating expenses were 1% of sectoral value added in the late 1990s, and have since fallen steadily, to less than 0.6% in 2013. Capital expenditure has fared little better; it was just above 0.2% of GVA in 1995, and fell steadily in the period to 2016 (to around 0.1% of GVA).

Nominal expenditure largely shifts in response to the number of missing data points; it is otherwise largely steady, reflecting a sector which does not tend to experience large rates of economic growth.

The relatively small proportion of output that is attributed to EPE in this sector is surprising; it is likely that this reflects the fact that a large proportion of expenditure in the sector is related (either directly or indirectly) to environmental activities, and therefore formal EPE is low.

Note: The two capital expenditure data are taken from two different Eurostat datasets covering different periods.

Waste and wastewater management

In the area of waste and wastewater management, the EPE is well-aligned with the other aspects of the study; it is therefore possible to present detailed data for this specific sector covering the whole time period 1995-2016. As was the case with the water supply sector, there is also much less missing data for this sector, reflecting the importance of environmental impacts (and their mitigation) in the sector.

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Note: The two capital expenditure data are taken from two different Eurostat datasets covering different periods.
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More up-to-date data\(^\text{12}\) is available for waste management than for the other sectors of interest. These show that capex is €7 bn for industry, €11.8 bn for specialist producers and €10.6 bn for government, totalling €29.3 bn euros in 2017 at the EU level.

### 2.5 Conclusions

For the five sectors analysed in this study, a number of potential policy pathways have been identified that could reduce the environmental impact of production and use within each sector. These policies all contribute to achieving the broad objectives of the 7th EAP set out above. However, in this chapter, we have identified various cross-cutting challenges to the successful implementation of the policy pathways identified, and to achieving the environmental targets of the EU’s nationally determined contributions. In some cases, these challenges create the need for additional policy to overcome barriers to successful decarbonisation, such as improving accessibility to finance or providing more support to low-carbon innovation. In other cases, policy challenges are encountered because the policy leads to other effects within the economy, such as rebound effects or the reallocation of resources with sub-optimal GDP and employment outcomes.

The sector policies identified in this study all fall within three pillars of environmental policy response (Grubb 2014):

- Standards and engagement for smarter choices – includes regulations, standards, labelling and engagement to facilitate better, more environmentally-friendly choices.
- Markets and prices for cleaner products and processes – including carbon pricing and taxation plus subsidies.

\(^{12}\) From Eurostat series \([env\_ac\_pepssp]\), \([env\_ac\_pepsgg]\) and \([env\_ac\_pepsnsp]\) – it is not included in the charts to keep these consistent between sectors.
• Strategic investment for innovation and infrastructure – price incentives only partly guide the market for innovation and infrastructure. Public intervention is required to encourage investment which looks beyond short-term returns.

For the five sectors considered in this study, the sector-specific policy pathways may not include elements of all three policy pillars. However, the policies should be combined with policy pathways aimed at the wider economy, addressing all three policy pillars and overcoming the potential challenges identified within this study.

It is difficult to draw strong conclusions based on the available EPE data as country-level time series have considerably gaps and aggregation can heighten data inconsistencies across countries. Reported country-level EPE series have so many missing observations that it requires heavy interpolation to infer spending trends. On the other hand, aggregate analysis often suffers from inconsistencies caused by different EPE definitions and reporting practices used by member states.

From the nominal aggregate EPE series it can be inferred that EU15 countries continue to dominate EU28 spending, but the EU13 typically is spending more in relative terms, and therefore (slowly) closing the gap. Most of the visible trends and fluctuations in the nominal spending graphs can be explained by missing observations, thus their trends are not interpreted directly.

Capital and operating expenses can also be understood in relative terms, by comparing to industry value added. This mitigates the fluctuations caused by missing data and inflation and shows the relative magnitude of EPE across sectors. The normalized EPE series are typically stable over time, suggesting that sectoral EPE spending relative to industry performance has not changed much during the observed period. Capital expenses are in the range of 0.1-0.75%, while operating expenses are about 0.8-3% of industry GVA in the sectors analysed, for EU 28 countries in the 2000s, when data coverage is the most extensive. It may also be concluded, that total expenditure in environmental protection can be estimated at two to four times the size of capital expenditure. As would be expected, there are sectoral differences with the metal products and other equipment sector showing particularly low rates of EPE, and the Electricity, Gas, Water Supply & Treatment Sector exhibiting higher expenditures.
3 Food, drink & tobacco

The food and drinks industry is the largest manufacturing sector in Europe, in terms of both output and employment – it has an annual turnover of almost €1,100 billion (FoodDrinkEurope, 2017) and employs over 4 million people. It sells 90% of its produce within the single market (FoodDrinkEurope, 2017). It is a heterogeneous sector consisting of a large number of small family-based companies operating alongside global food conglomerates (EEA, 2017b). Although the sector’s contribution to the EU gross value added is relatively small (1.7%) (FoodDrinkEurope, 2017), it has a fundamental social and cultural importance in many European regions, and is the main source of income for some local communities (EEA, 2014, 2017b).

Tobacco product manufacturing is by contrast a small sector in Europe. In 2010, 261 tobacco product manufacturing enterprises were operating in the EU, employing just over 42,000 people (0.2% of the manufacturing workforce) and generating almost €7 billion of value added (0.4% of the EU manufacturing total). Over 527 million cigarettes were produced in the EU in 2016, with Germany being the top producer. Production has gone down over time, with about a third less than ten years ago.

3.1 The current environmental impact of the sector

While the European food, drinks and tobacco sector is in various ways dependent on the quality of the environment – on the whole – it is a significant source of environmental degradation and sustainability challenges generated throughout the economy. One important example is the sector’s energy use. According to the EEA, the amount of energy necessary to cultivate, process, pack and bring food to our tables accounted for 17% of the EU’s gross energy consumption in 2013, equivalent to about 26% of the EU’s final energy consumption that same year. Another example is the sector’s significant leakage of nutrients, particularly during agricultural production. Of the total input of nitrogen and phosphorus fertilisers, only 20-30% is actually embedded in the food that reaches consumers plates (EEA, 2017b).

Each step in the value chain of food, drinks and tobacco has some impact on the environment. The links to the environment can be broadly broken down into four distinct phases;

- The manufacture of the bio-based content of the goods (i.e. the food, drink or tobacco product itself), including production and processing
- The manufacture and use of packaging
- The trade and distribution of products
- The waste (including the bio-based content)

3.1.1 Overview

The agricultural production on land of food, drink and tobacco is reliant on an adequate supply of uncontaminated fertile soil, productive pasture land and timely and not excessive rainfall or irrigated water supply (see Figure 3.1). Marine and freshwater aquaculture is furthermore reliant on clean and healthy waters and feed-fish (often wild-caught).
The majority of EU agricultural land is used for food production either directly for human consumption, or as animal feed. Environmental impacts from the production of food through agriculture generally include soil erosion with consequent loss of carbon sinks; the removal of hedgerows, trees and other landscape features to increase room for agriculture with impacts both on carbon sinks and biodiversity; loss of farmland and soil biodiversity due to the harmful effects of pesticides, and emissions of greenhouse gases and pollutants to air and water from livestock and arable farming. In addition, large amounts of animal feed are imported from outside the EU where adverse environmental consequences such as deforestation are sometimes associated with their production (European Commission, 2013).

Whilst some food is sold directly to consumers at the farm gate, over 90% of food consumed is processed (ESF and COST, 2009). Food processing takes place in a wide variety of ways, but most processes require large inputs of energy and clean water. Direct environmental impacts related to food processing include organic wastes which are liable to generate greenhouse gases if improperly managed and organic contamination of wastewater which, if discharged into watercourses, can result in excesses of both biological oxygen demand and nutrients. Processed food is commonly packaged as it moves along the value chain towards the final consumer. In 2016, 40% of Europe’s total demand for plastics was used for packaging (Plastics Europe, 2016), of which a majority is only used once (Ellen MacArthur Foundation, 2017). Less than 30% of plastic waste in Europe is collected for recycling, and landfilling and incineration are the dominant approaches to manage post-consumer plastic waste (Plastics Europe, 2016). Meanwhile, littering and environmental leakage of plastic waste impose significant socioeconomic costs (Watkins et al, 2017).

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**Figure 3.1 Links between production and the environment in the food, drink & tobacco sector**

<table>
<thead>
<tr>
<th>Key inputs/ resources from the environment</th>
<th>Key transformations/ processes</th>
<th>Key outputs/ emissions to the environment</th>
<th>Examples of available data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Production</td>
<td></td>
<td>Land use change</td>
<td>• GHG emissions from EU agriculture: 10% of EU’s total emissions (Eurostat, 2015 data)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>GHG emissions (CO₂, NO₂, etc.)</td>
<td>• Production of a 0.5 litre bottle of carbonated soft drink estimated to require 250-300 litres of water (Erlich et al, 2011)</td>
</tr>
<tr>
<td>Processing</td>
<td></td>
<td>Air pollution (NOx, etc.)</td>
<td>• EU food processing accounts for 28% of total energy used in food production (2013 data)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Loss/degradation of habitat</td>
<td>• Food processing is the 2nd largest contributor to food waste in EU; 28-17 million tonnes (2015 data)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pollution of water courses/ groundwater</td>
<td>• BOD in food processing wastewater can be 10-100 times higher than domestic (2006 data)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Soil pollution, degradation and erosion</td>
<td>• **</td>
</tr>
<tr>
<td>Trade and distribution</td>
<td></td>
<td></td>
<td>• Europeans throw away over 30kg of plastic packaging per person per year.</td>
</tr>
<tr>
<td>Waste management</td>
<td></td>
<td></td>
<td>• Food waste, European average 1.73 kg per person (2014 data).</td>
</tr>
</tbody>
</table>
Energy use in food processing is a major part of the environmental impact. In International Energy Agency (IEA) member countries, manufacturing is the second largest consumer of energy (27%), of which food and tobacco manufacturing is the fourth largest part (12%) (IEA, 2017). In the EU, food processing accounts for 28% of total energy used in food production (2013 figures) (Monforti-Ferrario et al, 2015). Of a total final energy consumption of about 28,300 ktoe in 2013, the food and beverage industry used an estimated 62% for process heating, 34% for electrical use and 10% for process cooling. While energy efficiency in the EU food, drink and tobacco sector has improved over time (IEA, 2010), efficiency gains in the sector are often offset by increasing production (IEA, 2010). In 2013, renewables accounted for only 7% of the energy used in food production and distribution in the EU, compared to 15% in the overall energy mix (Monforti-Ferrario et al, 2015).

Water use also contributes to the environmental footprint of the sector. Much of the water used in food processing is required to be of drinking water quality to avoid contamination, according to the EU rules on hygiene of foodstuffs. It therefore competes with the water needs of cities and local communities (Meneses, Stratton and Flores, 2017). A US study finds that a zero discharge process is possible in food processing by reducing the need for process water and reusing it in a closed loop system for those processes for which it is required (Lee and Okos, 2011). According to WRAP (2013), industrial sites in the UK, including the food and drinks industry, that have not previously tried to save water can often reduce their water and effluent bills by up to 30% by combining low and no-cost options with longer term water saving projects.

The processing sector is meanwhile the second largest contributor to food waste in the EU-28 (FUSIONS, 2015), accounting for 19% or 17 million tonnes (Secondi, Principato and Laureti, 2015). As food processing uses significant amounts of water, it also generates high volumes of wastewater. Food processing wastewater is often high in biochemical oxygen demand (BOD) – levels can be 10 to 100 times higher than in domestic wastewater (European Commission, 2006). BOD is the amount of dissolved oxygen required by microorganisms to decompose the organic matter present in water. Increased BOD reduces the amount of oxygen available to other organisms which can cause hypoxia which can be harmful to aquatic ecosystems. Organic matter can also pollute drinking and bathing water (EEA, 2017b).

3.1.2 Meat

Agricultural production

One of the food products with the greatest environmental impacts is meat (beef, pork and poultry) (Notarnicola et al, 2017). The main environmental impacts of meat are related to livestock production. In the EU context, these impacts are:

- Greenhouse gas emissions, especially CH₄ from enteric emissions from ruminants (the largest single source of CH₄ in the EU-28 (Eurostat, 2017a)) and from manures, loss of soil carbon from overgrazing, and CO₂ emissions from transport taking animals to and from markets and to places of slaughter;
- Loss of biodiversity when natural pastures are “improved” to allow more intensive livestock rearing. Conversely, extensive livestock rearing can help to conserve and increase biodiversity compared to a counterfactual of land abandonment although whether this happens is case-specific;
- Air pollution from NH₃ associated with manure management, and
- Water pollution with nitrate (NO₃⁻) also associated with manure management.

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13 All OECD countries apart from Chile, Iceland, Israel, Mexico and Slovenia [https://www.iea.org/countries/membercountries/](https://www.iea.org/countries/membercountries/)

14 In some circumstances non-drinking water (i.e. non-potable) is used by the food industry (e.g. for fire control, steam production). In these instances the water should be clearly identified as non-drinking water and not connect or mix with the drinking water supply used directly in food production [http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2004:139:0001:0054:en:PDF](http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2004:139:0001:0054:en:PDF).
An important indirect environmental impact of EU livestock – and pig meat in particular – arises because a high proportion of protein used in animal feed is imported. Such imports have the potential to cause degradation in exporting countries, for instance through land use change.

**Food production**

The most important EU meat products are pig meat, beef and veal, poultry meat and meat from sheep and goats. In 2017, the EU produced 23.5 million tonnes of pig meat, 14.7 million tonnes of poultry meat, 8.1 million tonnes of beef and veal and just over 900,000 tonnes of sheep and goats' meat. Animals are reared in very different circumstances but as a general rule the monogastric animals are reared indoors in intensive, enclosed facilities whilst ruminants such as cattle, sheep and goats are reared outdoors with varying degrees of intensity. In particular cattle may be reared “extensively” on semi-natural pasture land where the farmers’ activity is limited to occasionally hay cutting and scrub clearance, or “intensively” on so-called improved grassland where mineral fertiliser has been used to increase the production of grass in order to support more animals.

EU pig meat is very competitive on world markets and over 10% was exported in 2017. Although average EU prices for other meat products including beef and veal are close to world levels the market for beef is protected by very high tariffs and trade liberalisation discussions invariably arouse concerns about the future of the EU’s beef sector.

The environmental impacts of meat are vastly different depending on the type of meat. In terms of land use, poultry and pig meat have a mean demand of 12 at 17 m²/kg of meat, compared to 326 and 369 m²/kg for beef and lamb respectively. Similar are the differences in terms of GHG emissions, where a kg of meat delivered to the customer is associated with 10 and 12 kg CO₂eq for poultry and pig meat and 40 and 100 kg CO₂eq for lamb and beef respectively. Slightly different is the demand for freshwater, which starts from 660 l per kg of poultry and reaches 1451 l for beef, 1796 l for pig and 1803 l for lamb.

Table 3.1 Environmental impacts per kg of meat

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Beef</th>
<th>Sheep and goat</th>
<th>Pig</th>
<th>Poultry</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>m²</td>
<td>326.2</td>
<td>369.8</td>
<td>17.4</td>
<td>12.2</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kg CO₂eq</td>
<td>99.5</td>
<td>39.7</td>
<td>12.3</td>
<td>9.9</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>g SO₂eq</td>
<td>318.8</td>
<td>139.0</td>
<td>142.7</td>
<td>102.4</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>g PO₄³eq</td>
<td>301.4</td>
<td>97.1</td>
<td>76.4</td>
<td>48.7</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>l</td>
<td>1451</td>
<td>1803</td>
<td>1796</td>
<td>660</td>
</tr>
</tbody>
</table>

Extrapolated on a European scale, this would translate as the following total environmental impacts during the lifecycle of meat production in Europe\(^\text{15}\):

Table 3.2 Total environmental impacts of meat production in Europe

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>km²</td>
<td>348 792</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kt CO₂eq</td>
<td>1 254 469</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>t SO₂eq</td>
<td>7 495 703</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>t PO₄³eq</td>
<td>4 976 347</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>billion m³</td>
<td>65 034</td>
</tr>
</tbody>
</table>

**Meat substitutes**

Some meat alternatives offer potential benefits in terms of reduced energy consumption and land use requirements, though this is not uniform for all meat substitutes (Alexander et al., 2017\(^\text{8}\)). While meat alternatives have the potential to lower the environmental impacts of global meat consumption, they have a market share of 3-5% in Europe (MINTEL, 2013), though growth is expected, with one estimate purporting

\(^{15}\) Based upon EU production figures from FAOSTAT for 2016
the global protein analogue market to be worth approx. 37 billion EUR by 2020 (Business Wire, 2018). A key determinant of meat substitute acceptance is familiarity, which is two-faceted – on one hand, consumers (esp. non-users) prefer products that are more similar to meat in terms of sensory properties (taste, texture, smell etc.); on the other hand is “neophobia” (extreme dislike for new & unfamiliar things), which has been found to not differ between non-users and heavy users of meat substitutes (Hoek et al., 2011). In general, plant and mycoprotein-based meat alternatives have an advantage over lab-grown and insect meats due to the issue of neophobia, but still suffer from the stigma of not being an alternative to “real” meat in the eyes on non-users (ibid.).

Apart from mycoproteins & plant-based alternatives, insect protein is a promising alternative which faces larger consumer resistance – multiple studies in the EU have shown a low level of willingness to eat insects, with the only socio-demographic factor so far identified as an influence being gender, with men reacting somewhat more positively to insects as food than women (Hartmann and Siegrist, 2017). Positive taste experiences (and thus increased familiarity) have been shown to increase acceptability, e.g. via bug tastings or “bug banquets”, though consumers with particularly negative attitudes may not be swayed (Hartmann and Siegrist, 2016). Some studies have found that incorporating insects into familiar dishes such as salads or spaghetti can be a good strategy for increasing acceptance of unfamiliar foods (Schosler et al., 2012), though even more effective is processing that removes negative visual stimuli (Hartmann et al., 2015). In any case, first impressions matter, and increased future aversion is shown when insect products are not sensorially satisfying the first time they are tried (Schouteten et al., 2016).

There are less studies on the acceptability of cultured meat but in general, the emergent pattern is that determinants of acceptability include 1) attitude toward the morality of the technology; 2) expectations of the product itself – its likeness to meat, as well as health considerations, i.e. whether consumers believe that the product is “safe” (Verbeke et al., 2015). While there has been much ado about the promise of cultured meat as a replacement for conventional meat, Stephens et al. (2018) argue that at this stage this may be premature due to a lack of general outlook on what a mass-market cultured meat sector would look like - in terms of resource, environmental and ethical requirements.

3.1.3 Sugar

Agricultural production

The main environmental impacts associated with the production of sugar are:

- Water consumption, particularly in non-rain-fed systems such as beet production in Southern Europe and cane production;
- Soil erosion;

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Pollution and loss of biodiversity from the use of pesticides. However, beet is usually grown as part of a rotation with wheat or other cereals, acting as a break crop and reducing the inputs needed for those crops;

- Transport emissions and noise, and
- Factory emissions (restricted to BAT).

A study which examined 18 different systems for growing and processing sugar beet in the UK estimated average environmental impacts per hectare of sugar beet as 21.4 Gj energy consumption; 1.4 tonnes CO₂eq; 3.3 kg N leakage; and 15.2 kg N lost through nitrification (Tzilivakis et al, 2005). A water footprint study has estimated that 150-300 litre of water is required to produce a 0.5 litre PET-bottle of soft drink, concluding that agricultural ingredients of the soft drink have the biggest share of the total water footprint (Ercin, Aldaya and Hoekstra, 2011).

**Food processing**

Sugar is made by processing either sugar beet (an arable crop usually grown in rotation with other crops, such as wheat in cooler areas such as Northern Europe, but also under irrigation in drier parts including Spain) or sugar cane (a permanent crop grown in tropical regions including a number of France’s Overseas Territories which are part of the EU). It has a variety of uses. Around 12% of EU sugar beet production in 2017 was processed into ethanol – a first generation biofuel.

Both cane and sugar refineries are regulated under the Industrial Emissions Directive if – as is usual – they have a thermal capacity of at least 50 MW. Their direct impacts are as a result required to comply with Best Available Techniques and are not further considered in this note.

Biofuels are produced from sugar beet or cane (among other feedstocks) by distillation in which case the crystallisation process is omitted.

In 2017, EU farmers devoted 1.7 million hectares to sugar beet production, producing 105 million tonnes of sugar beet. 13 million tonnes were sold for the production of ethanol with 92 million tonnes destined for the food market.

20.5 million tonnes of sugar were produced. Approximately 0.9 million tonnes went to industrial users for pill coatings etc., with a small quantity sold as granulated sugar directly to consumers. The remainder was bought by food and drink producers.

In terms of environmental impacts, a recent meta-analysis (Poore and Nemecek, 2018\(^{23}\)) calculated the following global mean environmental impacts per kg of beet and cane sugar delivered at the customer, with cane sugar having higher values compared to beet sugar (respectively below):

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Beet sugar</th>
<th>Cane sugar</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>m²</td>
<td>1.8</td>
<td>2.0</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kg CO₂eq</td>
<td>1.8</td>
<td>3.2</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>g SO₂eq</td>
<td>12.6</td>
<td>18.0</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>g PO₄³⁻eq</td>
<td>5.4</td>
<td>16.9</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>l</td>
<td>218</td>
<td>620</td>
</tr>
</tbody>
</table>

Extrapolated on a European scale, based on FAOSTAT data on beet sugar production for 2014, this would translate as the following total impacts during the lifecycle of sugar production in Europe\(^{24}\):

\(^{23}\) Reducing food’s environmental impacts through producers and consumers, Poore, Nemecek, DOI: 10.1126/science.aaq0216

\(^{24}\) Based upon EU production figures for beet sugar production from FAOSTAT for 2014
Table 3.4 Total environmental impacts of sugar production in Europe

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>km²</td>
<td>3 549</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kt CO₂eq</td>
<td>35 103</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>t SO₂eq</td>
<td>245</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>t PO₄eq</td>
<td>105</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>billion m³</td>
<td>4.222</td>
</tr>
</tbody>
</table>

These impacts are associated with the full lifecycle of sugar production, including from activities occurring outside Europe - beginning with the extraction of resources needed to produce inputs for agricultural production, the initial impact of choice by farmers, and ending at the retail store, the point of choice for consumers.

3.1.4 The socio-economic footprint of the sector

The European food, drinks and tobacco industry is the largest manufacturing sector in the EU with an annual turnover of almost €1,100 billion (FoodDrinkEurope, 2017). The industry is characterised by great heterogeneity, with a large number of small family-based companies operating alongside global food conglomerates (EEA, 2017b). The industry is also the leading employer in the EU, with over 4 million employees, selling 90% of its produce within the single market (FoodDrinkEurope, 2017). Although the sector’s contribution to the EU gross value added is relatively small (1.7%) (FoodDrinkEurope, 2017), it has a fundamental social and cultural importance in many European regions, and is the main source of income for some local communities (EEA, 2014, 2017b).

There are over 14,000 aquaculture enterprises in the EU, directly employing 85,000 people in total. The value of EU aquaculture production increased by over €300 million in 2015 compared to 2014, reaching the highest values ever registered. While almost all main species farmed reported a value increase in 2015, the main drivers were Bluefin tuna (+€53 million), salmon (+€52 million) and oyster (+€44 million) (European Commission, 2017). Notably, Bluefin tuna farming involves fattening of wild-caught fish. Although the quota for Bluefin tuna fishing in the Atlantic has increased in recent years, the stocks have only just begun to recover thanks to intensive conservation efforts, and any exploitation of the stocks must be carefully managed in accordance with scientific advice.

3.1.5 Environmental pollutant releases

Between 11 and 16% of all pollutant releases recorded in the European Pollutant Release and Transfer Register (E-PRTR) for the food, drink & tobacco sector have sufficient data for the calculation of environmental impacts25.

The impacts upon human and ecosystem health from food, drink & tobacco sector pollutant releases largely mirror each other. Health impacts from pollutants peaked in 2009, and have been falling steadily since; by 2016, human health impacts were 35% lower than in 2009.

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25 The key characteristic that is often missing is the CAS pollutant code, which is required in order to assign a human health impact (measured in disability-adjusted life years) and ecosystem health impact (measured in species years lost).
Over the period 2007-16, the health impact of recorded pollutant releases from France were more than twice as much as any other Member State (at 31,212 DALYs cumulatively over the period, compared to 14,922 in Germany and 14,180 in the UK). As in the other sectors studied, health impacts are dominated by releases of CO2, due to the high volume of it released via air (though it should be noted that these are indirect via GHGs’ effect on climate change and are cumulative over a 100-year time horizon due to the nature of the Global Warming Potential). The next largest sources of (quantified) impacts were carbon monoxide and methane – but while carbon dioxide emissions were typically 8-10 million tonnes per year, these emissions were only tens of thousands of tonnes.

Figure 3.2 The environmental impacts of pollutant releases from the food, drink & tobacco sector

### 3.2 Current direction of travel

The heterogenous nature of the food, drink & tobacco sector makes identifying the impact of changing preferences and business models challenging. However, there is evidence of growing consumer awareness and demand for greater transparency of environmental performance of different food and drinks on the market. A 2010 Eurobarometer survey indicated that up to 40% of Europeans were willing to pay more for products whose production preserves the environment, respects social conditions or helps developing countries (Eurobarometer, 2010). Diets and markets for different food and beverages can reflect how consumers and industry are responding to growing environmental awareness.

In the EU, demand for products labelled as organic is arguably a reflection of this reality. In 2016, the market for organic food was estimated to be worth 30.7 billion in the EU. Sales of organic food and dedicated organic farmland have increased by 47.7% and 18.7% respectively between 2012 and 2016 (European Parliament, 2016). Support for fair trade products is also increasing. The 2014 annual report from Fair Trade International showed that global Fair-Trade certified sales reached EUR 5.9 billion, also experiencing year on year growth (WFTO, 2014).

Diets and food preferences also reflect consumer demand, and are also one of the pathways to reducing the environmental impacts from the food and beverage sector. As outlined, vegetarian and vegan diets
could significantly reduce the environmental impact of the food sector, primarily due to the land and carbon intensity of livestock and animal agriculture (Vettese, 2018).

Overall, European Commission data on per capita consumption for meat, fish and dairy suggests gradually declining meat consumption between 2015 and 2025. This is anticipated due to growing social concerns over animal welfare, human health and the environment. These trends are disaggregated for different products – for example fish, poultry and cheese are expected to experience increasing demand over that period. EU trends should, however, be put in the context of the global trend which suggest a growing demand for meat products. This will likely result in a greater export of EU meat products (European Commission, 2015).

3.2.1 Current environmental policy

Food, drink and tobacco production depends on the availability of a range of environmental resources and ecosystem services. Clean water, productive soils and pollinators are especially important. EU legislation and policy underpinning these public goods is therefore an important component of the food production system.

The most relevant EU environment policies are the Water Framework, Groundwater and Nitrates Directives which underpin the availability of clean water; the Common Agricultural Policy’s agri-environment climate measure, cross-compliance and greening measures, which help to preserve healthy soils, and the Pollinators Initiative under which the Commission and Member States have committed to a range of actions to support wild pollinators. More broadly, the Natura 2000 Directives, covering birds and habitats, protect a range of species and habitats providing other ecosystem services to food production, including pest control.

Sugar

Like other food and drink production, sugar production benefits from clean water, healthy soil and ecosystem services including pollination. No environmental resource needs are unique to sugar production but the industry benefits from imports of cane sugar which are governed by trade agreements.

As transport is another important source of environmental impact for the drinks industry, environmental policies impacting heavy road transports and incentivising more effective/environmentally benign transport options might have an impact on resource inputs of the drinks industry.

Finally, policies to incentivise healthier soft drinks may contribute to shifts from natural sweeteners to artificial sweeteners. Research suggest, however, that also artificial alternatives can have negative impacts on the environment, including unforeseen consequences for aquatic life (Borges et al, 2017).

Meat

The environmental resources used in livestock production include land (particularly grassland for ruminant livestock), nutrients such as nitrogen and phosphorus, animal feed (usually cereal-based) and energy to provide heating, lighting and transport (e.g. to market). The role of EU environmental policy in underpinning these is as follows:

- Restrictions on ploughing permanent grassland under the CAP help to ensure that the supply of land suitable for ruminants is maintained;
- Restrictions under the Nitrates Directive on the allowable concentrations of manure in Nitrate Vulnerable Zones incentivise alternative uses for manure, such as energy production, which can improve the economic viability and sustainability of livestock production.
- The Industrial Emissions Directive, which applies to the largest livestock units, helps to spread the use of best available technology which can have a positive impact on business viability, although as its primary purpose is the reduction of pollution it may also entail extra cost.

Policies encouraging more plant-based diets in Europe could contribute to significantly lower environmental impacts of food. Westhoek et al (2014) suggest that replacing 50% of animal-derived foods with plant-based foods would achieve a 40% reduction in nitrogen emissions, 25-40% reduction in greenhouse gas emissions and 23% per capita less use of cropland for food production.
3.2.2 The impact of changing consumer preferences

The scope of this review does not allow a comprehensive assessment of the impacts of consumer preferences on the largely heterogeneous range of products associated with the European food and drink sectors. However, rising pressure on healthcare systems along with a recognition that many consumers eat meat and dairy products well in excess of dietary guidelines suggests that national governments may eventually seek greater influence over consumer diets. This is likely to bear particularly heavily on production of sugar, dairy and meat products.

Consumer preferences for food are influenced by a range of factors, including psychological and marketing factors (see Figure 3.3). Many of these factors have been widely studied (Font-i-Furnols and Guerrero, 2014).

In general, European consumers consider meat to be a healthy and important component of the diet and have a negative view of excessive manipulation and lack of naturalness of beef production (Verbeke et al, 2010). In fact, health appears to be the primary reason for shifting diets and reducing meat in the diet or avoiding meat altogether (EEA, 2017a; Latvala et al, 2012). Further, a British study has found UK country of origin food labels to be the highest valued food label attribute for the fresh/chilled/frozen (i.e. not processed) meats (excluding chicken) (Fraser et al, 2015), and Verbeke and Ward (2006) show that meat quality and origin become more important for consumers after having been exposed to information campaigns.

Globally, consumption of meat is projected to continue to increase, driven mainly by poultry (Henchion et al, 2014). Consumer trends in meat demand are associated with higher incomes (OECD, 2018), however some research indicates that the influence of price and income factors on meat consumption is likely to decline over time while other factors, such as quality, becomes more important (Henchion et al, 2014). Further, the millennial generation has been associated with trends of reducing meat consumption for ethical and lifestyle reasons (Bord Bia Insight Centre, 2016).

Source: Font-i-Furnols and Guerrero (2014)
A number of recent reports have identified that even current levels of EU livestock production exceed the carrying capacity of the environment in respect of a number of pollutants. Buckwell and Nadeu, for example, (Buckwell and Nadeu, 2018) attempt to estimate the “safe operating space” for livestock production and conclude that quantitative reductions in meat consumption will be required in addition to achievable improvements in production itself. Policy means to deliver such reductions in consumption – along with a shift towards diets higher in plant-based foods – does not appear to be developed to a significant degree anywhere in the EU, although the issue is beginning to be taken up by NGOs.

Health is similarly a key driver for consumption of sugar. Globally, consumer behaviour has shifted in line with an ambition to eat healthier foods and live healthier lifestyles, including a rising demand for products free from sugar (Kelly et al, 2018). As one example, according to data based on consumption and shopper panels, purchase levels of sugar-rich carbonated drinks dropped 8.6% among all British consumer groups between 2015 and 2016. The authors associate this drop with health preferences. The same research concluded that consumer concerns about sugar has impacted sales of biscuits and chocolate (Quick, 2016). Multi-nationals have adopted various commitments to reduce the amount of sugar in their products or continue the shift to non-sugar sweeteners, including Nestle and Coca Cola.

The EU is the second largest consumer of sugar, preceded only by India (USDA, 2018). Consumption of sugars has increased over time but changing consumer preferences and increasing health concerns are expected to cause a 5% reduction in sugar consumption by 2030 (in favour of isoglucose and other sweeteners). Meanwhile, the use of sugar beet and molasses for biofuel production is projected to increase slightly, mentioned as a possible gateway for directing sugar oversupply following the 2017 end of quotas (European Commission, 2017a).

A report by DSM Food Specialities (2015) suggests that the sizeable increase in consumption of sugared dairy products around the world could be a result of dairy products generally being perceived as nutritious options and preferable to other snacks. This would add to the understanding that general health concerns about sugar have complex interactions with consumption patterns. Popkin and Hawkes (2016) argue that, in the absence of intervention, sugar content in various foods worldwide will follow the same increasing pattern as has been seen in the US, where 74% of products in the food supply chain today contain caloric or low-calorie sweeteners, or both.

3.3 Future policy priorities

3.3.1 Links between the sector and the Sustainable Development Goals

Food and drink as a manufacturing sector and as a source of environmental challenges is relevant to all of the UN SDGs (for example, the quality of education may help to determine choices as to what food is grown and eaten, how it is grown and how any residue is disposed of). SDG 2 and 3 have the strongest links, as they summarise the needs which the production of food and drink seeks to satisfy;

- SDG 2 – End hunger, achieve food security and improved nutrition and promote sustainable agriculture
  - Targets 2.1 and 2.2 on ensuring access to food and ending all forms of malnutrition can be linked to the use of agricultural inputs (such as fertilisers and pesticides) to ensure sufficient and reliable harvests in parts of the world with limited production. However, in Europe, the primary driver for lack of access to food and of malnutrition is poverty (FAO, 2017).

27 See for example the work of the UK-based Eating Better Alliance https://www.eating-better.org/
Target 2.3 on doubling the agricultural productivity and incomes of small-scale food producers has links to supporting Europe’s large number of small family-based companies. In terms of the pursuit for more sustainable food and drink systems in Europe, the most appropriate size of food production is a much-debated matter, and likely depends on a range of factors, including the type of produce in question.

Target 2.4 on sustainable food production provides the clearest link between the sector and its environmental performance. Although it does not explicitly mention links between sustainable food production and biodiversity, nor the role of agriculture in mitigating climate change, it does provide an incentive for adopting policies in support of low-impact agricultural production.

Target 2.5 on maintaining plant and animal genetic diversity has indirect implications for food and drinks production in that it has relevance for the ways in which crops are produced and agricultural land is managed.

SDG 3 – Ensure healthy lives and promote well-being for all at all ages

- The links between food and drink and SDG 3 are more indirect in that healthy diets can promote well-being.
- Target 3.9 on pollution to air, water and soil has perhaps the most direct link to food and drink production, as all stages of the value chain tend to generate some degree of pollution and contamination. Conventional agriculture has perhaps the clearest links to pollution of soils with potentially hazardous chemicals, whereas BOD contamination of waste water in food processing is a potential health hazard if not managed appropriately.

The following SDGs all require, inter alia, that food and drink are produced, consumed and disposed of in ways that care for the Earth’s resources and environment:

- SDG 6 – Clean water and sanitation
- SDG 7 – Affordable and clean energy
- SDG 12 – Responsible consumption and production
- SDG 13 – Climate Action
- SDG 14 – Life below water
- SDG 15 – Life on land

Tobacco is a threat to SDG 3 (the goal of good health) and also, indirectly as a result of death and illness, to SDG 1 (Zero poverty).

3.3.2 Policy options

The following potential policy options have been explored as part of this project:

Introducing a carbon tax on produce

Whilst the largest energy producers in the food, drink & tobacco sector itself fall under the EU ETS, inputs from agriculture do not. If a tax were levied on these inputs, it would incentivise low-carbon production methods and ultimately reduce the (indirect) environmental impact of food.

In this project, the introduction of an emissions tax applied to agricultural produce purchased by the food, drinks and tobacco sector was modelled. The emissions tax was applied to all GHG emissions, not only CO₂, with methane being a key GHG in agriculture. The tax rate was set equal to the EU ETS price by 2030: €34.25 per tonne of CO₂e, this was applied to each food group as an exogenous increase in price. The E3ME baseline uses the International Energy Agency, World Energy Outlook 2016, Current Policy Scenario values.

The incidence of such a tax would have to be carefully designed. Applying the tax to purchases by the food, drink & tobacco sector, rather than to sales from agriculture, means that European manufacturers would face the tax regardless of the point of origin of the agricultural produce, and reduces the risk of carbon leakage in the supply chain.
In the macro modelling undertaken, the first order effect of the tax is an increase in the costs of the food, drink & tobacco sector, leading to a reduction in output of 0.22% by 2030. However, net economic impacts are positive, with GDP 0.15% higher and employment 0.04% higher, as a result of the revenue balancing assumption – that increased tax take by government is balanced by a reduction in income tax rates of the equivalent monetary value.

A key effect in this carbon pricing scenario is an increase in consumer food prices. Consumer food and drink prices increase by 1% in this scenario by 2030. This estimate is to some extent a ceiling because it is assumed there is no change in diet; if consumers choose to adjust dietary choices, to products with lower embedded emissions, food and drink price increases would be lower. An increase in food prices is regressive, and the tax could therefore be difficult to implement. The targeted recycling of carbon tax revenues has potential to mitigate negative distributional impacts.

The environmental impacts are limited:

- Whilst consumers reduce spending on food and drink, expenditure is substituted to other consumer categories, increasing emissions throughout other parts of the economy. (so-called rebound effects)
- Assuming fixed diets and no substitution to less emissions-intensive food, the total production of the food, drinks and tobacco and agricultural sectors fall, by 0.22% and 0.21% respectively by 2030. However, given model limitations, there is no change in production across food types. Carbon emissions in the food, drinks and tobacco sector fall by 0.5% by 2050.

Land use change emissions are not captured, but they would be expected to be small in this scenario.

Extended producer responsibility for food and drink producers

The costs of excessive food (and other) packaging are to an extent internalised through the existing system of packaging waste legislation but the same is not true for the externalities associated with the supply of food itself. Actions taken by the food, drinks and tobacco sector to reduce food waste could include:

- Measures to discourage over-purchasing by consumers – e.g. removing multipack promotions and misleading advertising
- Shortening supply chains – e.g. sourcing from local agricultural suppliers to reduce transit losses and emissions, provide fresh produce (package-free where applicable), and support the local agri-economy
- Provide clearer messages to consumers on the storage, preparation and consumption of products – e.g. greater clarity on date labelling
- Optimised packaging to extend shelf life on high value (and high environmental impact) products – e.g. dairy and meat (avoiding unnecessary application of single use plastics)
- Utilise edible products close to their “use by” dates to generate value added for example in restaurants, or donate products to food banks.

A scenario was modelled in E3ME, to estimate the potential impacts of the elimination of household food waste by applying a waste tax of €50 per tonne, in 2016 prices. Economic impacts were found to be relatively small assuming no additional policies to reduce food waste. Unit costs in the food, drinks and tobacco sector increase by an average of 0.25% across the EU in 2020, and, while output of the agriculture and food, drinks and tobacco sectors is reduced marginally (by 0.01%), overall GDP is increased as a result of the balancing of government revenues through reducing income taxes (thereby shifting the incidence of tax slightly away from consumers and towards manufacturers).

However, implementation of this policy would be difficult. It would likely not be possible to trace household waste by manufacturer source. The tax would likely have to be industry-wide, calculated as a function of

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28 The edible share of food waste is estimated at 60%. To give a monetary value of waste, physical weight of edible food waste is multiplied by an estimate of per kg retail value of edible food waste (€3596 per tonne). Population data are then used to yield a total value of household food waste by Member State.
total household food waste collected/reported. This formulation, however, would severely blunt incentives for individual manufacturers to act.

**Excluding the production of sugar from CAP support**

Reduced EU production of sugar could reduce pressures to plough permanent grassland for arable use, avoiding significant GHG emissions associated with the release of carbon sequestered in the disturbed soil. Yet, sugar producers benefit from CAP support – even though Governments are increasingly trying to deter the consumption of sugar through taxation. Excluding the production of sugar from CAP support could be expected to reduce EU growers’ willingness to grow these crops, with knock-on impacts including higher imports of sugar. This carbon leakage could lead to increased overall environmental impacts; as cane sugar has, at a global average, 75% higher GHG emissions than sugar beet (Poore & Nemecek, 2018).

**Minimum harmonised public procurement rules**

This would serve to encourage healthy and sustainable food in public institutions, including limiting the amount of red meat and sugary drinks purchased with public funds and provided in public contexts. Certain Member States have developed different criteria for different food categories, including meat served in schools (Caldeira et al, 2017). The total magnitude of effect of such a policy is likely to depend on behavioural effects; the extent to which it delivers a wider dietary shift by consumers. Upstream effects of this policy on agricultural production are likely to be a shift of agricultural production in the EU from meat and sugar to more sustainable produce. The effect on the food, drinks and tobacco sector would be a shift in agricultural products processed.

**An EU-level requirement for variable charges for waste**

There is strong evidence for the deterrent effect of variable charges for household waste (of which a high proportion is food and food packaging) on householders. Charges for residual household waste have the potential to boost participation in recycling including through deposit refund schemes but could also have negative consequences including distributional ones.

Modelling the macroeconomic impacts of such a policy led to modest increases in GDP and employment in the EU (of 0.08% and 0.06% respectively in 2030), suggesting that there is no net economic cost to reducing food waste, although there were negative impacts in sectors associated with the food supply chain. In addition, there are clearly benefits in terms of reducing the environmental impact of food production and increasing utility for consumers (by allowing a switch in consumption away from wasted food and towards other goods and services).

The environmental impacts of this scenario are only partially captured in E3ME, given that there is no treatment of land use in the model. Levels of energy related carbon emissions change very little under elimination of food waste: by 2030 across the EU28 they increase by 0.85 million tonnes (0.03% compared to baseline). The reason for the increase in emissions is that the redistribution of consumer expenditure from food to other sectors pushes consumption to higher carbon products. Similar results were found in Martin van de Lindt et al. (2017).

Whilst land use change emissions are not measured in E3ME, it is possible to estimate the unaccounted environmental impacts. The reduction in land use emissions arising from elimination of the household food waste modelled is approximately 22.1 million tonnes CO₂ equivalent, by far outweighing the increased emissions in the E3ME model. This value corresponds to embedded emissions in EU consumption, not to production located in the EU.

### 3.4 Conclusions

The food, drinks and tobacco and agricultural sector is rather heterogeneous with a substantial environmental impact (both directly, in terms of product manufacture, and indirectly, through demand for agricultural produce). Consumer preferences have a substantial role to play in altering this; a key challenge for the sector is meeting demand for new products (such as meat substitutes).

A number of potential policies have been identified which could play a substantial role in changing the environmental footprint of the sector. Such policies typically have a relatively small economic impact at the macro level; however, the distributional impacts of these policies should also be considered. Since many of
the policies lead to higher prices in the sector (as a result of introducing taxes to shift production away from environmentally damaging behaviours which are typically also the cheapest to pursue), there is the potential for such policies to be regressive, given that poorer households spend more than average of their income on food, drinks and tobacco products.
4 Plastics

The rubber & plastics sector is a manufacturing sector that generates environmental impacts through use of materials and energy, as well as the generation of waste and emissions. The sector presents a number of opportunities for environmental policy to alter its environmental impact, including circular economy and issues of recyclability/reusability of products, as well as impacts of marine litter and the specific challenge of single use plastics.

4.1 The current environmental impact of the sector

4.1.1 Overview

The plastics sector is a large and economically important sector that is closely coupled to fossil fuel production which provides the sector’s primary feedstock. Rapid growth in the plastics sector has taken place over the course of the last 50 years, notably in the packaging sector and for single use plastics which account for a significant proportion of plastic applications. Mismanaged plastic waste and leakage to the terrestrial and marine environment have become a global sustainability crisis.

90% of the plastics produced globally are derived from fossil fuels (Bourguignon, 2017). In total, plastics consume 8% of the global oil production, 4% of which is used as raw material (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016). Plastics production has increased significantly since the mid-twentieth century, reaching 322 and 58 million tonnes in 2015 globally and in Europe respectively (Plastics Europe, 2016).

Plastics manufacturing requires large amounts of energy. The main processes for production are polymerisation and polycondensation, which require specific catalysts (Bourguignon, 2017). The production process is responsible for approximately 4% of fossil fuels (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016) and in 2012, CO2 emissions generated from plastics production amounted to 390 million tonnes (WEF, EMF and McKinsey&Company, 2016). Global plastics production has grown significantly, and it is expected to reach 1.2 billion tonnes annually by 2050 (Plastics Europe, 2016).

Total plastics demand in Europe amounted to 49 million tonnes in 2016. Packaging represents the biggest share, with 39.9%, followed by building and construction (19.7%), automotive (8.9%), electric and electronic (5.8%) and agriculture (3.3%). The rest of the market is dedicated to other materials (22%) (Plastics Europe, 2016). Plastics consumption can lead to a loss of material value due to single-use and low recycling rates (Bourguignon, 2017). This is particularly the case of plastic packaging.

4.1.2 Single use plastics and packaging

Single use plastics can include any disposable plastic item which is designed to be used only once. Single use items are often used in packaging, consumer products, cosmetics and healthcare. Examples include: lightweight plastic bags, disposable utensils, beverage containers, coffee capsules, wet wipes, and razor blades. 95% of plastic packaging is single-use, and results in the loss of €70-100 billion in material value from the economy per year (WEF, EMF and McKinsey&Company, 2016). Packaging accounts for 42% of the plastics produced globally since 1950 (Geyer, Jambeck and Law, 2017). Plastic packaging has witnessed a significant increase in production volumes and is expected to double by 2030 and to more than quadruple by 2050 (WEF, EMF and McKinsey&Company, 2016). In particular, plastics is widely used to package food, accounting for 37% of the European market share (Muncke, 2016). Within all plastics application, packaging has the highest recycling rate, 39.5% in 2014 (Plastics Europe, 2016). However, most of the plastic packaging is lost within the same year of its production, contrarily to all other plastic applications – see Figure 4.1 (Geyer, Jambeck and Law, 2017). Recycled plastics tend to be diverted to lower value applications, which could impact the demand for plastic recyclates for specific applications (WEF, EMF and McKinsey&Company, 2016).
The environmental impact of single use plastics

Life cycle assessment (LCA) modelling has been carried out in this project, as well contributing to the Marine Litter Impact Assessment. Full reporting of the results, their implications and limitations can be found in the Plastics report which forms an Annex to this document, but a summary of the methodology and results are presented below.

Twelve products & their potential alternatives were originally considered for modelling (see Table 5.1). Their selection was aligned with the Impact Assessment on the proposal on marine litter including single use plastics. The criteria for selection of plastics alternatives - single-use non plastic (SUNP) and multi-use (MU) products - were that:

1. The materials of which SUNP items are composed avoid the generation of microplastics. This thus excluded biodegradable plastics from the study scope as such biodegradability can only be insured in specific conditions which are seldom met in the marine environment (Thompson, 2006; Kershaw, 2015).

2. Alternative products meet the same function as the plastic products that they substitute in terms of properties that the materials ensure. Such products were not identified for product groups Crisps packets and Sweet wrappers (transmission of O₂ and water vapour, opacity), as well as for SUNP Drinks cups and lids (permeability and resistance of insulating layer to heat) and sanitary towels (permeability and absorbency).

3. Multi-use items need to ensure that use of single-use plastics is avoided. This ruled out reusable cigarette filters, as such are used in addition of a traditional cigarette (as an additional filter) and would thus not displace the use of a cellulose acetate filter.

4. Alternatives need to satisfy broadly the same market. This ruled out items such as e-cigarettes, which are tobacco substitutes and thus not necessarily targeting an analogous market segment.
Table 4.1 Product systems considered - with materials specified - for single-use plastic items (SUPs), single-use non-plastic items (SUNPs) and multi-use items (MU)

<table>
<thead>
<tr>
<th>Product category</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cigarette butts</td>
<td>Cellulose acetate filter</td>
<td>Natural fibre filter (hemp/cotton)</td>
<td>-</td>
</tr>
<tr>
<td>Drinks bottles</td>
<td>Average volume PET bottle</td>
<td>Average non-plastic container (Aluminium/glass)</td>
<td>Average multi-use container: Consumer-led: PET/Aluminium Industry-led: PET/Glass</td>
</tr>
<tr>
<td>Cotton buds</td>
<td>PP bud</td>
<td>Paper bud</td>
<td>Reusable MDPE bud</td>
</tr>
<tr>
<td>Crisps packets</td>
<td>Excluded from scope</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sweet wrappers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sanitary towels</td>
<td>Ultrathin pad (PE, PP, PET, SAP)</td>
<td>-</td>
<td>Washable cotton pad</td>
</tr>
<tr>
<td>Wet wipes</td>
<td>Synthetic fibre wipe (w/ lotion)</td>
<td>Cotton ball + lotion</td>
<td>Cotton handkerchief + lotion</td>
</tr>
<tr>
<td>Cutlery</td>
<td>Average PP utensil</td>
<td>Average wooden utensil</td>
<td>Average steel utensil</td>
</tr>
<tr>
<td>Straws</td>
<td>PP straw</td>
<td>Paper straw</td>
<td>Average reusable straw (steel/silicone)</td>
</tr>
<tr>
<td>Stirrers</td>
<td>PP stirrer</td>
<td>Wooden stirrer</td>
<td>Steel stirrer</td>
</tr>
<tr>
<td>Food containers</td>
<td>PS clamshell container</td>
<td>Paperboard + wax container</td>
<td>PE tupperware box</td>
</tr>
<tr>
<td>Drinks cups and lids</td>
<td>Paper cup w/ PE coating and LDPE lid</td>
<td>-</td>
<td>Reusable PP cup (w/ LDPE, rubber, silicone components)</td>
</tr>
</tbody>
</table>

Note(s): Products with materials separated by forward slash are market averages of separate products made of the materials given.

On a per-functional unit basis, the results indicate that single-use plastics are generally worse-performing in terms of the climate change impact category, with the noted exception of SUNP Drinks bottles (i.e. glass & aluminium). Their SUNP reference product is an average of both aluminium cans and glass bottles, and closer inspection shows that glass strongly dominates the result, chiefly due to the impacts from raw materials acquisition and manufacturing. SUNP straws (made of paper) are also worse-performing within their product group for the same reason.

For the water use impact category, SUNP items generally have larger impacts. As SUNP products are bio-based, we can expect larger water burdens at the raw materials stage. The influence of bio-based materials is strongly evident for both sanitary items where burdens from cotton production dominate the results and lead to a near-tie in impacts between alternatives, showing that the impacts of their plastic components are negligible compared to their bio-based inputs.

Water pollution impacts are varied with no clear root cause between items but rather a varied influence based on types of input materials, some of which entail proportionally large chemical emissions to water such as due to the use of pesticides for bio-based inputs.
SUP items are the worst performers in the air quality impact category, except for SUNP items with paper or wood as inputs, where manufacturing and raw materials burdens have the most significant contribution.

Upscaling to the whole market level, it is immediately evident that number of uses does not necessarily correlate with magnitude of impact. This is most strongly true for cigarette butts - while exceeding all other products in number of uses, they have relatively modest market-level environmental impacts. However, LCA cannot address the health impacts of cigarette use, nor is it currently able to capture marine litter impacts of discarded cigarette butts. The present analysis shows that LCA environmental impacts are not a primary concern for cigarette butts and that efforts should rather focus on their health and possibly marine litter impacts.

Drinks bottles surpass all other products in impacts by approximately an order in magnitude, even though their number of uses is exceeded by other products. Inspection of LCA results shows that this is primarily due to impacts of raw material manufacturing and their fossil fuel intensity. However, these results should be interpreted with care. Results on a per-functional unit basis illustrate that SUNP items greatly surpass SUP drinks bottles in impacts due to the intensity of raw materials production. Though these results are very sensitive to assumptions on recycling and reverse logistics and should be interpreted in this light, the data herein shows that there is no clear-cut case for replacing SUP drinks bottles with non-plastic alternatives. However, the lesser per-functional unit impacts of multi-use drinks bottles suggests that promotion of reuse and refill at least for water consumption (such as through municipal refill schemes) is an avenue for lessening the impacts of drinks bottles use.

Finally, food containers illustrate that the general trend of bio-based products having higher impacts on a per-functional unit basis holds at the market level. Though SUP and SUNP alternatives have roughly equal market shares, SUNP impacts exceed those of SUP for all categories apart from Climate change (greenhouse gas emissions). Inspection of life-cycle inventories shows that this is mostly attributable to the large impacts of paper production (incl. raw material production).

4.1.3 Waste management

The durability of plastics gives this material utility and makes it environmentally damaging. It is estimated that in 2015 plastic waste generation amounted to 6300 million tonnes. Of this amount, 9% was recycled, 12% incinerated and 79% accumulated in landfills or in the natural environment. Leakage into terrestrial and marine environment is a global issue. It is estimated that approximately 8 million tonnes of plastics are released into the world’s ocean each year. Of the amount of plastics already present in the marine environments, packaging represents the largest share. Today, there are 150 million tonnes of plastics in the ocean (WEF, EMF and McKinsey&Company, 2016). The accumulation of litter in marine environments originates principally from land-based sources, accounting for more than 80%, while less than 20% comes from sea-based sources (Ocean Conservancy and McKinsey&Company, 2015).

Identifying the origin and sources of marine litter represents a complex task with inherent levels of uncertainty in the results. A large fraction of marine litter is composed of unidentifiable items (such as small pieces of a larger item), but also identifiable items which could have multiple sources (such as a whole pieces of polystyrene packaging). Nevertheless, a large portion of marine litter can be attributed to specific products, and particularly to single use plastics. Table 4.2 summarises the top ten marine litter items in Europe as laid out in the European Commission Marine litter Impact Assessment, Annex 3\(^{29}\) (SWD(2018) 254 final).

### Table 4.2 Top 10 marine litter items by count

<table>
<thead>
<tr>
<th>Product Type</th>
<th>Total number</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Drinks bottles, caps and lids</td>
<td>24,541</td>
</tr>
<tr>
<td>2. Cigarette butts</td>
<td>21,854</td>
</tr>
<tr>
<td>3. Cotton bud sticks</td>
<td>13,616</td>
</tr>
<tr>
<td>4. Crisps packets / sweets wrappers</td>
<td>10,952</td>
</tr>
<tr>
<td>5. Sanitary applications</td>
<td>9,493</td>
</tr>
<tr>
<td>6. Bags</td>
<td>6,410</td>
</tr>
<tr>
<td>7. Cutlery, straws and stirrers</td>
<td>4,769</td>
</tr>
<tr>
<td>8. Cup and cup lids</td>
<td>3,232</td>
</tr>
<tr>
<td>9. Balloons and balloon sticks</td>
<td>2,706</td>
</tr>
<tr>
<td>10. Food containers including fast food</td>
<td>2,602</td>
</tr>
</tbody>
</table>


Microplastics (small pieces of plastic litter) which are smaller than 5mm in diameter represent a specific environmental challenge. These can have either primary or secondary sources, the latter being those which have fragmented from a larger item. Important sources include fragmentation of larger items in the environment, release of abrasives from cosmetics and other products, tyre wear and tear, fibres released from the washing of textiles and the spillage of pre-production pellets or powders that are in transit or process prior to being made into everyday plastic items (GESAMP\(^{30}\), 2017).

In 2014, 29.7% of the 25.8 million tonnes of post-consumer plastic waste generated were recycled. Plastic packaging has the highest rate of recycling, 39.5% in 2014 (Plastics Europe, 2016). The recycling potential of plastics exceeds this, but most plastic items are not collected for recycling, currently only 14% (WEF, EMF and McKinsey&Company, 2016). The recycling rate of plastics is also lower than for other materials, such as paper (58%) and iron and steel (70-90%). Plastics are often recycled into products of lower value which degrade in quality with successive cycles (WEF, EMF and McKinsey&Company, 2016).

#### 4.1.4 Socioeconomic footprint of the sector

The plastics industry operates across approximately 60,000 companies in Europe (Plastics Europe, 2016), representing a major sector of Europe’s economy. In Europe, 1.5 million jobs are directly linked to the plastics industry (Plastics Europe, 2016). These include 30,000 jobs within the plastics recycling industry (Plastics recyclers Europe, 2016) and 167,000 in the production phase (Plastics Europe, 2012). The plastic industry’s turnover reached €340 billion in 2015 and today is the 7th European largest industry in terms of value added (Plastics Europe, 2016). In addition to its economic relevance in Europe, the plastics industry largely contributes societal benefits through its various applications and properties. However, plastics are also highly damaging to the environment and there are costs of inaction. According to UNEP (2014), environmental damage to marine ecosystems from plastics amounts to $13 billion per year, including financial losses incurred by fisheries and tourism as well as time spent cleaning up beaches (UNEP, 2014). Figure 4.2 illustrates global natural capital costs associated with plastics from a number of consumer good sectors.

Recycling represents an important sector with associated economic and social benefits. As waste is turned into raw materials, recycling can contribute to job creation, economic growth, enhance competitiveness, foster innovation and secure access to critical resources (European Environment Agency, 2011). In addition, the direct employment opportunities associated with recycling include low-skilled work (European Environment Agency, 2011; Plastics Recyclers Europe, 2016), therefore offering potential improvements in social inclusion and poverty alleviation. The main direct jobs associated with recycling are found within sorting, separation and collection, while indirect jobs are found within construction of recycling facilities, manufacturing of equipment, research and innovation, as well as management related jobs (Plastics Recyclers Europe, 2016).

The European bioplastics industry presents opportunities for employment generation, with potential improvements in the development of rural areas. In 2013, there were 23,000 jobs associated with this industry in Europe. Within the industrial biotechnology sector, bioplastics has the second largest share, after bio-based chemicals, for direct employment generation in Europe, comprising 23% of total employment within the sector (Debergh, Bilsen and Van de Vekve, 2016).

4.1.5 Environmental pollutant releases

The number of E-PRTR releases for the plastics sector that has all required data available for the calculation of environmental impacts varies over time; in early years, around 25% of all recorded releases have pollutant (CAS) codes, although by 2016 this has fallen only 11%.

The impacts upon human and ecosystem health from the plastics sector pollutant releases largely mirror each other, although in more recent years the impacts on ecosystem health have accelerated slightly more rapidly, suggesting a bigger role for pollutants with a larger ecosystem impact. The impacts are rather

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31 The key characteristic that is often missing is the CAS pollutant code, which is required in order to assign a human health impact (measured in disability-adjusted life years) and ecosystem health impact (measured in species years lost).
volatile, with peaks in 2007 and 2010, while 2016 also looks likely to be another short-term peak (although due to an absence of data for 2017 this cannot be confirmed).

At a Member State level, Poland overwhelmingly dominates reported emissions – between 250m and 300m tonnes of pollutant were released in Poland every year, while all other Member States released less than 1m tonnes. This suggests a methodological issue with the manner in which emissions are reported in Poland; the E-PRTR data suggests that these emissions are linked to onsite energy generation (for consumption in plastics production), but such a stark contrast with the rest of the EU seems unlikely. Beyond CO$_2$ (which dominates these emissions impacts), the largest pollutant releases by volume are dichloromethane (c500kg released in 2007), trichloroethylene (c280kg released in 2010) and Toluene (releases of around 250kg in both 2013 and 2015).

Figure 4.3 The environmental impacts of pollutant releases from the plastics sector

4.2 Current direction of travel

Trends in plastics production and consumption imply that an increasing amount of oil will be dedicated to the plastics industry in the future. However, the scarcity of oil and the volatility of its price represent an obstacle to the development of petroleum-based plastics and call for the development of alternative feedstocks (WEF, EMF and McKinsey&Company, 2016). Alternatives to conventional plastics, such as bioplastics, recycling and CO$_2$ capture, are options which are increasingly being explored.

The Ellen MacArthur Foundation predicts that the plastics sector will account for 20% of total oil consumption by 2050. The environmental impacts associated with plastics production are expected to increase as well and, if current trends continue, the plastics industry is expected to account for 15% of global annual carbon budget by 2050, from 1% today (WEF, EMF and McKinsey&Company, 2016).

While the sector continues to be closely coupled to fossil fuel production, this also links it to climate change objectives. Closing the loop on plastic production, including through prevention, reduction, reuse and recycling activities has the potential to dramatically reduce the emissions of the sector. A study by Zero Waste Scotland estimated that circular economy activities in the Scottish economy could reduce territorial emissions by 11 million tonnes CO$_2$e per annum by 2050 compared to business as usual (Pratt and Lenaghan, 2015).
The bioplastics market is predicted to grow in the near future, with predictions of increased production capacity of approximately 6.11 million tonnes by 2021.

If current waste generation and management trends for all plastic continue, 12,000 million tonnes of plastics will be accumulated in landfills or leaked into the natural environment by 2050 (Geyer, Jambeck and Law, 2017). Furthermore, if no action is taken, it is expected that there will be almost 250 million metric tonnes of plastic in the oceans by 2025 (Ocean Conservancy and McKinsey&Company, 2015). The Ellen MacArthur Foundation estimates that by 2025, plastic litter in the oceans is expected to reach 1 tonne for every 3 tonnes of fish in a business-as-usual scenario, and to exceed the amount of fish (by weight) by 2050 (WEF, EMF and McKinsey&Company, 2016).

A growing accumulation has consequences on human activities which rely on these environments. For instance, the tourism sector can be severely impacted by plastic leakage on coastal and marine environments due to the consequent loss of aesthetic value. Moreover, marine litter causes harm to human and wildlife health (Werner et al, 2016). Waste generation and mismanagement are the main land-based sources of plastic litter.

Bioplastics are increasingly being used for packaging. However, replacing conventional plastics with bioplastics does not guarantee a solution to the marine litter issue, nor to low levels of reuse and recycling. Biodegradable plastics require specific temperatures and time to properly degrade, these are conditions that cannot be guaranteed in the environment (Surfrider Foundation et al, 2017). In addition, bio-based and biodegradable plastics need appropriate infrastructure for recycling which does not always match the process used for conventional plastics. Nevertheless, bio-plastics are often included in the existing recycling process, generating concerns for the plastic converters industry as the quality of recyclates is not ensured. When landfilled or incinerated, bioplastics are associated with greenhouse gas emissions. Bioplastics might also act as a perverse incentive and lead to increase littering trends as consumers tend to consider them as easily biodegradable products (EPA Network, 2017; Surfrider Foundation et al, 2017).

Socio-economic benefits associated with the plastics sector are expected to be influenced by future trends in recycling, eco-design and bioplastics in the EU. Labour demand is expected to increase as a response to changes in chemical and manufacturing processes of plastics. For instance, redesigning plastics requires new skills and technologies (European Commission, 2011).

Growth in sub-sectors including waste management and plastic conversion have the potential to create new SMEs and employment opportunities, contributing to socio-economic objectives in the SDGs. Recycling is expected to contribute to employment by generating 50,000 new direct jobs by 2020. These are then expected to impact the wider economy and generate an additional 75,000 indirect jobs in Europe. The jobs derived from recycling have the potential to reach 80,000 direct jobs and 120,000 indirect jobs by 2025. A Club of Rome study showed that employment activities linked to the circular economy, including activities favouring reuse and recycling, could create 75,000 new jobs in Finland, 100,000 in Sweden, 400,000 in Spain and 500,000 in France (Wijkman, Skånberg and Berglund, 2016).

Increasing recycling of plastic packaging can bring significant benefits in terms of reduced environmental costs. Trucost investigated how environmental costs would change if plastic packaging recycling reached a target of 55% and landfilling was reduced to 10% in both Europe and North America. The results of the study show that the environmental costs of plastic use would drop by US $4.8 billion per year. This includes an environmental benefit of material and energy recovery of US $3.9 billion (Trucost, 2016). Jobs associated with the bioplastics industry are expected to reach 200,000 by 2030 (European Bioplastics, 2017).

Indirectly, the plastics sector could also contribute to targets relevant to other sectors in the economy. For instance, innovation in polymers could facilitate lightweighting in transport (9.1). Plastic packaging is often cited as contributing positively towards food waste (12.3) but trends towards over-packaging threaten to undermine these benefits, with packaging often linked to wastefulness and over-purchasing (include ref). Developing comprehensively sustainable solutions for packaging food will be a key challenge for the sector in the future.
4.2.1 The impact of changing consumer preferences

Plastics properties can also influence consumption levels by attracting or discouraging consumers. In the first case, plastic properties such as lightweight or food preservation increasingly encourage the application of the material due to the associated benefits. Plastics can be used to lightweight motor vehicle chassis in order to reduce fuel use and emissions, likewise plastics are often lighter than alternative packaging materials such as glass or metals which can reduce emissions in logistics (Thompson et al, 2009). Despite such benefits, plastics are associated with a throw-away society and high levels of waste, especially when it comes to single-use plastics and packaging. Growth in plastics applications in packaging has come as a result of a global shift to single-use containers (Geyer, Jambeck and Law, 2017). This is particularly the case of food packaging where plastics are widely used to protect, market, and deliver food products. The characteristics of food packaging, such as size and shape, have an impact on consumers’ purchasing choices (Aschemann-Witzel et al, 2015). However, food products are often considered over-packaged by consumers (INCPEN, 2008) and such feature has been reported as a reason to avoid purchasing (INCPEN, 2008; Which?, 2011). By 2020, Europe is estimated to consume more than 900 billion items of packaged food and drink annually (Smithers Pira, 2015). Plastic packaging has been found to be associated with the migration of harmful chemicals such as endocrine disruptors (Karamfilova and Sacher, 2016). These chemicals are largely used in food contact materials and plastic packaging in particular. Evidence of health impacts associated to chemicals migration into food and beverages can have an impact on consumers’ purchasing choices. Issues related to plastics, such as marine and terrestrial litter, as well as toxic additives are contributing to making consumers more and more aware of the environmental consequences of plastic products (Worldwatch Institute, 2015). As a consequence, consumers are becoming increasingly aware of their responsibility of making more environmentally-friendly choices at the purchasing stage (Karlaite, 2016). In the case of packaging, purchasing choices have generally been based on factors such as aesthetics, convenience of use, and design. However, a preference for ethical or green products has lately been observed among consumers, even though the behaviour-attitude gap persists (Rokka and Uusitalo, 2008). As a response to such growing consumer environmental preferences, coupled with the demand for increased transparency, alternatives to conventional plastics are being explored (Green Dot Bioplastics, 2017), with inevitable consequences on demand.

4.3 Future policy priorities

4.3.1 Links between the sector and the Sustainable Development Goals

Activities throughout the plastic sector’s value chain can contribute both positively and negatively towards the Sustainable Development Goals and targets at the global and European level. The future development of the sector will determine how it contributes towards the goals. A number of the SDGs targets are closely linked to the plastics sector, notably those which relate to waste management (12.4, 12.5), and marine pollution (6.3, 14.1). This reflects the reality that poorly managed plastic waste has become a sustainability challenge with global relevance.

4.3.2 Policy options

The following potential policy options have been explored:

A plastics tax

Environmental taxes are one type of market-based instrument, which may be used to implement environmental policy priorities. These measures can “raise fiscal revenues while furthering environmental goals” (OECD, 2005, World Bank 2005). Environmental taxes are already present in a number of sectors including energy, transport, carbon and natural resources. On a political level growing attention has been given to environmental taxes, recognising their potential advantages in comparison to traditional ‘command and control’ regulations. Furthermore, specific environmental challenges, such as climate
change, can be important drivers of such reforms (Withana\textsuperscript{32}, 2015). If well designed environmental taxes can bring a range of benefits which can support policy objectives in a range of areas.

Table 4.3 Potential benefits of a plastics tax

<table>
<thead>
<tr>
<th>Benefit</th>
<th>Examples/explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Financial</td>
<td>Budget contributions and macroeconomic stability</td>
</tr>
<tr>
<td></td>
<td>Mobilisation of revenues for policy priorities</td>
</tr>
<tr>
<td></td>
<td>Reduced requirements for budget expenditure (due to savings, e.g. via ESS)</td>
</tr>
<tr>
<td>Economic</td>
<td>Revision of price signals (internalised externalities)</td>
</tr>
<tr>
<td></td>
<td>Catalyse innovation</td>
</tr>
<tr>
<td></td>
<td>Reduce growth distorting taxes</td>
</tr>
<tr>
<td>Social</td>
<td>Health (reduce pollution)</td>
</tr>
<tr>
<td></td>
<td>Employment creation (double dividend hypothesis)</td>
</tr>
<tr>
<td></td>
<td>Availability of natural resource/public goods for future generations</td>
</tr>
<tr>
<td>Environmental</td>
<td>Incentivise efficient resource use</td>
</tr>
<tr>
<td></td>
<td>Reduce harmful emissions/pollutants</td>
</tr>
<tr>
<td>Poverty reduction</td>
<td>Mobilise funds for pro-poor investments</td>
</tr>
<tr>
<td></td>
<td>Address environmental impacts unfairly impacting the poor</td>
</tr>
<tr>
<td></td>
<td>Improve access to ESS</td>
</tr>
</tbody>
</table>

A European wide plastic tax has been suggested as an instrument which could help to increase the update of recycled material in the plastics sector, as well as addressing other policy priorities (including budgetary issues). In January 2018, Budget Commissioner Günther Oettinger suggested a plastics tax (EC\textsuperscript{33}, 2018). In the Strategy for Plastics, it is also noted that “national and regional authorities should make better use of taxation and other economic instruments to reward the uptake of recycled plastics and favour reuse and recycling over landfilling and incineration.”

Socioeconomic modelling of the potential impacts of a plastics tax show that aggregate level economic consequences are likely to be minimal. Economic activity in production of virgin plastics would likely contract, but demand would be displaced to waste management and recovery of materials: key sectors to the circular economy. This analysis suggests that there is no significant net cost of taxation designed to reduce use of plastics.

Three scenarios have been modelled:

1) a tax at the point of generation of non-recyclable plastic waste, imposing additional cost to each sector, dependent upon its magnitude of waste generated. Rate of taxation is 0.5€ per kg of plastic waste. The key responses in E3ME are: increased prices resulting from increased costs of production and a reduction of waste generation (crucially through substitution of virgin plastic to alternative materials).

2) a tax at the point of production: a tax is applied proportional to the weight of plastic material produced. The tax imposes an additional cost to the plastics sector, the rate of taxation is 0.102€ per kg of plastic. The key responses in E3ME are: increased price in the plastics sector, increased prices across the economy from increased costs of production in industries using plastic as an input to production, reduction in final consumer demand for plastic products and reduction of intermediate demand for plastic. An important issue to consider in this scenario was the potential of production leakage.

3) a border tax on select plastic external imports, applied to sheets of plastic only, at a rate of 0.102€ per kg. The key responses in E3ME are: increased prices of imports (both final consumer and intermediate goods),

\textsuperscript{32} http://www.greengrowthknowledge.org/sites/default/files/Withana_Overcoming_obstacles_to_green_fiscal_reform.pdf

increased prices across the economy from increased costs of production in sectors that use plastic as an input to production, a reduction in final consumer demand for plastic products, a reduction in intermediate demands for plastic.

All scenarios result in very similar effects: contraction in the plastics sector, significant tax revenue, and a small net positive impact on GDP and employment. Magnitude of impact is driven by size of the tax: the per kg tax on waste is almost five times higher than that for production and imports. The impacts of the border tax scenario are significantly smaller than the waste and production tax. This is because the tax is only applied to imports, not any EU domestic production, and only to plastic sheets.

The positive net effects on GDP and employment to 2030 in these scenarios is a result of tax revenue and changing composition of economic activity. All the revenue from the plastics tax is spent by government, providing a demand stimulus. The results illustrate the trade-off between reducing plastics production and generating revenue through environmental taxation. Change in intermediate demand from the plastics sector to the waste management and materials recovery sector reduces imports of plastics and directs demand to economic activity within the EU. This increase in domestic activity is partly an artefact of assumption: whilst waste collection is necessarily a domestic activity, materials recovery and recycling is not, the EU exports a significant volume of waste for processing in other regions.34

Demand response to the change in price of plastic use is the most important variable in the modelling. Three stylized substitution elasticities for plastic were used to adjust intermediate demand. Under assumption of fixed production behaviour, the tax has very limited impact on output of the plastics sector. Estimation of behavioural response to changes in price of plastic use is beyond the scope of this project, but given the importance of this variable it is an important subject for further study.

An important impact in the production tax scenario is production leakage. External exports from the EU28 contract by €667 million by 2030 in the first production tax scenario. Production outside of the EU increases, with the largest absolute increase in production being in China. In the first border tax scenario, EU domestic output increases, replacing otherwise imported products.

Table 4.4 E3ME results for different plastic tax scenarios, EU28

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Price demand response (%)</th>
<th>EU28 GDP</th>
<th>EU28 emp. ('000s)</th>
<th>NACE 22 Rubber &amp; Plastics output (%)</th>
<th>NACE 22 Rubber &amp; Plastics emp. ('000s)</th>
<th>Tax revenue (bn 2018€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waste tax</td>
<td>0</td>
<td>0.048</td>
<td>59.5</td>
<td>-0.02</td>
<td>-0.91</td>
<td>5.40</td>
</tr>
<tr>
<td></td>
<td>50</td>
<td>0.051</td>
<td>52.4</td>
<td>-0.82</td>
<td>-7.60</td>
<td>5.26</td>
</tr>
<tr>
<td></td>
<td>150</td>
<td>0.056</td>
<td>37.8</td>
<td>-2.41</td>
<td>-21.08</td>
<td>5.00</td>
</tr>
<tr>
<td>Production tax</td>
<td>0</td>
<td>0.046</td>
<td>60.4</td>
<td>-0.29</td>
<td>-6.61</td>
<td>5.40</td>
</tr>
<tr>
<td></td>
<td>50</td>
<td>0.05</td>
<td>55.5</td>
<td>-0.97</td>
<td>-12.61</td>
<td>5.36</td>
</tr>
<tr>
<td></td>
<td>150</td>
<td>0.057</td>
<td>46.7</td>
<td>-2.34</td>
<td>-24.63</td>
<td>5.29</td>
</tr>
<tr>
<td>Border tax</td>
<td>0</td>
<td>0.004</td>
<td>5.0</td>
<td>0.03</td>
<td>0.15</td>
<td>0.39</td>
</tr>
<tr>
<td></td>
<td>50</td>
<td>0.004</td>
<td>4.5</td>
<td>-0.01</td>
<td>-0.30</td>
<td>0.39</td>
</tr>
<tr>
<td></td>
<td>150</td>
<td>0.005</td>
<td>3.9</td>
<td>-0.11</td>
<td>-1.21</td>
<td>0.39</td>
</tr>
</tbody>
</table>

Notes(s): NACE 22 output is presented as % differences across scenarios for values in 2018 constant prices.

34 Collection of waste and materials recovery are both categorised under NACE 38 ‘Waste collection, treatment and disposal activities; materials recovery’. Wholesale of waste and scrap is found under NACE 47.6 ‘Other specialised wholesale’.
Environmental impacts of a plastics tax

In addition to the economic modelling of plastics tax scenarios, environmental impacts have further been estimated via coupling E3ME and the EXIOBASE environmentally extended input-output database. As for economic results, all scenarios display very similar environmental impacts, with small variation with price demand response.

Acidification, eutrophication and photochemical oxidation potentials see falls in impacts from combustion emissions in industry but in the cases of acidification and eutrophication, these are respectively balanced and outweighed by emissions of NOx and NH3 from fertiliser/manure application, stemming from the growth of the agricultural sector (which is linked to rising food use rather than the effects of the modelled policy instruments). In the case of summer smog potential, decreases in impact drivers from combustion and respective growth in resource use impacts driven chiefly by falling CO2 emissions.

Human toxicity potential shows the largest increase out of non-resource use environmental impacts on account of increasing industrial emissions of heavy metals and polycyclic aromatic hydrocarbons (PAH). Global warming potentials show the largest decreases in non-resource use impacts driven chiefly by falling CO2 emissions.

In the case of all environmental impact drivers apart from CO2 and resource use, technological assumptions are held fixed at EXIOBASE 2011 levels given lack of concrete data on future technological development. Hence, the observed increases in industrial emissions are likely overestimated, as well as to a lesser extent the increases in emissions from the agricultural sector, where even given fixed technologies, a more direct link between demand, production and manure/fertiliser emissions can be expected.

Overall, the results for environmental impacts show a weak overall response to the policy instruments modelled but rather reflect overall economic trends such as rising food use and respective growth in agricultural production. Where increased industrial emissions are seen, this is partly confounded by the assumption of unchanging production technologies toward the future for non-CO2 emissions.

<table>
<thead>
<tr>
<th>Environmental impact potential</th>
<th>BAU</th>
<th>Waste tax</th>
<th>Production tax</th>
<th>Border tax</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>0%</td>
<td>50%</td>
<td>150%</td>
</tr>
<tr>
<td>Acidification potential SO2 eq</td>
<td>th tonnes</td>
<td>-0.78</td>
<td>-0.78</td>
<td>-0.79</td>
</tr>
<tr>
<td>Eutrophication potential PO4 eq</td>
<td>th tonnes</td>
<td>3.23</td>
<td>3.24</td>
<td>3.25</td>
</tr>
<tr>
<td>Photochemical oxidation potential ethylene eq</td>
<td>th tonnes</td>
<td>-0.06</td>
<td>-0.03</td>
<td>-0.03</td>
</tr>
<tr>
<td>Human toxicity potential 1,4-DCB eq</td>
<td>th tonnes</td>
<td>4.98</td>
<td>5.01</td>
<td>5.03</td>
</tr>
<tr>
<td>Land use</td>
<td>km2</td>
<td>6.05</td>
<td>6.05</td>
<td>6.04</td>
</tr>
<tr>
<td>Food use</td>
<td>th tonnes</td>
<td>59.46</td>
<td>59.48</td>
<td>59.47</td>
</tr>
<tr>
<td>Feed use</td>
<td>th tonnes</td>
<td>-0.10</td>
<td>-0.12</td>
<td>-0.12</td>
</tr>
<tr>
<td>Forestry use</td>
<td>th tonnes</td>
<td>58.94</td>
<td>58.97</td>
<td>58.96</td>
</tr>
<tr>
<td>Construction minerals</td>
<td>th tonnes</td>
<td>26.03</td>
<td>26.05</td>
<td>26.06</td>
</tr>
<tr>
<td>Ferrous ores</td>
<td>th tonnes</td>
<td>15.61</td>
<td>15.65</td>
<td>15.64</td>
</tr>
<tr>
<td>Non-ferrous ores</td>
<td>th tonnes</td>
<td>20.68</td>
<td>20.70</td>
<td>20.70</td>
</tr>
</tbody>
</table>
Increased recycled content in products

Increasing the use of recycled material can provide an alternative feedstock for some plastic products, replacing virgin material inputs. The advantages of this are efficiencies in material and energy use, as well as potential for improvements in waste management and markets for secondary materials. Possible disadvantages exist in a real, or perceived, reduction in the material quality of recyclates compared to virgin material. An increase in the rate of recycling may not translate into an increase in the recycled content of the collected product, for example where recyclates are diverted to different lower value applications (e.g. packaging to synthetic fibres) and/or where there is a net growth in the material output of the sector (i.e. Jevons paradox).

Changing sector priorities away from virgin inputs towards the management and conversion of plastic waste has the potential to generate employment as these activities are foreseen to be more labour intensive. Increasing recycled content in the manufacturing of plastics products can reduce emissions as less energy is needed per unit of output (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016). Emissions can be reduced by 1.1-1.3 tonnes of CO₂e by recycling one additional tonne of plastics rather than producing it from virgin fossil feedstock (WEF, EMF and McKinsey&Company, 2016). According to a life cycle analysis, producing plastic bottles by using 100% recycled PET rather than virgin PET could reduce CO₂ emissions by 27% (WRAP 2008; Hopewell et al, 2009).

A range of policy measures could be useful to increase the uptake of recycled plastic in plastic products. Increasing the uptake, quality and economics of recycling requires measures to be adopted on upstream design (e.g. eco-design) and downstream collection, sorting and reprocessing.

Extended Producer Responsibility (EPR) schemes present opportunities to incentivise the uptake of recycled plastics in plastic products through eco-modulation of fees. Fee modulation implies a diversified cost applied to producers according to the criteria of the products placed on the market. In addition to criteria based on weight and materials, fees can be modulated based on aspects of eco-design related to the level of recyclability and separability of products and their components as well as on the amount of recycled plastics.

Deposit-refund schemes on one-way plastic containers have been proven to significantly increase the rate of recycling (e.g. PULPA) as the containers (e.g. PET bottles) are returned to the producers and sent to recycling companies where new products are generated and transported back to the consumers.

Increase the re-use of plastics

Reusing plastics can offset the input of virgin material in the value chain. With reference to the waste hierarchy, re-use is viewed as preferable to recycling. In practice, the re-use of materials might have context-specific environmental strengths and disadvantages, for example in relation to the product weight-based emissions, the number of iterations of re-use, or the distance and mode of reverse logistics (WRAP35, 2010). Re-usable plastics are relevant in B2B and B2C scenarios, for instance in reusable pallets for distribution or re-usable carrier bags respectively. Increasing the re-use of products can also help to reduce the use of single-use plastics.

Specific policies might facilitate re-use of plastics, e.g. deposit refund schemes, or taxes/charges on single use products. Re-usable plastics might also necessitate a change in business models, for example introducing service over ownership models for dispensers and re-usable cutlery in the catering sector, in order that the higher upfront costs of durable materials can be made affordable to businesses.

Measures addressing single-use plastics are increasingly being implemented. Instruments include bans and taxes on specific items, such as single-use plastic bags. Instrument of this kind promote product substitution to single-use non plastic as well as reusable plastic and non-plastic alternatives.

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35 WRAP, 2010, Single Trip or Reusable Packaging - Considering the Right Choice for the Environment
EPR schemes can encourage improved design and therefore promote a shift from single-use plastics items to products designed with an after-use pathway in mind, facilitating reuse (ten Brink et al., 2017).

Additional potential measures exist addressing consumption patterns of single-use plastic items. Green public procurement rules in municipalities addressing single-use items can lead to substantial behavioural changes (e.g. City of Hamburg). Similarly, the recent development of zero packaging stores represent a relevant example which can lead to a considerable reduction of single-use containers in food retailers. The growing number of retailers adopting this measure demonstrates a growing consumers’ demand for less packaging.

4.4 Conclusions

The plastics sector is a large and economically important sector that is closely coupled to fossil fuel production which provides the sector’s primary feedstock. Rapid growth in the plastics sector has taken place over the course of the last 50 years, notably in the packaging sector and for single use plastics which account for a significant proportion of plastic applications. Mismanaged plastic waste and leakage to the terrestrial and marine environment have become a global sustainability crisis.

Three scenarios of a plastics tax were developed to assess first-order effects of a plastics tax imposition at various points in the life-cycle of plastic products. The aggregate economic consequences of a plastics tax are likely to be minimal. Economic activity in production of virgin plastics would likely contract, but demand would be displaced to waste management and recovery of materials: key sectors to the circular economy. This analysis suggests that there is no significant net cost of taxation designed to reduce use of plastics.

5 Motor vehicles

Motor vehicle manufacture is an economically important and politically significant sector. Links between the sector and the environment are many and significant: production is important in the scale of raw material and energy use; product design is important from the perspective of the circular economy with regards to end-of-life vehicle (ELV) waste; and product use is crucial because of localised noise & air pollution and emissions of greenhouse gases (GHGs).

5.1 The current environmental impact of the sector

5.1.1 Overview

The motor vehicles sector is one of Europe’s most important manufacturing sectors but also a major contributor to a range of environmental pressures. The life-cycle of motor vehicles involves significant use of resources and energy, as well as the creation of waste and pollutants at later stages in the supply chain. Environmental impacts from the sector in the use phase of vehicles are considerable. Environmental pressures linked to motor vehicles include:

- Production emissions (e.g. car plants and steel production)
- Production resource use (extraction of critical raw materials – e.g. palladium for catalytic converters)
- Use phase emission (e.g. CO$_2$, NO$_x$ and fluorinated greenhouse gases – from mobile air conditioning, tire wear and tear leading to microplastics emissions)
- Use phase resource use (e.g. fossil fuel use)
- Use phase land use conversion (e.g. urban, highway infrastructural demands & land conversion with biodiversity impacts)
- Use phase noise pollution (e.g. urban noise pollution with health impacts)
- Use phase environmental health impacts (e.g. non-communicable disease, physical inactivity)
- End of life pollution (e.g. ELV management or tire dumping)

The manufacture of internal combustion motor vehicle sector relies on the environment to source raw materials needed for manufacturing, as well as fossil fuels needed for energy. Raw materials needed for the manufacturing of cars include steel, iron and aluminium, and increasingly polymers (Tagliaferri et al. 2016, Schulz 2016). Terrestrial acidification is caused by the production of platinum-group metals during the production phase and the emission of sulphur dioxide during the use phase of the car (Hawkins et al. 2013). In general, extractive industries for raw materials and the production processes themselves are energy intensive.

Notable in the sector are the impacts of motor vehicles in their use phase; these are linked to fossil fuel use, GHG emissions and air pollution. One-fifth of EU’s total CO$_2$ emissions come from road transport. Particulate matter (PM) is emitted during the production phase by metal supply chains and during the use phase by fuel combustion, brakes and tyres wear and road dust (Hawkins, et al. 2013, EEA 2017a, Rogge et al. 1993). Air pollutants including particulate matter (PM), sulphur oxides, nitrogen oxides, methane and ozone constitute the largest environmental health risk in Europe and lead to premature deaths and an increased incidence of various diseases (e.g. cancer and cardiovascular diseases) (Lim et al 2012, WHO 2015). Emissions of PM2.5s alone were responsible for around 399,000 premature deaths in 2014 (EEA, 2017c). During the life cycle of motor vehicles, toxic chemicals are released that can have harmful impacts on humans (AD Little 2016). Terrestrial eco-toxicity is mainly caused during the use stage by zinc emissions from tire wear and copper and titanium emissions from brake wear. Eutrophication of freshwater is caused by the disposal of sulfidic mine tailings (Hawkins et al 2013).

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38 COM(2017) 283 final
Through the introduction of EURO emissions standards over the past decade, there have been substantial reductions in air pollutants from the tailpipe of motor vehicles. According to the EEA, nitrogen oxides (NOx) from tailpipe emissions has fallen by 60% over 1990-2015. Similarly, particulates (PM10s) from tailpipe have seen a 60% reduction between 2000 to 2015. However, as mentioned above particulates from brake and tyre wear have remained largely constant over the past decade.

5.1.2 Electrification

The electrification of motor vehicles presents an opportunity to reduce the environmental impacts of motorised transport. The global stock of electric vehicles (EVs) has been on the rise since 2010. European frontrunners are Norway (albeit not an EU Member State) and the Netherlands (IEA, 2016). Key barriers for greater adoption of electric cars include battery costs and ranges (Haddadian et al. 2015), both areas where recent developments show promising signs (IEA, 2016). Additionally, adequate charging infrastructure, the lack of awareness, and a lack of consumer confidence have also been raised as barriers (IEA, 2016; European Commission, 2016c). Furthermore, there are various societal obstacles that hamper greater adoption, including for example perceptions about the safety of EVs, societal scepticism and lack of awareness about financial incentives (Haddadian et al. 2015).

Policy instruments that promote the adoption of electric cars can be divided into “technology push” policies and “market pull” instruments (IEA, 2016). Policies that push the technology support direct research, development and demonstration. Mechanisms from the second group include EV purchase incentives (e.g. sales tax exemptions, VAT exemptions, rebates at registration or sale), incentives for the use of EVs (circulation tax exemptions, waivers on fees, electricity supply reductions) and waivers on access restrictions such as access to bus lanes and tailpipe emission standards (IEA, 2016). A comparison of EVs markets in 30 different countries worldwide shows that financial incentives have a substantial impact on the adoption of EVs (Sierzchula et al. 2012). Additionally, the number of charging stations is a significant factor explaining the variation in EV adoption between countries (Sierzchula et al. 2012). See also the discussion on EV incentives below.

There are currently around 200,000 charging stations in the EU, and it is envisioned that this needs to increase to 800,000 by 2020 (Teffer, 2017). The European Commission has allocated €800 million that will finance an accelerated roll-out of charging infrastructure for electric cars (Teffer, 2017) and has also established stakeholder platforms. The objective put forward by the Commission in the European Strategy for Low-Emission Mobility (EC, 2016c) is to make electrical charging as convenient as filling a tank such that an electrical car journey across Europe becomes possible.

Extensive research has been done on the environmental impact of electric vehicles (EVs) compared to that of internal combustion vehicles (ICVs). It is shown that EVs can deliver significant GHG emission savings compared to ICVs as long as the GHG intensity of grid electricity used to power the electrical vehicle is sufficiently low, the GHG emissions related to battery manufacture and disposal are brought down, and smart charging is adopted (Ma et al, 2012; Contestabile et al, 2012). The largest environmental impact of EVs is associated with the manufacturing stage; in specific the metals used in the battery pack can have toxicological impacts (Hawkins et al. 2012; Taglieferri et al 2016, AD Little, 2016).

5.1.3 Connected cars and autonomous vehicles

As with many sectors of the economy, the increasing digitalisation of one or more aspects of the motor vehicle sector has the potential to significantly disrupt business activities and, in some cases, reduce a number of environmental impacts. Technologies are under development to increase the connectivity of vehicles, including internally (between the different operating parts of a vehicle) and externally (between the vehicle and other road users, as well as with the road environment itself) (McKinsey 2014). There may also be connectivity between vehicles and their manufacturers or other stakeholders who can utilise data from one or more vehicles in their activities. Vehicle connectivity will likely be linked to other innovations in the sector including electrification and alternative business models.

An opportunity which exists within the increased connectivity of vehicles is a varying automation level of driving functions. Within the spectrum of automation, a final level would allow vehicles to drive without a human driver. Automated driving is forecasted to bring a number of changes to the sector:

- A significant reduction in the rate of motor vehicle crashes – near zero crash risk could be achieved via deep neural network learning
- Increase the accessibility of mobility to those who are not typically able to drive (e.g. blind, disabled, young)
- An optimisation of congestion and traffic via vehicle connectivity
- A reduction of the socio-economic costs of traffic – as passengers can engage in alternative activities while driving
- Optimisation of driving speeds and breaking in order to reduce energy use
- Loss of employment in sectors focused on driving

New regulations and standards would need to be developed in order to ensure public / environmental health and safety. The connectivity and automation of vehicles will also bring new technological and data orientated challenges such as data privacy and cyber security linked to personal and fleet vehicles (Accenture, 2017). Further, liability uncertainties exist with respect to who would take responsibility for risks associated with automated drive.

It is not clear how automated vehicles will affect environmental impacts from motor vehicles in the future. Analysis of the travel, energy and carbon use impacts suggests that vehicle automation could reduce energy consumption and carbon emissions, however there is also a risk of a rebound effect, whereby efficiencies result in more travel overall, and thereby offsetting any net benefits (Wadud et al, 2016). In addition, the computing power required to enable autonomous driving could potentially substantially increase energy consumption by individual vehicles.

One scenario exercise carried out by Boston Consulting Group showed that vehicle automation in the city of Boston could bring about a number of benefits, including a reduction in the number of vehicles on the road, a reduction in total distance driven by the fleet, reduced travel time, reduced emission and a reduction in the number of parking spaces needed (BCG, 2017). In 2017, Tesla presented a prototype electrified and driverless HGV. Amongst reported benefits of the driverless model of the truck were that multiple vehicles could drive in convoy reducing drag.

The European Commission has recognised how quickly the technology is being developed and is already starting to act on connected and automated driving. In April 2017, the first conference on automated driving was held in Brussels, supported by two Horizon2020 projects. The transport strategy launched in May 2017 forecasts that driverless cars will be integrated with traffic by 2025.

5.1.4 Critical raw materials and the manufacture of motor vehicles

The manufacture of motor vehicles is reliant on the input of a range of resources to produce the necessary (and increasingly complex) components. Different vehicle designs, including conventional, hybrid and electric vehicles require different inputs. Critical raw materials are present in conventional, hybrid and electric vehicles. Key applications include (EC, 2018):

- Graphite (brake linings, exhaust systems, motors, clutch materials, batteries)
- Cobalt and lithium (EV batteries)
- Platinum group metals (palladium, platinum, and rhodium in auto-catalysts)
- Niobium (alloying agent in steel and nickel alloys)
- Rare earth elements (magnets, auto catalysts, and filters)

For conventional internal combustion vehicles, the recycling of parts and valuable materials contained in cars when they reach their end of life is not always optimised. One example is the loss of palladium in catalytic converters when end of life vehicles are exported outside of the EU. Research demonstrated that the recovery of platinum metal groups in the EU was less than 70%, even though 100% was technically feasible. Furthermore, recycling of auto-catalysts is around 50-60%, with an estimated €115 million worth
of catalytic converters leaving the EU each year in end of life vehicles (EC, 2016b). Circular economy measures, including better recovery and recycling of auto-catalysts, present an economic opportunity for the EU automotive industry.

Growth in the market for HEVs and EVs will change the demand for specific materials necessary for their manufacture. EVs are likely to increase the demand for CRMs – particularly those used in the manufacture of batteries. In 2015, the EU market for EVs required 510 tonnes of cobalt and 8330 tonnes of graphite. Currently, recycling of EVs is not expected to be widespread until 2025 – presenting a further risk of the loss of materials and their economic value.

Analysis of the use of critical raw materials and rare earth minerals in EVs suggests that although their availability is unlikely to hinder the development and uptake of vehicles, there is a need for the EU to reduce its dependence on the import of these minerals in the future (T&E, 2017a).

Policies can be used to support the effective recycling of materials in vehicles. Existing relevant measures include:

- Directive 2000/53/EU on end of life vehicles – requiring 95% reuse and recovery and 85% for reuse and recycling by average weight of a vehicle since 2015
- Directive 2005/65/EC on type approval of vehicles regarding their recyclability, which aims to ensure manufacturers allow parts to be reused, recycled or recovered
- Directive 2006/66/EC on barriers – covers automotive batteries
- Circular Economy Action Plan (COM/2015/614) – aims to reduce the leakage of raw materials from high value waste streams including end of life vehicles.

In order to ensure better recovery of materials from the motor vehicle sector, further policies may be necessary. Possible tools include a better information exchange on dismantling vehicles, clearer distinction between second hand and end of life vehicles, greater investment in recycling EV batteries, research into reducing the CRM content of motor vehicles and developing capacities for battery reuse (EC, 2018). A number of EU initiatives are already exploring the potential for battery reuse. The H2020 project SmartEnCity explores how batteries can be reused in electric taxis in Estonia (SmartEnCity, 2018).

### 5.1.5 Socioeconomic footprint of the sector

Motor vehicles are important thanks to their contribution to society and the economy in providing mobility, as well as contributing to economic growth and employment more widely. The sector can broadly be broken down into the production and sale of motor vehicles and associated supply chains, and as providing transportation services for both personal mobility as well as goods and services. In providing mobility, MVs enable companies to access their resources and transport their goods, as well as provide citizens with an access to goods and services, and participate in activities. It is an important source of revenue for various sectors of the economy and is a vital tool in the lives of everyday citizens. In the same time, the sector is a key driver for rising carbon emissions and air pollution, amongst other impacts.

**Accounting for 6.8% of Europe’s GDP, the automotive sector provides over 12.5 million jobs; 3.3m in manufacturing, 4.3m in sales and maintenance, and 4.8m in transport.** The sector’s current needs are highly dependent on the mining sector (for extraction of minerals), the oil and agricultural sectors (providing fuel for vehicles), and infrastructure development (roads and energy grids). This is exemplified through the feedback mechanism between the transport and industrial sectors in which industrial growth requires the construction of efficient modes of transportation which in turn feeds industrial growth.

While some congestion can serve as a signal of high economic activity, too much of it limits the capacity for markets to grow. Economists have measured these economic losses through hours lost in traffic jams (see box below which highlights an example from Belgium). Congestion costs Europe about 1% of its GDP every year and also causes heavy amounts of carbon and other emissions (EC, 2011a). Delays caused by congestion also result in increased stress levels, negatively impacting the health of the population. While degraded infrastructure may also result in economic losses, the cost of poor infrastructure and risks associated to motor vehicles is reflected in an average of 51 road deaths per million inhabitants in Europe in 2016, with significant variations across the continent.
5.1.6  Environmental pollutant releases

On average, around 10% of the releases captured in the E-PRTR for the motor vehicles sector have sufficient data for the calculation of environmental impacts.\(^{40}\)

The impacts upon human and ecosystem health from pollutant releases follow precisely the same trend (so overlap on the figure below), demonstrating that the composition of releases does not shift in a way which influences the overall balance between human health and environmental damages. There was a sharp increase in pollutant releases in 2014, with environmental impacts growing five times larger compared to 2013. This trend was led by a substantial increase in pollutant releases in Germany (to over 3bn tonnes), and a smaller increase in Hungary (to just over 0.6bn tonnes). These releases were of CO\(_2\), and linked to energy consumption by motor vehicle manufacturing sites, although carbon monoxide releases also increased substantially (to over 1.1m tonnes), due to release from the production and processing of metals in Poland.

![Figure 5.1 The environmental impacts of pollutant releases from the motor vehicles sector](image)

5.2  Current direction of travel

Changes to the motor vehicle industry since the millennium have been driven by a number of factors at the European and global level.

Globally, socio-economic and demographic changes have led to increasing demand for vehicles. Although these changes are most evident outside of Europe, notably in emerging markets such as China and India, growth in demand has been highly relevant to EU manufacturers. Between 2015 and 2025, the number of cars on the road is anticipated to grow from 1.1 billion to 1.5 billion (WEF, 2016). In the EU, the

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\(^{40}\) The key characteristic that is often missing is the CAS pollutant code, which is required in order to assign a human health impact (measured in disability-adjusted life years) and ecosystem health impact (measured in species years lost).
motorisation rate is already very high at 573 vehicles per 1,000 inhabitants in 2015, compared to 83 per 1,000 in China – reflecting big differences in the maturity of markets (Vieweg et al, 2017).

Emerging markets also increasingly contribute to car production. In 2016, for the first time, China overtook the EU as the world’s biggest producer of cars. In 2001, the EU accounted for 35% of world motor vehicle production; by 2016 this had decreased to 23%, though in absolute terms the EU’s production has grown over the same period (ACEA, 2018a).

The state of financial markets in Europe is a key determinant of the output of the motor vehicle industry. The financial crisis in 2008 had a strong impact on car registrations in the EU. Following the financial crisis car manufacturers in Europe were faced with surplus production, as the already mature domestic market was faced with austerity and tightening credit availability. In general, motor vehicle registrations tend to correlate with GDP growth in the EU (ACEA). Europe’s motor vehicle industry has reported declining profits so far this century. In 2007, the European automotive industry had record profits of €15 billion. By 2012, this became a financial loss of €1 billion. Overcapacity and competition in a saturated market keep profits down. Data suggests that in 2016 the industry returned to profitability (Strategy&, 2017).

Investment in research and development have brought some changes to motor vehicles in the last two decades, and represent significant sources of expenditure. In 2015, it is estimated that the ACEA members invested €41.5 billion in research – equal to more than 5% of revenues. Key areas of research are safety, digitalisation/connectivity and environmental performance (ACEA, 2017b). Electrification and autonomous vehicles are widely discussed, but other technologies such as light weighting, powertrain improvements and emission abatement systems have been more relevant in delivering improvements in environmental performance so far.

The international agenda on climate change and the 2016 Paris climate agreement are highly relevant for the motor vehicle industry, but it is still unclear how industry and mobility will respond to high level environmental objectives. The EU is committed to agreements to reduce emissions as part of the Kyoto Protocol. As motorised transport is one of the major contributors to CO\textsubscript{2}, growing pressure to reduce emissions will also have to address transport.

There has been limited action to address transport emissions though with no net impact – reflected in the overall increase in emissions from transport since 1990. Nevertheless, evidence suggests that EU emissions standards have reduced average tailpipe emissions from passenger vehicles. This can be seen by contrasting increased vehicle activity (based on passenger km) against total stock emissions for cars (EC, 2017e).

Though emissions reductions can be achieved in conventional engines, more stringent targets on emissions reduction will likely lead to more investment in e-mobility, including electrical and hybrid power trains. However, a rapid change is not anticipated; by 2020 it is expected that conventional combustion engines will still account for more than 90% of vehicles on the road (McKinsey, 2013) – this reflects the fact that stock turnover is not particularly rapid, with an average age of in-use vehicle of 10.7 years across the EU as a whole\textsuperscript{41}.

The emissions scandal involving VW, and subsequently other OEMs, had a significant impact on the industry. For VW the immediate implications were financial and political, as its business reported losses and chief executives faced criminal charges. For the wider industry this has had potential spillover effect on the demand for diesel vehicles and helped to create a political environment which is more conducive to supporting alternative drivetrains. In Germany, one implication of the scandal was that the highest court in Germany, the Bundesverwaltungsgerichts (Federal Administrative Court), made provisions for German cities and municipalities to ban diesels (and some polluting petrol vehicles). A number of German cities including Hamburg, Dusseldorf and Stuttgart will introduce bans shortly (Bundesverwaltungsgericht, 2018).

\textsuperscript{41} \url{http://www.acea.be/statistics/tag/category/average-vehicle-age}
The evolution of motor vehicles

According to the ICCT (2017c), the average fuel efficiency of new vehicles in the EU in 2016 is around 118gCO₂/km on the NEDC test cycle, which is 30% lower than in 2000. This is still 25% higher than the 2021 target of 95gCO₂/km.

For passenger cars, research from Ricardo AEA (CCC, 2015), suggests there are a number of fuel-efficient technologies that can improve fuel efficiency of conventional ICE vehicles. However, these technologies alone are unlikely to be deployed fast enough to meet the EC’s target for 2021 and are insufficient to meet the proposed targets for 2025 and 2030. Meeting these will require the deployment of advanced powertrains through a combination of hybrid, plug-in hybrid and battery electric vehicles.

For heavy good vehicles, there has not been much progress over the past decade in part due to a lack of targets on fuel efficiency. Additionally, there are challenges in the lack of incentive for fuel efficiency within the market for haulage services:

- Fuel expenses are covered by the clients as part of standard contracts
- The haulage sector has a large number of SME operators that lack the capacity to finance investments in more fuel-efficient rolling stock
- Technical challenges in supporting advanced powertrains

In the short to medium term, the deployment of fuel efficiency technologies to conventional ICE powertrains will need to play a substantial role in ensuring HGV reach the proposed targets of 15% by 2025 and 30% by 2030. Data on available fuel-efficient technologies for heavy duty vehicles from the ICCT (2017e) shows potential fuel efficiency improvement of 27% in the short to medium term and as much as 43% by 2030. The proposed standards for HGVs, announced in 2018, require emissions of new vehicles to be 15% lower in 2025 than 2019, and reach 30% below 2019 levels by 2030. The ICCT analysis suggests that this should be possible through the deployment of fuel-efficient technologies, although it is likely that advanced powertrains will make some inroads into the market (for example BEVs for use cases that require relatively low mileage, and FCEVs in use cases where long running distances are required).

Modal shift in transport demand

In many areas, there is very strong evidence demonstrating economic, social and environmental benefits of alternative transport modes, and the need to develop policies to support a shift in the modal share of transport away from personal motor vehicles towards a greater share of public and mass transit systems (e.g. trains, buses, taxis, and ferries) as well as soft mobility (e.g. walking and cycling). Potential benefits from reducing the modal share of motor vehicles includes reduced emissions, reduced congestion, and improved public health. Some of these benefits are also outlined in the European Commission’s Strategy for Low Emissions Mobility (COM/2016/501).

For public transport systems, net efficiencies are often determined by urban population densities and travel distances. Investments in infrastructure for alternative transports have arguably declined in comparison to other points in history. This is notably the case for rail transport, which has a declining share for journeys travelled. Today rail travel in Europe accounts for 8% of the passenger-km and 17% of the tonne-km, but just 2.5% of the emissions (Vieweg et al, 2017) (see Figure 5.2).

Figure 5.2 EU Passenger-km per mode 2015
Short distances for typical journeys could easily be covered on foot or by bicycle. In mechanical terms, the bicycle presents the single most efficient form of transport available today, but remains underutilised for many journeys. Soft mobility provides additional benefits in comparison to other forms of transport as a result of being zero emission, low noise, low cost, active/healthy, and placing low demands on infrastructure or public investment. Some European cities, such as Utrecht, Copenhagen and Amsterdam, benefit from high rates of cycling. Research by the European Cyclists Federation (ECF) demonstrated that cycling creates economic benefits of €513 billion — with the most significant benefits relating to health.

Promoting soft mobility cannot be achieved by single measures, but is more often the result of a range of initiatives, which simultaneously reduce the incentives to drive and make it easier to walk or cycle, resulting in changes to social norms.

Innovative solutions relevant to motor vehicles, including electrification, social enterprise and the digital connectivity of vehicles, can be just as relevant to other modes of transport. E-bikes for instance may help to overcome barriers to cycling, such as steep gradients in local geographies or old age. Likewise, docked and dockless bike sharing systems provide citizens with the convenience of cycling without owning a bike. Mobile applications and big data can also contribute to personal and network efficiency of public transit systems, reducing journey times and operating costs.

Overall, the evidence suggests that a significant modal shift away from motorised transport is not taking place in the EU, although there are measurable differences between Member States. This can be seen in Figure 5.3, which demonstrates that overall demand for passenger transport has grown on average in the EU28; however demand for car transport has grown at a greater rate. Some Member States, notably Italy and the Netherlands, show an overall decline in the demand for car transport (EEA, 2015).
5.2.1 The impact of changing consumer preferences

This section primarily focuses on personal motor vehicles and its conclusions can thus not necessarily be extended on to other types of motor vehicles such as buses, trucks, motorcycles. Changing consumer preferences have been driving a shift in the type and characteristics of motor vehicles, as well as their ownership model. Nonetheless, current trends and all major predictions forecast continued growth in motor vehicle demand (PwC, 2016). In terms of consumer preferences, the three major trends that can be identified are:

- Demand for low emission vehicles continue to grow at the expense of diesel-fuelled vehicles.
- Advanced driver assistance systems (ADAS) and the move toward autonomous driving is driving an innovation rush across both the traditional automobile industry and high-tech firms – shedding light on an increasing number of alternative business models and cross-sectorial partnerships.
- The combination of alternative business models, socio-economic factors and changes in urban landscapes and public awareness is contributing to changes in ownership trends, as well as to a limited extent, changes in demand.

Source: EEA, 2015
Since the “diesel-gate” scandal, carmakers have rushed towards the production of “cleaner” electric vehicles (EV). Nearly all major automakers have announced EV plans for the next few years. Pressure on the industry has intensified since a growing number of cities and governments have stepped up efforts to phase out diesel and petrol vehicles altogether in a near to medium-term.

Furthermore, the recent European Commission proposal for post-2021 CO₂ standards incentivizes EV sales by awarding less stringent CO₂ reduction targets to manufacturers that exceed Zero Emissions Vehicle sales targets. Financial incentives at the national level will also contribute to demand for low emission vehicles (ICCT, 2017a). Some predictions suggest that electric car sales will reach 28 percent, and those of electric buses could reach 84 percent, of their respective global markets by 2030 as a result of declining battery costs and large-scale manufacturing. The rapid growth of electric buses (outpacing electric cars) stands out particularly because “in almost all charging configurations [they have] a lower total cost of ownership than conventional municipal buses by 2019” (Bloomberg New Energy Finance). Nonetheless, in certain instances, the decline in diesel car sales has been offset by greater demand for petrol-fuelled cars, with important environmental implications (ACEA, 2017a). Moreover, underinvestment in electric vehicle recharging infrastructure across Europe, and the strong appeal of petrol and hybrid vehicles are still hindering the switch to electrically-chargeable cars on a large scale.

ADAS features such as automatic emergency braking and Vehicle to Vehicle (V2V) communication are increasingly being installed in vehicles. This is mostly driven by falling technology prices and the popularity of such features with consumers, governments and insurers. The associated long-term result will likely be the availability of fully autonomous, or self-driving vehicles. This spur in innovation is gradually blending the tech and automotive sector with the effect of reshaping the landscape of actors. While large technology firms like Apple and Google are developing vehicle sensors and ride-sharing platforms to gain competitive advantages in autonomous driving and capitalize on mobility as a service, some OEMs are moving to strategic partnerships with Blockchain/tangle technologies to cope with demand and grid integration for EVs (Deloitte, 2018). Furthermore, increased demand for EVs is driving suppliers higher up the supply-chain (in the traditionally oil and gas sector) to diversify their portfolio and contribute to infrastructure development (such as installing charging stations) (CleanTechnica, 2017).

Germany provides some valuable insights in how changes in the socio-economic situation of young adults in the country is linked to a decrease in car ownership, in contrast to older age groups. Contrary to the popular narrative, Germany is experiencing two major trends in regard to motor vehicles:

1. There is a trend towards one car per driver; car sharing within households is decreasing, not growing; and
2. There is a trend towards using privately owned cars less; i.e. there is a trend toward owning instead of using, as well as an increase in multimodality of young drivers (Kuhnimhof, 2017).

In spite of the growing political support for soft and shared mobility, and an increasing number of commuters in Europe, these trends reveal a limited impact on motor vehicle demand. Driving public awareness campaigns, as well as a number of local and institutional initiatives, are stressing the negative impacts of air pollution, and promoting the significant health and economic value of increasing bike-friendly infrastructure and public transport, including electric buses (European Commission, 2016a). While growing support for the use of public transport through fiscal incentives and improved services may contribute to the growing number of commuters in Europe and theoretically contribute to a decrease in the demand for individual motor vehicles, no evidence suggests any significant impact on demand currently (Eurostat, no date).

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42 For example, BMW plans to build a fully electric version of the Mini at its Cowley plant near Oxford from 2019. Volvo has announced that from the same year, all its new models will have an electric motor. VW has earmarked €70bn (£62bn) to produce battery-powered versions of all models by 2030.
43 https://www. scapeproject.eu/about/
44 http://www.cleanairbxl.be/
45 https://ec.europa.eu/transport/themes/urban/cleanbus_en
In summary, other than the switch to EVs as a result of incentives and air pollution awareness, environmental preferences -per say- may not have a significant impact on the demand. The shift in ownership due to socioeconomic factors and thanks to the sharing economy could potentially improve the efficiency of motor vehicles during their use phase, while increased infrastructure and technological advances may better integrate EVs into our energy systems. Despite some identifiable trends in terms of urban mobility, these fall short of having any significant impact on the overall demand for motor vehicles at this point.

5.2.2 Sustainability in the sector

The initial focus of regulations on motor vehicles was to improve passenger safety. Increasing attention has been given to regulating the environmental performance of motor vehicles, notably fuel consumption and the emission of specific pollutants. In this way, regulations have been applied to reduce vehicles’ consumption of fuel, their emission of pollutants including CO₂ and nitrous oxides, as well as addressing other issues such as noise pollution.

Regulations in European countries have focused on gradually increasing the stringency of emission limits for new vehicles – as defined through the Euro standards (1 – 6). However, existing standards and monitoring systems came under scrutiny after “diesel-gate” revealed several motor vehicle manufacturers were deliberately cheating emissions testing procedures and exceeding legal limits on pollutants. Unreliable information on CO₂ emission values for vehicles poses a barrier to the effectiveness of measures. As mentioned, evidence shows that CO₂ emissions in tests compared with real-world values are increasingly diverging. Between 2006 and 2016 the gap between test values and real-world values increased from 14% to 42% (ICCT, 2017e). This evidence suggests that improvements to emissions are potentially smaller than reported.

Since September 2017, new car models have to pass new type approval procedures, both via the on the road Real Driving Emissions (EC, 2017b) and in lab World Harmonised Light Vehicle Test Procedure (EC, 2017c), in order to drive on European roads. Further regulations exist for mobile air-conditioning units (Directive 2006/40/EC), and for reducing noise pollution from motor vehicles (Regulation (EU) No 540/2014).

At national, sub-national and city levels there are further regulations which can be applied in order to reduce the environmental impact of vehicles. At city level, in recent years there has been increasing interest in the application of the low emissions zones (LEZs). In Europe, many cities and towns have established LEZs. Brussels Capital Region, for instance, introduced a LEZ in January 2018. LEZs also provide a basis for regulating older vehicles which were introduced before more stringent emissions standards existed. Even stricter zero emissions zones (ZEZs) are foreseeable in the future.

Regulations on emissions from vehicles present a policy option which could help to dramatically reduce negative impacts from car exhaust emissions. These measures might also have complex implications for the development of the sector more widely, for example favouring the development of low emission, hybrid and electric vehicles, and how the production of these vehicles remains competitive on the global market. The development of LEZs reflects the extent of the air quality crisis in some European geographies, and the potential for municipal policy makers to develop more stringent policy than EU-wide standards.

Sustainability may have a different meaning depending on whether one considers the sustainability of motor vehicles industry, or sustainability of mobility as a whole. At the European level there are a number of measures in place to define what sustainability might mean in different contexts.

In November 2017, the Commission launched a new Clean Mobility Package containing a number of measures with a particular focus on reducing the emissions from motor vehicles, and in order to “boost the demand and supply of clean mobility solutions” (EC, 2017d). Earlier in May 2017, the Commission launched the Europe on the Move Communication (COM/2017/0283). Linked to the Communication there are a number of legislative proposals covering issues such as road infrastructure charging (Eurovignette and European Electronic Toll Service), haulage services and goods vehicle leasing. The European Strategy for Low-emission Mobility provides a broader overarching strategy for sustainable mobility, focusing on increasing the efficiency of the transport system overall, promoting the electrification of transport, moving
towards zero emissions vehicles, and engaging local authorities in promoting more sustainable modes of
transport including public transport and active travel, in order to reduce pollution and congestion.

The environmental impact of battery manufacture
The shift towards e-mobility is a key step in reducing the environmental impact of the motor vehicles
sector. However, batteries are a key enabling technology in this shift, and there is a substantial
environmental footprint associated with the manufacture of these. As part of this analysis, an LCA of
battery manufacture was carried out, with the specific aim of comparing the environmental footprint of
manufacture in the EU and China – to address the question of whether onshoring battery cell manufacture
will lead to improved environmental outcomes at a global scale. The full analysis is presented in the motor
vehicles Annex report, but summary results and interpretation are presented below.

Figure 5.4 shows the impact categories used and demonstrates that across most indicators, battery packs
produced in Europe exhibit a lower environmental impact compared to China. In particular, the impact on
human health toxicity, as well as water pollution in Europe, is less than one-third that of China. In addition,
the climate change impacts in Europe are approximately half that of China. The same is also true in terms of
air pollution. There are certain impact categories where Europe does not fare as favourably compared to
China. Overall traction battery production in Europe requires only marginally less water and land use and
involves as much resource use impacts. However, resource use impacts need to be interpreted with
cautions. The largest component in the resource use category would be the Battery Management System,
which has multiple electronic (and thus rare-earth element) inputs. All such electronic inputs are only
available as “Global” datasets in Ecoinvent. Thus, the available data does not allow a comparison between

Figure 5.4 Ratio of impact assessment results for a battery pack produced in the EU vs China

EU and China.

Note(s): 100% = equal impacts; less than 100% = Chinese battery pack has higher impacts. Difference from 100% = how
much more/less impactful the EU battery pack is. In general, differences of less than 10% are likely not significant, given
the uncertainty of the Ecoinvent data. “CTU” is a chemical toxicity unit (similar to a CO2-equivalent); Sb is the element
antimony.

The cooling system exhibits the most significant fluctuation among examined impact categories. Whereas
land use, air pollution and climate change impacts are distinctly more favourable in Europe, most other
environmental impacts are more or less the same in both examined regions. Most notably, the production
of cooling systems in China seems to require less water than in Europe, primarily due to lower water
consumption in the aluminium production process.

Conversely, battery management system (BMS) production is relatively consistent among all examined
impact categories. Further, it also displays approximately the same environmental impact for production in
both Europe and China. It is absolutely critical to bear in mind that these results can largely stem from the
fact that Ecoinvent’s database does not offer regionalised datasets for electronics components, which are
over 50% of the BMS by mass, and can be expected to contribute strongly to impacts due to inputs of rare-
earth elements.
Next, packaging production demonstrates tangibly lower environmental impacts for Europe in six out of eight categories. Resource and water use impacts, on the other hand, are essentially the same in Europe and China.

Last but certainly not least, the environmental impacts from battery cell production in Europe are also substantially lower than those in China. In five out of eight categories the European environmental impact is barely half of the Chinese. What is more, in three out of those five categories the European impact is between 12% and 24% that of the Chinese. In other words, producing battery cells in Europe leads to much lower impacts in terms of human toxicity and water pollution, as well as reasonably lower impacts on air pollution and climate change. This finding is significant, insofar as battery cells comprise almost two-thirds of the battery pack by weight. As a result, they account for the better part of the overall environmental burdens caused by traction battery production in both Europe and China.

*Figure 5.5 Ratio of impact assessment results for a battery pack produced in the EU vs China, broken down by battery pack components*

Note(s): 100% = equal impacts; less than 100% = Chinese battery pack has higher impacts. Difference from 100% = how much more/less impactful the EU battery pack is. In general, differences of less than 10% are likely not significant, given the uncertainty of the Ecoinvent data.

In summary, the results generally give credence to traction battery manufacturing in the EU as opposed to China, at least insofar as environmental impacts are concerned. The battery cell is by far the largest contributor to environmental impacts for all impacts apart from resource use. For resource use, the lack of geographical differentiation in Ecoinvent does not allow a true comparison between EU and China, given that electronics are a significant user of rare-earth elements. Water use is the only category where the EU performs worse than China, specifically to do with water use in aluminium production. Whether this is a true result or a quirk of the Ecoinvent data requires more detailed analysis. In any case, worse performance in a single impact category does not strongly change the conclusions drawn.

This suggests that moving battery cell production from China to the EU would have beneficial environmental impacts, and therefore should be considered as a potential area for further policy action.

### 5.3 Future policy priorities

#### 5.3.1 Links between the sector and the Sustainable Development Goals

Although motor vehicles are not addressed explicitly within the Sustainable Development Goals (SDGs), the development of sustainable mobility systems is an intrinsic facet of several of the goals. Relevant targets include:
- SDG 3 on health (3.6 road traffic injuries, 3.9 mortality attributed to ambient air pollution),
- SDG 7 on energy
- SDG 9 on resilient infrastructure
- SDG 11 on sustainable cities (11.2 sustainable transport systems, 11.6 air quality)
- SDG 12 on sustainable consumption and production (12.c Rationalize inefficient fossil-fuel subsidies)

5.3.2 Policy options

The first issue to address when setting out policy options related to motor vehicles is to set out the goals, and understand how those goals might be achieved. The overarching aim of environmental policy in this sector is clear; to reduce the environmental impact of the manufacture and operation of motor vehicles, through improvements in the way that such vehicles are manufactured and used.

The follow-on issue is then how such outcomes can be brought about. **There are, broadly, three ways that reducing the environmental impacts of the manufacture and use can be achieved;**

- Ensuring that production of components and vehicles is carried out in the locations where environmental damage can be most limited
- The development and deployment of incremental improvements in vehicle technology
- Wholesale technological shifts which substantially alter the manufacturing and use processes (and therefore can fundamentally shift the environmental impacts)

The first addresses the fact that, with regards to global production, some regions (such as the EU) have tighter environmental legislation than others, and therefore production carried out in these regions will typically have a smaller environmental impact than production in other locations. In terms of motor vehicles, the economics of transportation mean that most components are manufactured either within or in close proximity to the European market; the notable exception currently is batteries for use in electric vehicles, which are currently primarily produced in China and shipped into the EU.

The second and third both relate to the deployment of new technologies, but with the potential for different magnitude of environmental impacts. The second category would include the incremental technology improvements to motor vehicles that have been seen since the initial introduction of EU emissions standards, such as regenerative braking, which serve to improve the fuel efficiency of vehicles (and therefore reduce the environmental impacts of use). The third group covers more substantial technology shifts, such as the replacement of internal combustion engines with electric motors (which completely alters the environmental impacts of use, removing the combustion of fossil fuels within the vehicle), but also technologies which change how vehicles are used, such as the increased take-up of shared mobility, and (in the longer term) the deployment and take-up of autonomous vehicles.

In this light, the following potential policy options to realise such changes have been explored:

- Production taxes (e.g. extraction charges for raw materials)
- In use vehicles specific taxes (i.e. registration taxes, annual circulation taxes, and company car taxation)
- In use energy specific taxes (i.e. fuel taxes)
- In use road charges (e.g. tolls, congestion charges, and parking fees)
- End of life charges and deposits

Market based instruments are one types of measure which can correct market failures, by which the costs from motor vehicles are poorly internalised. The existence of such instruments and their design can also be used to favour the manufacture or purchase of less polluting vehicles, or can lead to less polluting in-use behaviour, including an overall reduction in the distance travelled and diversion of traffic away from problematic areas. At the European level, the directive on passenger car related taxes (COM/2005/261) and the energy taxation directive (2003/96/EC) provide the main fiscal orientated instruments relevant to motor vehicles.
In practice, fiscal measures are national competences, and MS governments determine the design of fiscal instruments addressing motor vehicles. There is significant variation in practice on vehicle and transport fuel taxes between member states. The most important fiscal measures on motor vehicles are the taxes on the registration of newly purchased vehicles, circulation taxes, and fuel taxes (on petrol and diesel). Various factors determine rates in the MS.

The design of specific measures as well as the combined policy mix can help to support sustainability mobility systems. Taxation policies can help to determine new motor vehicle purchases based on environmental criteria – for example vehicle emissions. New car purchases are important, as though they represent a small portion of the whole fleet in any given year, they will remain in the car fleet for years and determine future impacts – for example when sold on second hand markets.

Increasingly EU MS have adopted vehicle taxation systems based on CO₂ emissions or fuel consumption. All MS apply VAT to new car purchases, but some apply exemptions such as those for EVs. The rates of VAT vary between countries, but a legal minimum requirement is for MS to have VAT of 15%. Most MS also apply a one-off tax on the purchase of new vehicles in the form of a registration and sales tax, as well as circulation taxes or periodical taxes. The rates for these taxes are usually determined by the characteristics of the car (weight, engine capacity, fuel consumption, CO₂ emissions).

Purchasing taxes are seen to be a key tool in reducing emissions from vehicles, providing a strong upfront signal to buyers. The Netherlands, Denmark and France, are noted for having high incentives for low emission vehicles. Recurring taxes on ownership, such as road taxes based on engine power, size or cubic capacity can also help to determine vehicle purchasing behaviour, but are potentially less effective as they occur in the future, and they also have no impact on usage.

Company cars represent a significant number of new car purchases – for instance in Germany 65% of all new passenger vehicles are registered to companies. Company car taxation can also be used to support a shift to low emissions vehicles. Most company car taxation is taxed as income, and few MS tax company cars on the basis of CO₂ emissions. The current design of company car taxation in most countries is based on the list price of a vehicle, however the list price doesn’t represent the financial benefit of private use of a company car. Consequently, company cars represent a subsidy to their users, and the size of the subsidy increases with the size of income or the car purchased. Poorly designed company car taxation can encourage people to purchase more polluting cars and use their cars unnecessarily.

LCA analysis of the environmental impact of alternative fuel technologies

While many LCA studies exist comparing different vehicle technologies, their comparison is difficult and shows wide divergence. Nordelöf et al. (2014) review 79 vehicle LCA studies, which can be divided into two main types – well-to-wheel (WTW) studies (covering just the life-cycle of the fuel, including its use phase) or “full” LCAs (which also include the production of the vehicle itself).

Reviewing WTW studies finds important determinants of GHG performance to be electricity production, degree of vehicle electrification and driving modes. Comparing BEVs powered with different electricity mixes modelled after a large EU study, grid GHG intensities above approx. 900 g CO₂eq/kWh (roughly that of oil-fired power production in the study) lead to BEVs emitting more than the reference average EU vehicle. The study shows that a decarbonized grid leads to substantially lower BEV WTW emissions. For degree of vehicle electrification, BEVs, PHEV (plug-in hybrids) and E-REVs (extended range EVs) are compared to a reference EU vehicle, with petrol as a liquid fuel and an EU-average grid mix for electricity consumption. Given an EU-average electricity mix and compared to the reference EU vehicle’s 143 gCO₂eq/km intensity, all electric vehicles perform better across the range of uncertainty in the results, with BEVs performing best with a maximum 76 gCO₂eq/km intensity, while PHEVs maximum intensity is worst at 126 gCO₂eq/km.

It is worth noting that a large amount of WTW studies are based on laboratory-tested fuel consumption figures (as per the New European Driving Cycle). A review by Fontaras et al. (2015) shows that such figures do not correspond well to real-life driving conditions with a divergence of 30%-40%, a conclusion also

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46 Although not all registered company cars are also used for private use.
highlighted in ICCT (2017). Further still, the charging time-of-day of EVs can influence their GHG footprint due to the patterns of electricity supply in response to demand (Messagie et al., 2014 find a factor of 2 difference in the GHG intensity of the Belgian grid). Again, focusing on Belgium, Rangaraju et al. (2015) find charging BEVs in off-peak periods (e.g. at night) is beneficial for GHG emissions compared to charging in periods of peak demand.

The so-called equipment cycle (impacts from production of the vehicle itself) is important for the full life-cycle performance of vehicles and its inclusion provides a more comprehensive view of the vehicle’s impact. The Nordelöf et al. (2014) review shows that in general (85% of the 79 studies examined), WTW GHG emissions dominate the full vehicle life-cycle, including for ICES. The impact of the vehicle’s production increase with increasing electrification (i.e. largest for BEVs) due to the increasing importance of additional components, chiefly the battery pack, but WTW emissions still remain the most significant contributor. Nonaka and Nakano (2010) note that the balance between vehicle production and WTW emissions varies between countries and is sensitive to the assumptions on average driving distance, but that BEV and PHEVs always increase in benefits compared to ICE vehicles with longer lifetime distances driven.

At the full vehicle life-cycle level, van Mierlo et al. (2017) demonstrate in a Belgian analysis that BEVs and PHEVs have lowest GHG performance across a range of options including diesel, CNG and biomethane ICES. BEVs have lowest life-cycle impacts for photochemical oxidant formation. BEVs and CNGs have comparable life-cycle PM emissions, though when considering local (at point of use) and external occurrence (e.g. due to mining), BEVs are best-performing out of all compared vehicles (due to zero tailpipe emissions). Time of charging of EVs is also shown to be important for overall emission performance. Van Miero et al. also conclude that the battery pack is the most important component in terms of impacts in the vehicle production stage.

Related to resource use are the toxicity impacts of resource extraction. Less data is available but overall, batteries which lack certain metals (nickel, cobalt) perform better due to less impactful production but also end-of-life handling (lithium iron phosphate batteries perform best in the Peters et al. review). Overall for non-GHG impacts, Peters et al. note high uncertainties due to the limited amount of data obtained in the review. However, normalizing all considered LCA impacts to a common endpoint (all impacts are converted to a comparable unit), the study ranks resource depletion as the largest environmental impact, followed by acidification and human toxicity; GHG effects are fourth. Though endpoint normalization in LCA is uncertain and caveat-prone, it is nonetheless useful to illustrate the importance of non-GHG impacts of battery production.

In summary, battery electric vehicles outperform other fuel technologies in terms of GHGs but with the electricity grid GHG intensity being a major determinant and with off-peak charging being important. This outperforming is most pronounced in an urban (and somewhat so in a suburban) setting and is eroded on highways due to limited potential for regenerative breaking and no idling of ICES. The battery pack & related components are the prime source of non-GHG impacts in BEVs, with end-of-life management of batteries being important for mitigating these impacts.

The socioeconomic and environmental impacts of decarbonising road transport

The modelling tools applied in this analysis are not well suited to understanding the impact of specific taxes on the deployment of motor vehicles. Instead the modelling focuses on the socioeconomic outcomes of successful policy implementation; i.e. what is the economic outcome of the deployment of e-mobility in Europe?

Two scenarios have been modelled; one in which fuel-efficient technologies have been deployed (TECH), with no shift in powertrains (i.e. no increase in sales of battery electric and plug-in hybrid vehicles), and another in which this technology deployment is accompanied by a rollout of advanced powertrains (EV). The modelling is carried out for Germany only – this is to provide a ‘high end’ estimate of the economic impacts of the transition; Germany has a large motor vehicle industry, and therefore would expect to experience larger absolute changes in economic outcome from a shift which impacts substantially on the traditional motor vehicles sector.
To model the impacts of these uptake scenarios, CE’s vehicle stock modelling tool (VSM) and CE’s macroeconomic model (E3ME) were used. The vehicle stock model calculates vehicle fuel demand, vehicle emissions and vehicle prices for a given mix of vehicle technologies. The model uses information about the efficiency of new vehicles and vehicle survival rates to assess how changes in new vehicles sales defined in the uptake scenarios, affect stock characteristics. The model also includes a detailed technology sub-model to calculate how the efficiency and price of new vehicles are affected, with increasing uptake of fuel-efficient technologies. The outputs of the VSM are then used as inputs to E3ME, an integrated macroeconomic model, which has full representation of the linkages between the energy system, environment and economy at a global level. The high regional and sectoral disaggregation (including explicit coverage of every EU Member State) allows modelling of scenarios specific to Europe and detailed analysis of sectors and trade relationships in key supply chains (for the automotive and petroleum refining industries). E3ME was used to assess how the transition to low carbon vehicles affects household incomes, trade in oil and petroleum, consumption, GDP & employment.

Over the long term, both scenarios show positive impacts on GDP relative to the reference scenario. Figure 6.5 shows the relative changes out to 2050. The main driver of the increase in GDP is the reduction of fossil fuel demand in the scenarios. As Germany is dependent on imports of crude oil and manufactured fuel to meet the demands for road transport, a reduction in demand for oil in the German economy leads to an improvement in the balance of trade and so leads to an increase in spending on good and services on goods with a higher domestic content.

This is partially offset by high prices for motor vehicles, which raises the costs for consumers and diverts spending away from other parts of the economy and towards the manufacture of motor vehicles. In the case of Germany, as the economy has a strong position in the manufacture of motor vehicles, this diverting of spending would still be to a mostly domestic sector. For other member states which do not have a large vehicle production sector this would have a larger negative impact. However, we see in the short term it has a noticeable negative impact on GDP as the increase in the cost of motor vehicles lead ahead of fuel savings over the lifetime of the vehicle.

Even then, we would expect the reduction in oil imports to dominate as over the medium to long term, as the reduction in the lifetime cost of owning and refuelling a vehicle with fuel efficient technologies or an advanced powertrain outweighs the initial cost of the technology.
In explaining the relative impacts of the scenarios, we can also see the dominance of the oil import effect drives this as by 2050 the reduction in oil in TECH is around half that of EV and overall the GDP impact of the TECH scenario is slightly less than twice the impact of EV by 2050. In the short to medium term out to 2030, the impact is broadly similar, as the difference in oil imports between the scenarios is small enough to be offset by the slightly higher vehicle costs for advanced powertrains.

Table 5.1 Economic and environmental impacts for different vehicle uptake scenarios, Germany 2050

<table>
<thead>
<tr>
<th>Results</th>
<th>Scenarios</th>
<th>TECH</th>
<th>50% of cells imported</th>
<th>All cells domestically produced</th>
<th>All cells imported</th>
</tr>
</thead>
<tbody>
<tr>
<td>Battery Assumptions</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>GDP (%)</td>
<td></td>
<td>0.4%</td>
<td>0.8%</td>
<td>1.3%</td>
<td>0.2%</td>
</tr>
<tr>
<td>Employment ('000s)</td>
<td></td>
<td>132</td>
<td>229</td>
<td>342</td>
<td>169</td>
</tr>
<tr>
<td>Investment (%)</td>
<td></td>
<td>0.5%</td>
<td>1.1%</td>
<td>1.4%</td>
<td>0.8%</td>
</tr>
<tr>
<td>Tailpipe CO2 emissions from passenger cars (MTOE)</td>
<td></td>
<td>-47</td>
<td>-99</td>
<td>-99</td>
<td>-99</td>
</tr>
<tr>
<td>Fossil Fuel demand from passenger cars (MTOE)</td>
<td></td>
<td>-699</td>
<td>-1459</td>
<td>-1459</td>
<td>-1459</td>
</tr>
<tr>
<td>Electricity demand from passenger cars (MTOE)</td>
<td></td>
<td>0</td>
<td>244</td>
<td>244</td>
<td>244</td>
</tr>
</tbody>
</table>

Notes(s): All values represented as difference from baseline REF scenario

In terms of employment, we see a net increase in jobs in both scenarios as with the overall economic impact on GDP. However, underneath the aggregate impact the benefits of the transition are not felt equally across all sectors of the economy as shown for the EV scenario in Figure 5.7. The largest increase is in the service sectors of the economy, fuelled by additional consumer expenditure on goods and services other than fossil fuels. Manufactured fuels show a decline in jobs from lower fossil fuel demand although this is quite small due to the low employment intensity in refining. Similarly, the electricity sector sees a modest increase in jobs from the additional electricity demand for electric vehicles.

Figure 5.7: Employment impact by sector in the EV scenario
The shift in motor vehicles and electrical equipment reflect the changes in the automotive supply chain due to electrification, as production moves away from internal combustion engines sourced from the motor vehicle sector to batteries and electric motors sourced from electrical equipment sector. Overall the net impact on the automotive supply chain is positive reflecting the higher value of advanced powertrains.

In addition, the transition to EVs has the potential to substantially reduce the in-use emissions associated with motor vehicle use. **CO2 emissions are 33% lower in the EV scenario compared to reference by 2030 (73MTCO2 vs 109 MTCO2 in the reference), and by 2050 CO2 emissions are 93% lower in the scenario, at only 7MTCO2 (compared to 106MTCO2 in the reference).**

*Figure 5.8 MT CO2 emitted by the German motor vehicle fleet*

Reductions in NOx and PM10s occurs more slowly than the reduction in CO2, as it is linked only to the deployment of EVs (with no impact from the improved fuel efficiency of new ICE vehicles). By 2030, NOx emissions are 7% lower than in the reference case, while PM10s are reduced by 9%. However, by 2050, as many more EVs are deployed into the vehicle fleet, emissions of these pollutants are reduced substantially; NOx is reduced by 88%, to less than 8MT, while PM10 emissions are reduced by 89% to almost zero.

Note also that both NOx and PM10 emissions fall substantially in the reference case; this is due to the fact that all new vehicles sold conform to the latest Euro 6 standards, and these vehicles enter the stock to replace older vehicles which conformed to earlier, less restrictive, Euro standards.

**The impact of the location of battery manufacturing**

The results of the scenario are sensitive to a number of assumptions about the transition to electrified vehicles. One of largest considerations is the location of battery cell manufacture. Currently battery cell manufacture is largely situated outside of Europe, but it is expected as the production of electric vehicles expands that such activity will start to take place within Europe. Although it is unclear exactly where in Europe this will take place, in this analysis, we have taken a simple assumption in the central case that Germany imports 50% of the battery cells it needs. To explore the potential range of impacts, we have also explored scenarios variants where Germany produces all battery cells domestically or imports all battery cells. In Figure 6.8, we show the GDP impact of the scenarios under these different assumptions of battery production.
From the modelling, we see that the location of battery cell production could have a substantial impact on overall economic effect of a shift to advanced powertrains. However, even in the most pessimistic case where all battery cells are imported which would have a negative impact on the German trade balance, the overall impact of the transition is positive for the German economy.

Figure 5.9: GDP Impact from battery cell production sensitivities

As mentioned above, the impacts as assessed for Germany will not entirely reflect the impacts of the rest of the EU and would be expected to represent the ‘top end’ of potential impacts relative to the rest of the EU, where the manufacture of motor vehicles is not such a large part of the economy. However, regardless of the magnitude, the key drivers of the economic results hold true for the EU28 as much as for Germany. The clearest example is with oil imports, which all member states are reliant on to fuel their passenger car fleet and it is reduction of imported oil which drives the economic impacts. This result is borne out in Harrison (2018) where decarbonising transport across the EU leads to smaller scale impacts for the EU28 from a similar decarbonisation scenario; an increase in GDP of +0.5% by 2050, while reducing tailpipe CO2 emissions from passenger cars by around 90%.

5.4 Conclusions

Motor vehicle manufacture is an economically important and politically significant sector. Links between the sector and the environment are many and significant: production is important in the scale of raw material and energy use; product design is important from the perspective of the circular economy with regards to end-of-life vehicle (ELV) waste; and product use is crucial because of localised noise & air pollution and emissions of greenhouse gases (GHGs). A transformation of the motor vehicles sector is a necessary component of achieving the SDGs and EU-specific goals. Take-up rates of electric vehicles are on the increase, and are likely to increase as cost parity is achieved.

However, there remains a substantial role for policy in facilitating and accelerating this transition. Market-based instruments such as production and in-use taxes can be used to encourage take-up of advanced powertrains on the demand side, in addition to the supply-side measures (such as tightening CO2 standards) that have so far been the primary focus of policy at the EU level.

Several scenarios have been modelled to estimate the impacts of such a transformation, i.e. the deployment of more fuel-efficient vehicles and advanced powertrains into the European fleet. Germany as
a large EU member state, where motor vehicle production plays a large role in the economy was chosen for
the modelling, as it has extensive and robust data available on sales and the characteristics of the stock.
Conclusion is that a reduction in oil imports would dominate over the medium to long term, as the
reduction in the lifetime cost of owning and refuelling a vehicle with fuel efficient technologies or an
advanced powertrain outweighs the initial cost of the technology. Also, the investment in charging
infrastructure boosts investment in the electricity sector.
6 Water treatment & supply

The water treatment & supply sector, while directly drawing on natural resources itself through the extraction of water, also has links with other sectors with substantial environmental impacts. Water is a key input to both agriculture and some manufacturing processes. At a broader level, although most of the rest of the economy is not heavily reliant on water in value terms as an input to production, there are clear implications from a lack of access to clean water that make it an essential input to almost all sectors of the economy; for example, services firms operating out of offices spend very little money on water supply services, and they form a very small part of the total costs of inputs to these firms; however, it would be very difficult for these businesses to operate without access to water for drinking and hygiene purposes.

As such, a steady supply of clean, potable water is essential for the successful operation of the European economy. European legislation already aims to provide such a supply, adhering to minimum standards. However, while investment in water supply and use is essential on social and environmental grounds, these arguments do not necessarily translate into a compelling financial case for investment. There is evidence that current investment levels are sub-optimal relative to the economic, social, and environmental benefits that would accrue from additional sector investment (OECD-WWC-Netherlands, 2017; EIB, 2016). This situation is well recognised by EU policy makers and poses the central question as to how public institutions in Europe work to increase investment levels.

6.1 The current environmental impact of the sector

6.1.1 Overview

Water treatment & supply has been identified in this study as a sector of interest, on the basis that it is an environmental services sector, with a strong influence on other sectors and strong links to the SDGs. In addition, it has a strong dependence upon ecosystem services, and a number of policy challenges exist around the abstraction, reuse and supply of water, to both households and industry.

The focus of this analysis was on access to water in Europe. The Drinking Water proposal includes provisions on access to water and aims to improve the quality of drinking water and provide greater access and information to citizens. It draws heavily upon the water investment gap in Europe, to explore the economic aspects of the sector – specifically, what are the requirements made of the sector by existing EU water legislation, what is the size of the investment ‘gap’ required to be bridged to ensure that all existing legislation can be met, how might such a gap be met, and what would be the socioeconomic impacts of addressing that gap.

6.1.2 The investment gap for infrastructure in Europe

Public investment in infrastructure in many Member States has fallen, both as a percentage of public spending and as a percentage of GDP. In the EU-15 infrastructure spending fell by more than 1% of GDP between 1970 and the financial crisis in 2007-2008.

The period after the financial crisis led to a slowdown in investment amongst developed countries. Gross fixed capital formation (GFCF), which provides a measure of investment in the economy, fell below the level seen in the pre-crisis period (1995-2006). The difference was €150bn in 2015Q3 across the EU Member States. Overall, investment was 2% lower than the pre-crisis average in the EU (and 2.2% lower within the Euro area) (EIB, 2016b).

Forecasts suggest that other macroeconomic challenges, such as an aging population and the consolidation of public budgets, will lead to further reductions in spending on infrastructure (Heise et al, 2014).

Addressing Europe’s investment gap was central to the Juncker Plan (Investment Plan for Europe) and the development of the European Fund for Strategic Investments (EFSI), which aims to leverage €315bn of investment across the EU economy.

Water infrastructure faces challenges in terms of financing. However, existing attempts to assess the investment gap of the European water sector are scarce. Below we summarise existing estimates, which
can broadly be broken down into three categories; assessments carried out at the global, European and Member State level.

6.1.3 Global assessments of the investment gap

Although the literature regarding investment needs in water infrastructure in Europe is limited, there have been several attempts in a global context to estimate current spending and the amount of capital required to meet the world’s water supply and sanitation needs. These attempts were typically a part of a broader process of assessing the global needs for infrastructure including transport, energy, and telecommunications.

The first comprehensive study reviewed was undertaken by Fay and Yepes (2003), who valued the global infrastructure stock in 2003 at $15 trillion, of which 7.5% ($1.125 trillion) accounted for water supply and sanitation infrastructure. The corresponding investment needs between 2005 and 2010 implied an increase in spending by 2.1%, 1.5%, and 0.4% per annum in Low-Income, Middle-Income, and High-Income countries respectively.

Using a sectoral bottom-up method, OECD (2006) estimated that the OECD countries together with Brazil, China, India, and Russia would need to invest $2,380bn in total in the 2000-2030 period on water supply and wastewater treatment. Current expenditure on water infrastructure in 2005 were estimated at around $576bn, which would need to increase to $772bn by 2015 and $1,038bn by 2025. Furthermore, Volume 2 of the same report suggests that OECD countries will have to increase their investment in maintaining, upgrading, and replacing existing infrastructure in the water supply and treatment sector by almost 50 percent (OECD, 2007).

The McKinsey Global Institute (2016) estimated the global infrastructure needs for water and sewage to be $500bn annually between 2016 and 2030, while the current investment in 2013 was $200bn, leaving a gap of $300bn. Bhattacharya et al. (2016) calculated the projected investment for the same period to be $0.9 trillion per year (the gap is due to methodological differences and the great complexity of such estimations).

6.1.4 European assessments

Very few studies have explored in detail the investment gap for the water sector in the European Union. In most cases attempts to do this are done by extrapolating from best available data, or have a focus on a specific aspect of the water sector. A summary of the relevant assessments focusing on the EU water sector are given below:

EIB – Restoring EU Competitiveness

The European Investment Bank publishes reports analysing and forecasting their financing activities in the EU and third countries. As part of the EIB’s engagement with the Investment Plan For Europe, launched in 2015 and including the EFSI (see below), the EIB carried out assessments of potential investment gaps which might hinder Europe’s competitiveness (EIB, 2016). The EIB carried out an assessment of the investment needs for water with respect to:

- Water risk management (including scarcity and flooding)
- Water supply and treatment
- Compliance with water legislation
- Water infrastructure in urban areas
- R&D for competitiveness in European water technology

A summary of this assessment is given below.

<table>
<thead>
<tr>
<th>Investment need/objective</th>
<th>Annual investment (€bn)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Required</td>
</tr>
<tr>
<td>Water security (including flood risk management)</td>
<td>15</td>
</tr>
<tr>
<td>Investment need/objective</td>
<td>Annual investment (€bn)</td>
</tr>
<tr>
<td>---------------------------------------------------------------</td>
<td>-------------------------</td>
</tr>
<tr>
<td></td>
<td>Required</td>
</tr>
<tr>
<td>Compliance and rehabilitation of Europe’s water infrastructure</td>
<td>75</td>
</tr>
<tr>
<td>Enhancing waste management/materials recovery</td>
<td>8</td>
</tr>
<tr>
<td>Additional needs for resilient and efficient urban infrastructure</td>
<td>40</td>
</tr>
<tr>
<td>TOTAL:</td>
<td>138</td>
</tr>
</tbody>
</table>

Source: EIB, 2016

In general, these values were based on extrapolating data for a single Member State or for another region (e.g. the U.S.) across the whole EU28 using a conservative conversion (EIB, 2017b).

**COWI - Urban Waste Water Directive (UWWD)**

COWI (2010) estimated the costs of compliance with the UWWD in all EU Member States, in order to assess the financing gaps. To do so, they apply cost functions derived from their own model, FEASBLE. This includes investment costs for additional collection and treatment infrastructure, but neither the costs of renovation of existing systems nor the costs related to sludge treatment and disposal are included. The information sources are national registries, additional information at the Member State level and a survey. The study estimates that the total expenditure required in 2006 to reach full compliance with the UWWD in the EU was €45.3bn. 63% of this amount (€28.3bn) was due to investment costs for waste water collection (Article 3 of the UWWD), 28% (€12.5bn) to the advanced treatment required by Article 5, and the remaining 10% (€4.5bn) to costs for generic treatment (Article 4). COWI also estimated re-investment needs for the Member States who have full registry file data, including both the annual re-investment needs due to the current infrastructure and those related to the new infrastructure still to be built.

**Eurostat - Protection expenditures for waste water**

Eurostat collects annual data for all EU Member States on protection expenditures related to wastewater, which include pollution prevention, sewerage networks, wastewater treatment, treatment of cooling water and all other activities and measures aimed at wastewater management. According to the Urban Waste Water Treatment Directive (Article 17) Member States must report to the the Commission on their implementation programme. There are various databases and indicators in use:

47 Based on the most recent Environmental protection expenditure accounts (env_ac_peps)

- In 2017, the environmental capital expenditure for waste water management was € 7 bn for industry, €11.8 for specialist producers, €10.6 bn for governments, totalling to €29.3 bn p.a. at EU level.
- In 2017, the total market output of companies working in the field of wastewater management in the EU28 was €44.5bn.

Based on Eurostat Structural Business Statistics:

48: Production of environmental protection services of corporations as specialist producers by economic characteristics [env_ac_pepssp]


In 2016, "Investment in equipment and plant for pollution control" in waste water was €2.1bn while "Investment in equipment and plant linked to cleaner technology (‘integrated technology’)" was €0.8bn. Therefore, total capital expenditure was €2.9bn in the waste water treatment sector.  

Based on the Environmental protection expenditure in Europe:

- In 2013, the total environmental protection expenditure (including Opex, Capex, Pollution treatment and prevention) on waste water management was €14.1bn by governments and €44.3bn by private and public companies specialised in sewerage, waste collection, treatment and disposal, and remediation activities.
- In 2013, the environmental investment (Capex) in waste water management was €5.4bn by governments and €13.8bn by private and public companies specialised in sewerage, waste collection, treatment and disposal, and remediation activities.
- In 2013, the environmental current expenditure (Opex) for waste water management was €6.1bn by governments and €30.5bn by private and public companies specialised in sewerage, waste collection, treatment and disposal and remediation activities.

Ecorys - Drinking Water Directive (DWD)

Ecorys (2016) calculated that the overall annual investment needed to provide drinking water in the EU28 was €46.5bn in 2014 and €630bn between 1998 and 2014, including costs due to ‘normal’ pipeline network (e.g. maintenance costs). This figure is based on the expenditure costs for drinking water in 6 Member States (as calculated by VEWA, 2015, through a survey) and extrapolated to the whole EU using data on total population and differences in income per Member State. Using a range of sources, including a survey and published studies, Ecorys (2016) estimated that around 16.5% of the total costs (€8.3bn in the EU28) can be attributed to the implementation of the DWD, i.e. €109bn over the DWD life span. According to a study carried out in Germany by Aquabench, which was used in the Ecorys (2006) report, costs related to drinking water provision are due to:

- Taxes, levies, fees, concession fees, Water abstraction charges (7%)
- Metrology / quality control (3%)
- Building management (5%)
- IT technical support processes (15%)
- Resource Management / Water procurement / Extraction / Processing (18%)
- Treatment of drinking water (18%)
- Imputed Costs, such as the pipeline system and overall amortization (33%)
- Other costs, such as travelling to international events (1%)

IEEP - Metering

According to IEEP et al. (2012) the installation of metering in all irrigated EU land could cost around €0.2bn and full-scale implementation of metering in the whole of EU would costs €3.0bn (calculated based on the French experience).

Investment in water quality and quantity

Spit et al. (2017) estimated the costs related to public expenditures for water quality, quantity and waterways, based on the values calculated by Tauw and Twynstra Gudde (2015) for the Netherlands, which are extrapolated to the EU28 using the benefit-transfer methodology. Such costs are additional to the costs of the drinking water sector mentioned above, because these expenditures reduce the costs for the water
sector by decreasing treatment costs and other costs for the water sector. The authors calculated that investment in water quality for the EU28 are around €39.5bn per year and those in water quality approximately €8.8bn per year. They also estimated that investment needed for the maintenance of water ways in the EU28 are between €45.6bn and €131.7bn per annum. Finally, they calculated the operational expenditure of economic sectors (intake, treatment, recirculation and discharge treatment costs) as the sum of the additional treatment required for abstracted water (€11bn) recirculation of water (€4bn) and the cost to discharge water having appropriate quality (€19bn).

6.1.5 Member state assessments

Within this scoping study several Member State level assessments of the investment needs for national water sectors have been identified. It is possible that more national level assessments have been carried out across the EU28 but it was not possible to review all national literature in the scope of this study.

Investment needs naturally vary between EU Member States. In central and eastern Member States, maintenance costs are perhaps the most pressing investment need. In western Member States with fewer pressing compliance issues, such as the Netherlands, prioritisation is a more involved process of weighing up costs/benefits, with approaches such as detailed impact assessments based on risk matrices employed. Network coverage is a prominent issue in multiple Member States, particularly in relation to urban wastewater standards, which necessitates large capital investment (D. Simidchiev, Hydrolia).

Spain

The Directorate General for Environment and the Centre for Hydrographic Studies (2017) summarised the foreseen investment in Spain required to meet the requirements of the Water Framework Directive (WFD) and water demand in general (regulation and transportation works), as well as the measures needed to mitigate the effects of floods and droughts, including the investment required by the Flood Risks Management Plans. The investment needed to increase water supply is estimated to be €3.4bn between 2016 and 2021, €2.8bn between 2022 and 2027, and €32.3bn between 2028 and 2033. The investment foreseen for flood prevention is €0.4bn in 2016-2021, €0.1bn in 2022-2027 and €0.04bn in 2028-2033. According to a recent study (SEPAN, 2017), the investment need for water infrastructure in Spain between 2017 and 2021 will be €12.0bn, including €0.6bn for water provisioning, €4.4bn for water treatment, €4.9bn for water distribution and €2.1bn for water regulation. 30% of this investment is needed to comply with the Water Framework Directive and the Flood Directive.

United Kingdom

The infrastructure investment planned for the water sector in the UK totals £5.03bn (€5.64bn) in 2016/17, £4.99bn (€5.60bn) in 2017/18, £4.66bn (€5.23bn) in 2018/19, £4.03bn (€4.52bn) in 2019/20, £0.5bn (€0.5bn) in 2020/21 and £0.1bn (€0.1bn) afterwards. Between 9 and 11% of these amounts will finance water and sewerage projects of above £50m (€56m), i.e. big projects such as the Thames Tideway Tunnel, a major new sewer system that is being built to protect the tidal River Thames from pollution and should be functioning by 2023. In addition, around £0.5bn (€0.6bn) will be invested in flood defences per year over 2016 and 2021, and £1.39bn (€1.56bn) of investment are planned after the 2020/21 programming period. In terms of past investment, according to Phippard S. (2015), investment from privatised water and sewerage companies during the first WFD cycle in the EU have been €30bn, of which €6.1bn was specifically to meet environmental obligations. These costs are recovered by billing of customers and result in a reduction of phosphate and ammonia input to water bodies.

Bulgaria, Hungary and Romania

A recent report from the EU Court of Auditors concluded that Bulgaria, Hungary and Romania will need to invest approximately €6bn by 2020 to improve access to quality water and to comply with the DWD. These countries received €3.7bn in regional development and cohesion funds, which played a significant role in improving drinking water supply, but further investment is needed from the Member States. In particular, only 62% of the Romanian population is connected to the public water supply system; in Bulgaria 60% of water is leaking from the supply network before reaching the final consumer, and in Romania the equivalent figure is 40%.
Bulgaria

In Bulgaria, the water network is extensive in terms of coverage, but often fails to meet modern engineering requirements, and is deteriorating more quickly than current levels of investment are able to address. The Bulgarian strategy for the water supply and sanitation (WSS) sector (Republic of Bulgaria’s Ministry of Regional Development, 2014) calculates that about BGN 12.2 (€6.2bn) will be needed between 2014 and 2023 to meet the national WWS requirements. BGN 6.7–7.2bn (€3.4 – 3.7bn) will be required for improving water treatment. Urgent investment needs for renewal and replacement investment in water supply (abstraction, treatment, transmission and distribution) are estimated at BGN 5.0bn (€2.6bn), of which BGN 0.4bn (€0.2bn) are costs related to water supply compliance costs. BGN 4.4bn (€2.3bn) are needed for wastewater collection, and BGN 2.8bn (€1.4bn) are required for wastewater treatment. EU funds can cover 30-40% of the required investment for WSS, while the remaining 60-70% will have to be financed by the government and utilities. Stakeholders generally agree with the assessment, but believe that the cost estimates may be slightly higher. To comply with the EU water acquis, plus sectoral effectiveness as per international benchmarks, stakeholders believe that the required investment is in the range of €5-6bn (D. Simidchiev, Hydrolia).
### Table 6.2 Non-exhaustive summary of data assessing the investment gap in European water infrastructure

<table>
<thead>
<tr>
<th>Source</th>
<th>Geographic</th>
<th>Sector focus</th>
<th>Actual estimate (€)</th>
<th>Investment needs estimate (€)</th>
<th>Methodology</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>OECD, 2006</td>
<td>OECD plus Brazil, China, India, and Russia</td>
<td>Water supply &amp; wastewater treatment</td>
<td>€462.8bn* by 2005</td>
<td>€1,915bn* in total from 2000-2030, €620bn* by 2015 and €833.4bn* by 2025</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mc Kinsey, 2016</td>
<td>Global</td>
<td>Water supply &amp; wastewater treatment</td>
<td>€180.7bn* in 2013</td>
<td>€450bn* annually between 2016-2030</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bhattacharya et al., 2016</td>
<td>Global</td>
<td>Water supply &amp; wastewater treatment</td>
<td></td>
<td>€813bn* annually between 2016-2030</td>
<td>Cost functions developed using the FEASBLE model, developed by COWI</td>
<td></td>
</tr>
<tr>
<td>COWI, 2010</td>
<td>EU</td>
<td>Urban waste water treatment</td>
<td></td>
<td>€45.262bn required in 2006 to reach full compliance with the UWWD</td>
<td>This figure only includes new investment</td>
<td></td>
</tr>
<tr>
<td>ESIF, 2014</td>
<td>EU (2014-2020)</td>
<td>Water supply</td>
<td>ERDF will cover 36% of the investment and CF the rest of it, which will benefit more than 12 million EU citizens</td>
<td>This will bring water supply to 56% of the EU population, which is currently without</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ESIF, 2014</td>
<td>EU (2014-2020)</td>
<td>Wastewater treatment</td>
<td>ERDF will cover 51% of the investment and CF the rest of it, which will benefit almost 17 million</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Source</td>
<td>Geographic</td>
<td>Sector focus</td>
<td>Actual investment estimate (€)</td>
<td>Investment needs estimate (€)</td>
<td>Methodology</td>
<td>Comments</td>
</tr>
<tr>
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<td>-------------------------------------------------------------------------------</td>
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<tr>
<td>EIB, 2016</td>
<td>EU (2007-2013)</td>
<td>Municipal and industrial water/wastewater</td>
<td>€30bn</td>
<td>€90bn a year for the period 2014 to 2020</td>
<td>Secondary resources</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Material recovery</td>
<td>€3bn</td>
<td>€8bn per year</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Urban water infrastructure</td>
<td>€13bn</td>
<td>€40bn</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Water technology (R&amp;D)</td>
<td>€4bn p.a.</td>
<td>€7bn per annum by 2020</td>
<td></td>
<td></td>
</tr>
<tr>
<td>GWI, 2016</td>
<td>EU</td>
<td>Water supply &amp; wastewater treatment</td>
<td>Annual increases (2-5% range) in investment to 2020, resulting in an average expected yearly investment of €33bn</td>
<td></td>
<td>Not open access.</td>
<td></td>
</tr>
<tr>
<td>Ecorys, 2016</td>
<td>EU</td>
<td>Drinking water provision</td>
<td>Costs related to the supply of water: €46.5bn in 2014, €630bn between 1998 and 2014; of this €8.3bn is due to compliance with the DWD</td>
<td>The expenditure costs for drinking water in 6MS, as provided by VEWA (2015) (which obtained it through a survey), are used to extrapolate the overall figure for the EU, based on information on total population and differences in income per MS.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Source</td>
<td>Geographic</td>
<td>Sector focus</td>
<td>Actual investment estimate (€)</td>
<td>Investment needs estimate (€)</td>
<td>Methodology</td>
<td>Comments</td>
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<td>----------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Split et al., 2017</td>
<td>EU</td>
<td>Public investment in water quality, quantity and waterways</td>
<td>Investment in water quality: €39.5bn per year; investment in water quality: about €8.8bn per year; investment in the maintenance of waterways: between €45.6bn and €131.7bn per year</td>
<td>The values calculated by Tauw and Twynstra Gudde (2015) for the Netherlands are extrapolated to the EU28 using the benefit-transfer methodology</td>
<td>These costs are additional to those calculated by Ecorys (2016) as the cost of provision of drinking water</td>
<td></td>
</tr>
<tr>
<td>EIB, 2017</td>
<td>Italy</td>
<td>Water sector</td>
<td>Investment gap in Italian water sector growing by around €3bn per year</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ASCE, 2016</td>
<td>USA</td>
<td>Water supply &amp; wastewater treatment</td>
<td>The total investment gap through 2025 is expected to be €94.9bn* and €137.4bn* by 2040 if left unaddressed.</td>
<td>National economic model</td>
<td>Provides a comparison for scale</td>
<td></td>
</tr>
</tbody>
</table>

*Note: The currency was converted from US Dollars to Euros using the corresponding yearly average exchange rate.
6.1.6 Environmental pollutant releases

The number of E-PRTR releases for the water treatment and supply sector that has all required data available for the calculation of environmental impacts\(^51\) fluctuates slightly, but is around 10% over the period 2007-16.

Apart from a slight divergence in the peak in 2011 and 2012, human and environmental health impacts closely mirror one another. There are two spikes in impact, and both relate to reported CO\(_2\) releases (there are no reported releases of CO\(_2\) in years other than 2011, 2012 and 2015). In 2011-12, there were substantial CO\(_2\) releases in Czech Republic, while in 2015 the source of the emissions was Germany. The remaining environmental impacts are largely attributable to methane releases, although there were also releases of nitrous oxide (2012-16) and ammonia (at a low level throughout the time period).

*Figure 6.1 The environmental impacts of pollutant releases from the water treatment and supply sector*

6.2 Current direction of travel

The investment needs regarding water in Europe are strongly linked to the implementation of EU water law. Meeting EU objectives requires investment. Therefore, a key consideration in examining investment needs is the extent of compliance with EU water law, i.e. where there is non-compliance this might indicate that further investment is needed and where there is compliance this might indicate that there is not an EU legal driver for further investment.

In considering the EU water acquis, focus should be on those measures which have the largest potential consequence for spending. These are:

- The Urban Waste Water Treatment Directive 91/271/EEC (UWWTD), which requires investment to collect and treat waste water.

\(^{51}\) The key characteristic that is often missing is the CAS pollutant code, which is required in order to assign a human health impact (measured in disability-adjusted life years) and ecosystem health impact (measured in species years lost).
The Bathing Waters Directive 2006/7/EC (BWD), which has required significant improvements/investment in sewage treatment to meet its objectives.

The Drinking Water Directive 98/83/EC (DWD), which requires investment for water distribution and treatment.

The Water Framework Directive 2000/60/EC (WFD), which sets objectives for all water bodies that may require investment to address pressures impacting on these objectives.

It is important to note that the compliance deadlines for the UWWTD, BWD and DWD have all passed and, therefore, non-compliance is strict legal non-compliance. For the WFD, the use of exemptions based on cost has effectively allowed Member States to delay meeting these objectives. Such exemptions can only be used for three River Basin planning periods and therefore all objectives should be met by 2027.

The stakeholders presented a unanimous opinion that the EU water acquis and compliance with EU policy is by far the principle driver of investment. While some projects can have better economic returns than others, or provide higher added social and environmental benefits, regulatory compliance is the primary consideration for decision-making. Following compliance, maintenance of sustainable services and higher efficiency are a major aim, through improving connectivity, infrastructure optimisation and aiming to cutting losses.

6.2.1 Sources of finance and financial instruments in theory

Water utility infrastructure is financed in a variety of ways, including private, public, national and international sources of finance – often in combination – and via a range of different financial instruments. Generally, recurrent financing is based on a combination of tariffs, taxes and transfers (the ‘3Ts’). This revenue stream is also used as basis for attracting and repaying private finance, including loans, bonds (debt) and/or equity, which is used to address financing gaps or meet short-term budgetary needs (Lago et al, 2011; OECD, 2009). A different mix of financial instruments might be necessary to fund each phase in an infrastructure project, as each phase has different risk and return characteristics (Ehlers, 2014).

While the 3Ts concept is still useful for understanding financial flows in the European water sector, in 2015, the World Water Council (WWC) and OECD suggested an updated categorisation to reflect new financial sources and instruments made available for water infrastructure in recent years (see Table 6.3). The view that additional financial sources for investment is necessary is shared with important stakeholders, who emphasise that a balance must be struck to ensure financial stability.

<table>
<thead>
<tr>
<th>Sources of finance</th>
<th>Loan and bond finance</th>
<th>Equity finance</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tariffs and user charges</td>
<td>Public development banks</td>
<td>Institutional investors</td>
</tr>
<tr>
<td>Taxes (national budgets)</td>
<td>Commercial banks (incl. project finance)</td>
<td>Sovereign Wealth Funds</td>
</tr>
<tr>
<td>ODA</td>
<td>Institutional investors</td>
<td>Specialised water funds</td>
</tr>
<tr>
<td>Philanthropic funds</td>
<td>Sovereign Wealth Funds</td>
<td>International Financial Institutions</td>
</tr>
<tr>
<td>Property taxes and other levies and contributions</td>
<td>Public bond issue</td>
<td>Private equity funds</td>
</tr>
<tr>
<td>Self-finance by users</td>
<td>International Financial Institutions</td>
<td>Venture capital</td>
</tr>
<tr>
<td></td>
<td>Project bonds</td>
<td>Public-Private Partnerships</td>
</tr>
<tr>
<td></td>
<td>Microfinance</td>
<td>Individual shareholders</td>
</tr>
<tr>
<td></td>
<td>Climate finance</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Export credits</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Institutional bondholders</td>
<td></td>
</tr>
</tbody>
</table>

Source: Developed from World Water Council and OECD, 2015

Note(s): This table represents the global water sector. Some financial sources might not be relevant to Europe.
6.2.2 Sources of finance available in practice

Water sector infrastructure investment in the EU has primarily been financed by national governments and local authorities through public budgets. It has traditionally been common for large European cities to finance urban water services by issuing municipal bonds as debt security (World Water Council and OECD, 2015). There are also various schemes set up by individual EU Member States. The Netherlands has for example established a public bank – the Netherlands Water Bank (NWB Bank) – which arranges short- and long-term loans for water authorities, municipalities, provinces, social housing, healthcare, educational institutions, public-private partnerships (“PPP”) and activities in the field of water supply and the environment. In 2016, NWB Bank issued new long-term loans totalling over €7bn (NWB Bank, 2017).

EU funding sources and instruments

The EU water sector is eligible for European cohesion policy funding from the European Regional Development Fund (ERDF) and, depending on region, the Cohesion Fund (European Commission, 2015). In the 2007-2013 programming period, around €17.8bn was allocated from the Cohesion Policy funds to 22 Member States to carry out investment to meet the provisions of the Council Directive 91/271/EEC concerning urban waste water treatment (COM(2016) 105 final). So far in the 2014-2020 period, funding via all the European Structural and Investment Funds has contributed to improved water supply for over 12 million people (including planned investment) and improved wastewater treatment for almost 17 million people (including planned investment) (European Commission, 2017).

European Investment Bank

The EIB has been the largest source of loan finance to the water sector to date – both in Europe and globally – compared with other international financial institutions, providing €64bn for 1,400 water sector projects worldwide (European Investment Bank, 2017, n.d.). This equates to an average of €4bn annually (T. van Gilst, EIB).

The EIB lends to public and private utility companies, national and local authorities, or directly to individual projects. It offers project loans, intermediated loans, Natural Capital Financing Facility (NCFF) and Joint Assistance to Support Projects in European Regions (JASPERS). It lends on average 30% of project investment costs, but can lend up to 50%. Lending focuses on the “modernisation and extension of existing distribution, collection and treatment networks as part of large-scale national/ regional/municipal capital expenditure programmes”. Investments have been made in all Member States.

Between 2001 and 2007 the EIB lent €9.1bn to the water sector in 19 of the EU-27 countries. Over this period, 37% of the loans went to the UK and Germany (EIB, 2008). In the period 2003-2007 the EIB increased its lending to the water sector to an annual lending of around €2.1 bn. Tightening of water legislation has been a major drive of investment in the EU water sector (EIB, 2008). Between 2008 and 2012, the EIB lent about €17bn to water-related projects, including irrigation and sewerage, 89% of which was for schemes in EU Member States (European Investment Bank, 2013). In 2016, the EIB lent €37m to EU water and wastewater projects related to climate action (European Investment Bank, 2017). In the period 2016 to 2017 the EIB lent a total of €4.4bn to EU member states for projects relevant to water and waste water management (EIB, 2017).

European Fund for Strategic Investments

In June 2015, EIB and the European Commission launched the European Fund for Strategic Investments (EFSI), intended to mobilise private financing for strategic investment. EFSI has capital of €33.5bn (€26.0bn from the European Commission and €7.5bn of EIB’s own capital) and a ring-fenced budget for infrastructure and innovation, including water infrastructure (Mestres Domènech, 2017). The target is to mobilise a total of €315bn over three years, and in 2016 alone, EFSI-related total investment reached €163.9bn. Since the 2015 launch to the end of 2016, EFSI investment contributed to the construction or upgrade of almost 120km of water mains or distribution pipes with over 2 million people benefiting from safe drinking water (European Investment Bank, 2017).

At the EU level, for the period 2014-2020, the ESIF has planned investment in water supply, wastewater treatment, and flood protection benefiting 12.4, 16.9, and 13.2 million people respectively. By the end of
this programming period, the EU will have reduced the European population that is currently without access to public water supply by 56% (Investing in Jobs and Growth). The financing of these planned projects will be realized by the ERDF and the CF, with different levels of participation in each case. The ERDF will provide 36% of the funds needed to improve water supply, 51% of the investment in water treatment and 57% in the flood protection schemes, while the remaining capital needs will be covered by the Cohesion Fund.

EU-wide financial programmes

Water sector investment in the EU is supported by a number of programmes which provide funding, such as LIFE and the Danube Transnational Programme; or those which facilitate networking and advice in relation to accessing funding for water-related investment, such as ACQUEAU Open Calls, INNEON and RIS3 Regions (EIP Water, n.d.).

Other International Financial Institutions and development banks

Some EU countries are eligible for borrowing through the World Bank. In 2016, the World Bank lent almost €370m for Water, Sanitation and Flood Protection in Europe and Central Asia (World Bank, 2016). The improvement of water and wastewater systems is also a mandate of the European Bank for Reconstruction and Development (EBRD). By the end of 2013, EBRD had financed 153 water and wastewater projects to a value of €2.18bn, focusing on the efficient provision of drinking water in larger towns and cities in developed countries (European Bank for Reconstruction and Development, 2014). Further, multi-lateral development banks, such as the Council of Europe Development Bank (CEB) offers finance to infrastructure projects in the areas of “urban renewal and rural modernisation”, where these are part of national, regional or municipal budgets (CEB, 2017). The CEB have financed a number of water relevant projects in EU Member States including:

- Large-scale irrigation projects in Spain;
- Wastewater treatment and irrigation in Cyprus

Over the period 1957-2016, the total volume of projects approved in this sector was €9bn, representing 15% of all loans approved.

Private financial sources and instruments

Private repayable financial instruments to fund water industry investment include debt (various loans, including bonds and export credits), with fixed (and often interest) payments to the provider, and to a lesser extent equity (the buying and selling of stock). One example of the former is green bonds, where water accounts for about 10% of the €55bn raised globally. “Sustainable water management (including clean and/or drinking water)” is one of the categories of projects deemed eligible according to the Green Bond Principles (GBPs) (Boccaletti, 2015). As one example from Europe, the NWB Bank launched a 5-year, €500m green bond in 2014 to lend to Dutch water authorities (World Water Council and OECD, 2015), and in August 2017, British Anglian Water issued a utility sector green bond of over €270m (WaterBriefing, 2017).

There is an increasing interest from non-traditional private financial sources to invest in European water infrastructure, including, for instance, various kinds of institutional investors such as pension funds, insurance companies, Sovereign Wealth Funds, specialised water funds and new international development banks (Linklaters LLP, 2014; World Water Council and OECD, 2015). Institutional investors often invest long-term and have large pools of capital (Collins, 2017). For example, the Thames Tideway Tunnel in London, UK, is partly funded by UK pension funds (Mooney, 2016). The UK defence group BAE Systems and the Swedish national pension fund AP1 have recently invested in a €250m water fund that will focus on industrial water projects in Europe and Asia (Flood, 2016).

Specialised water funds are a type of private equity funds, with the Swiss investment manager Pictet’s water fund being the world’s largest at €2.4bn. Between 2010 and 2011, global assets of funds focused on water and specialist water funds nearly doubled to just over €20bn in 2011 (de Sa’Pinto and Menon, 2012).

Further, with the significant and increasing need for new and updated water supply and waste water treatment technology, venture capitalists have recently turned their attention to the water industry (Gies, 2012). While venture capital is still significantly lower in Europe than, for instance, in North America
(European Investment Bank, 2016), the venture capital stakeholder that provided input to this report emphasised that the current inefficiencies create opportunities for both investment and innovation within the EU.

6.3 Future policy priorities

6.3.1 Factors that are holding back investment

There are several factors and sector-specific characteristics that contribute to the underinvestment in the water sector. These include the long-term nature of investment projects, poor management of existing stock, emerging challenges linked to climate change, the risk of low return on investment, and complex site specific and legislative requirements. Based on the literature reviewed, an analysis of some of the drivers of these challenges is outlined below:

Water infrastructure investments represent sunk costs

As water infrastructure is a long-term asset, its performance is dependent upon physical factors (such as geography, topography and climate) as well as socio-economic drivers (investment, maintenance, quality of execution). Generally, because investment in the water infrastructure tends to be based upon returns over a long period of time, existing infrastructure has the characteristic of a “sunk cost”. Where infrastructure is already in place, it is likely that policies will focus on maintenance, repair and upgrades. Where new infrastructure is developed there is likely to be a greater emphasis on cost recovery (EEA, 2013; EEA, 2014; ACTeon et al 2015).

Water infrastructure in Europe is aging

Like other sectors in Europe, water infrastructure for supply and treatment is aging. Consequently, investment to cover maintenance will become insufficient in some areas (EIB, 2016). The EEA also acknowledges the importance of investment in Europe’s aging infrastructure, including investment in new technologies, training of staff and public awareness raising (EEA, 2014).

In Eastern Europe, the rapid rate of urbanisation has resulted in pressures on fresh water supply in some places (CEB, 2017). In the EU-13 (the newest Member States, in central & eastern Europe) it is anticipated that there is a need for significant infrastructure investment. Capital expenditure is necessary to comply with legislation, and to cover a backlog of low investment in water infrastructure in these countries (EIB, 2008).

Flooding and droughts

Specific challenges to water are also the increased risk of water scarcity and flooding (EIB, 2016). There is a need for investment to reduce vulnerability to floods and droughts and to support natural water retention measures (e.g. green infrastructures and green Common Agricultural Policy) (EEA, 2014). Floods continue to be the largest course of GDP losses from natural disaster, costing €150 over 2002-13. The economic cost of droughts was €86bn across Europe over the period 1980-2010 (EEA, 2010).

Riskiness of investment

A common misconception within the water sector is that infrastructure, upkeep and maintenance costs are covered by the normal revenues of water utilities. This is often not the case, and shortfalls on repairs and maintenance lead to a need for higher investment (EIB, 2016). Due to lack of clarity on the economics of water resource planning, the sector has seen decades of underinvestment (OECD-WWC-Netherlands, 2017; OECD, 2011).

Competitiveness is largely determined by the risk-return profile of a project, which is influenced by expected revenue streams and the underlying risk associated with the investment (OECD-WWC-Netherlands, 2017). To provide an adequate risk-return profile, institutions need to identify the drivers of water related risks, such as policies of urban development, measure and monitor water sector costs, and identify the benefits of water sector investment and communicate where they occur (OECD, 2016). Due to limited data, estimation of benefits is complex, and such benefits are often hard to monetise. But several
stakeholders encourage extensive cost-benefit analysis as a tool for investors to evaluate projects (OECD-WWC-Netherlands, 2017).

Prioritising member state investment needs
In general, national governments agree on the importance of water sector investment, but their spending performance does not support this. The sector is given a disproportionally small share of the public budget, and no current policy is attracting private investor to pick up the slack (OECD-WWC-Netherlands, 2017; Winpenny, 2003). Based on the evidence presented in the previous section, there are several sector-specific characteristics to consider for successful policy formulation. Water sector investment projects are competing with other infrastructure projects for financier’s attention.

Infrastructure needs – metering and leakages
The EEA note that leakages are a common infrastructure issue throughout Europe. As well as water losses, leakages also risk reductions in quality as pressure in distribution becomes lower (ACTeon et al, 2015).

A further infrastructural challenge for developing Europe’s water sector is the lack of metering infrastructure. This means that there is a lack of incentive to use water efficiently (ACTeon et al, 2015).

Heterogeneous legislative implementation
Water is a regulated sector in the EU, with the Water Framework Directive as the key regulatory instrument (EIB, 2013). The regulation does not require a specific sector organisation, and Member States have different opinions on how best to organise it. The difficulties of determining an optimal sector structure depend on a set of characteristics, listed below (Gee, 2004):

- Water distribution and wastewater collection, i.e. local transport of water to the final consumer and local collection of wastewater, are normally natural monopolies;
- Entry is associated with large sunk costs; fixed costs linked to water distribution represent up to 70% of the supply costs (Gee, 2004);
- Water is difficult and expensive to transport; transport costs per 100km represents approximately 50% of the water wholesale cost (compared to 5% for electricity);
- Water infrastructure is typically unusable for any other purpose and cannot be removed;
- There is a lack of transparency in the sector;
- National, regional and local authorities have traditionally imposed public service obligations on water sector operators due to requirements of health and environment, which is associated with exclusive rights.

The characteristics also explains why liberalisation of the water sector has not brought the same benefits of private sector investment as liberalisation of other network industries.

Stakeholder views of barriers to investment
Three principle barriers emerged from the stakeholder views:

- Regional differences – there are substantial regional disparities across the EU in terms of investment. While in some countries, utilities and institutions have adequate financial capacities, some Member States rely exclusively on EU funds, which disrupts the balance of investment sources through a strong dependence on transfers. In addition, justifying how much money the sector in a Member State needs is quite a lengthy exercise (T. van Gilst, EIB).
- Inefficient pricing – there is a big variation across Member States in how water tariffs are set up. In some cases, tariffs are not a suitable device for revenue raising, as they increase the price and reduce the affordability of water services. However, public authorities and utilities are almost wholly reliant on tariffs as a revenue stream, and a systemic revenue shortfall can also negatively impact their ability to borrow from banks (e.g. the EIB) due to a perceived reliance on transfers. Inefficient pricing remains a considerable constraint to investment and in the long-term leads to systematic underinvestment.
- Political considerations – the water sector is strongly politicised in a number of countries, where strong political and lobbying influences exist, and local utilities can operate as de-facto monopolies.
Political interests in the water sector are a major barrier for reform, and serve as a barrier to investment as they add considerable uncertainty in the market.

Some additional barriers that have been discussed are:

- **A slow-moving market** – although the water sector is a large market, is rather slow and very conservative in regard to governance, which presents a considerable barrier, particularly for transformative and innovative technological investment (A. Figeac, INNEON).

- **Administrative barriers** - for newer Member States such as Bulgaria, which requires large capital investment to construct infrastructure to ensure compliance with the water acquis, there is a lack of administrative capacity for handling the often large and complex tendering procedures that characterise the water sector (D. Simidchiev, Hydrolia).

### 6.3.2 Policy options

The following potential policy options have been explored:

- **Extending blended finance** - governments may provide risk mitigation to long-term investment projects, where it would result in more appropriate allocation of risks and their associated returns. It may be beneficial for the EU to develop a typology of water sector infrastructure projects based upon their risk and return attributes, to determine the bankability of projects. Economic analysis of the allocation of public finance, including concessional finance, is necessary to improve the efficiency of public spending, and an evaluation is required of where concessional finance might be crowding out private finance instead of catalysing. With analysis that goes beyond cost-benefit analysis of stand-alone projects, the EU would design investment pathways that maximise water security over the long term (OECD-WWC-Netherlands, 2017).

- **Offering standardised green bonds** - the use of green bonds as an investment vehicle has grown rapidly over recent years, and is a viable option for expanding investment in the EU. So far, green bonds have generally been steered toward low carbon investment projects (OECD-WWC-Netherlands, 2017). In the US a standard for water climate bonds, specific to the water sector, has been developed. The standard is intended to provide investors with verifiable science-based criteria for evaluating water-related bonds (OECD, 2016). The EU could develop a similar standard.

- **Consider how to encourage downstream measures in water-dependent industries** - sectors at risk of water scarcity include agriculture and power generation. Firms have incentives to engage in investment that reduce water risks for their own production, but also to reach higher environmental, social, and governance (ESG) factor scores (OECD-WWC-Netherlands, 2017). For large scale implementation, it is important to ensure synergies and complementarities with investment in other sectors (EIB, 2016). A policy to foster sustainable agricultural practises has the potential to bring substantial benefits to the sector in terms of avoided costs of expanding or improving water treatment plants. Such schemes may have modest funding requirements, but can be difficult to fund under a constrained regulatory framework, where such payments may be seen as a transfer of public funds to private farmers (OECD-WWC-Netherlands, 2017). The nexus between water and energy is another area to explore for policy makers, where industry may be interested in investing in more efficient infrastructure, technology, or procedures in the right policy environment (2030 Water Resources Group, 2009).

- **Investment in water tech** - discussions of sector finance tend to focus on upgrading infrastructure, but other areas of resource management, including data collection, weather forecasting, afforestation, land use regulation, conjunctive use of surface and ground water, conservation measures, ecosystem management and pollution control are generally less risky and equally important to emphasise in policy forums (EIB, 2016). Furthermore, EU countries’ urban design increasingly relies on computer tools, inspection robots, and geographical information systems to gain precise knowledge of the state and performance of assets. This information allows better planning of investment in maintenance, and is an area where innovation could attract venture capital if accurately communicated (OECD, 2016).
As was the case with motor vehicles, the modelling tools available in this project are better able to model the socioeconomic impacts of the successful impact of these policies, rather than the marginal impact of a single policy. The quantitative analysis of policy therefore focussed on measuring the impact of bridging the water investment gap; namely, additional annual investment of 2016EUR 58bn\(^{52}\), compared to recent historical expenditure, over the period 2019 to 2025.

The principal effect in the short term is a significant investment stimulus across the Union. Investment in 2025 is 2018EUR 77.7bn higher than baseline. The investment stimulus results in a positive GDP difference of 0.48% to baseline, and 0.45m increase in EU wide employment in 2025. The increase in economic activity from 2019 to 2025 is greater than the investment stimulus itself, given indirect and induced effects of the water sector investment. The impact of these multiplier effects is shown in Table 3: both wider investment in the economy and consumer expenditure are greater than baseline values. There is some leakage of the effect, shown in negative changes to net exports.

**After the investment period, the cost of the investment reduces growth in economic activity, compared to the baseline.** Increased direct taxation in the scenario reduces personal disposable income, reducing consumer expenditure, see Table 6.4. Following full repayment of the debt, EU wide economic growth recovers. From 2046 onwards, EU GDP is at a level very similar to baseline.

Given that EU-level resources are focused on lower-income Member States, the water investment has redistributive effects, reducing economic disparities. The lower-income Member States benefit from significant investment stimuli, but pay only part of the subsequent cost.

### Table 6.4 Water investment scenario: EU28 disaggregate GDP results differences to baseline (2018EUR bn)

<table>
<thead>
<tr>
<th>Year</th>
<th>GDP</th>
<th>Investment</th>
<th>Consumption Expenditure</th>
<th>Government Expenditure</th>
<th>Net Exports</th>
</tr>
</thead>
<tbody>
<tr>
<td>2020</td>
<td>87.5</td>
<td>82.0</td>
<td>15.9</td>
<td>0.0</td>
<td>-10.4</td>
</tr>
<tr>
<td>2025</td>
<td>81.8</td>
<td>77.7</td>
<td>14.6</td>
<td>0.0</td>
<td>-10.5</td>
</tr>
<tr>
<td>2030</td>
<td>-28.1</td>
<td>-9.7</td>
<td>-20.4</td>
<td>0.0</td>
<td>2.0</td>
</tr>
<tr>
<td>2035</td>
<td>-25.5</td>
<td>-3.9</td>
<td>-22.9</td>
<td>0.0</td>
<td>1.3</td>
</tr>
<tr>
<td>2040</td>
<td>-19.0</td>
<td>-2.9</td>
<td>-17.7</td>
<td>0.0</td>
<td>1.6</td>
</tr>
<tr>
<td>2045</td>
<td>-26.4</td>
<td>-7.6</td>
<td>-17.3</td>
<td>0.0</td>
<td>-1.5</td>
</tr>
<tr>
<td>2050</td>
<td>-7.0</td>
<td>1.3</td>
<td>-7.5</td>
<td>0.0</td>
<td>-0.8</td>
</tr>
</tbody>
</table>

### 6.4 Conclusions

Water treatment & supply has been identified in this study as a sector of interest, on the basis that it is an environmental services sector, with a strong influence on other sectors and strong links to the SDGs. When considering the environmental impact of water treatment & supply, it is perhaps not helpful to think of it as a conventional economic sector; since its whole aim is the abstraction, supply and recirculation of a scarce resource. The sector therefore has a significant environmental impact, and there are substantial opportunities for environmental policy to alter this impact, including in reducing abstraction levels through increased reuse, and management of the environmental impacts where abstraction does take place.

There is substantial evidence, as presented in this chapter, that planned investment in the water supply sector is insufficient to meet the requirements of the water acquis. A gap of around 58bn EUR has been identified; and there is a role for policy to help to bridge this. Policy options focus on encouraging investment from the private sector, which has substantial implications for the macroeconomic impacts of delivering this. CE’s E3ME model was used to assess the benefits and costs of increased investment in water infrastructure. The principal effect in the short term is a significant investment stimulus across the EU. The

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\(^{52}\) See Table 6.1. 2016EUR 58bn is the sum of the annual investment gap in ‘water security (including flood risk management)’ and ‘compliance and rehabilitation of Europe’s water infrastructure’.
investment will improve water security, avoiding disruptions to industry and to power generation. Investment will increase resilience to flooding, reducing economic and human costs of natural disasters, while the provision of water improves human health, increases productivity and facilitates levels and types of socioeconomic activity that would not be possible without such a supply.

If Europe can develop a competitive industry in water technology, improving the efficiency of water supply operations and ensuring the efficient use of water, then this can present an export opportunity to improve the trade balance of Europe vis-à-vis the rest of the world.
7 Sewerage & waste management

The sewerage & waste management sector is an environmental services sector with a heavy dependence on ecosystem services. The sector has strong links to other sectors, which are dependent on water as a resource. There are significant environmental policy challenges relating to waste and emissions.

Circular economy policy opportunities exist in this sector in the recovery of materials and biomass; again, using waste as a resource. The recovery of material is related to the three manufacturing sectors under analysis in the project, given that all produce waste in production processes and in the consumption of their products.

1. Waste collection:
   1. Extend collection of organic waste from municipal solid waste stream to contribute to maximising recovery of biomass
   2. Extend capture of organic waste from waste water treatment plants to contribute to maximising recovery of biomass
   3. Adapt waste water treatment plants to capture (more) microplastics to reduce this type of marine litter

2. Regulation:
   4. Increase recycling targets to drive separate collection to aid material recovery
   5. Introduce additional bans on landfilling and incinerating certain types of waste
   6. Improve minimum requirements for certain waste streams to facilitate recycling and reuse

3. Processing:
   a. Increase use of optical sorting machines to optimise sorting processes
   b. Develop measures to support the interface between chemical, product and waste legislation (i.e. innovative tracing technologies, to improve information flows)

7.1 The current environmental impact of the sector

7.1.1 Overview

Waste management

Each year, 2.7 billion tonnes of waste are produced in the EU, of which 98 million tonnes (4%) are hazardous. In 2011, per capita municipal waste generation averaged 503kg throughout the EU, but ranges between 298 and 718kg across individual Member States.

The collection and transportation of waste from the point of generation to the point of reuse or treatment may generate significant greenhouse gas and NOx emissions, as well as resulting in significant fossil resource depletion and traffic/congestion (EC, 2016a).

On average, only 40% of solid waste is prepared for re-use or recycled, although some individual Member States achieve a rate of 70%, demonstrating how waste could be used as one of the EU’s key resources. However, many Member States still landfill over 75% of their municipal waste (EC, 2013).
Figure 7.1 EU municipal waste management

Sewerage

Sewage, or municipal wastewater, contains nitrogen and phosphorus from human waste, food and soaps and detergents. On a global level, nutrients recovered from these waste streams could contribute approximately 2.7 times the nutrients currently contained in chemical fertilizers. Increasing urbanization trends are leading to growing nutrient concentration in solid waste and sewerage as sewage sludge (EMF, 2017).

Sewage sludge, the by-product of waste water treatment processes, is mainly disposed of through incineration, landfilling and application to land. Sewage sludge is composed of six categories of components, some of which can lead to environmental impacts at the final disposal stage: non-toxic organic carbon compounds; nitrogen- and phosphorous-containing components; toxic inorganic and organic pollutants (e.g. dioxins, pesticides, polychlorinated biphenyls etc.); pathogens and other microbial pollutants; inorganic compounds (e.g. silicates, aluminates, calcium- and magnesium-containing compounds); water (Sathir et al., 2017).

In 2014, 8.7 million tonnes of dry solid matter of sludge were produced in the EU, representing approximately 17 kg per inhabitant (EC, 2017). Of the total sewage sludge produced, 58% was reused, mostly in agriculture. Some countries showed ratios below 10kg, suggesting low levels of collection and treatment – Italy, Cyprus, Portugal, Bulgaria and Romania.

In the EU, 70% of the phosphorous concentrated in sewage sludge and solid waste is not recovered, suggesting untapped potential for the recovery of nutrients which could be looped back into the soils rather than being discharged. A way to recover nutrients from wastewater is to produce concentrated NPK fertilisers. Estimates indicate that capturing all nutrients from excreted waste in household sewage could lead to the recovery of 30 million tonnes of nitrogen, 5 million tonnes of phosphorous and 12 million tonnes of potassium – a third of global fertilizer demand (EMF, 2017).

7.1.2 Recycling, reuse and waste pathways

Recycling

Environmental impacts associated with the reprocessing operation and transportation of materials to the reprocessing facility. GHG emissions come from the carbon dioxide associated with electricity consumption for the operation of material recovery and sorting facilities (see e.g. EC, 2016a). The key environmental advantages of recycling and recovery are reduced quantities of virgin material use and disposed waste, and
the return of materials to the economy. For example, it takes 5 tonnes of bauxite ore and 32 barrels of oil to make a tonne of aluminium (Upstream, 2015). Recycled aluminium generates energy and air pollution impacts 75-90% lower than virgin aluminium whilst also avoiding most of the resource depletion associated with aluminium ore extraction. Production of recycled glass uses around 20-30% less energy than virgin glass (EC, 2016a). Recycling of high-quality paper products e.g. office paper can result in significant environmental benefits (WRAP, 2010), and recycling rates of over 80% are not uncommon in EU Member States with EPR schemes for graphic paper (e.g. in FI, NL and SE) (Bio by Deloitte, 2014). Recycling and incineration of paper with the latest energy recovery efficiencies are broadly comparable in terms of climate change impacts (WRAP, 2010), but the use of recycled rather than virgin paper pulp can help to prevent land use change and indirect emissions from the loss of carbon stored in forests (James, 2013). It takes one gallon of used industrial oil to produce 0.25 gallons of new high-quality lubricating oil (as opposed to 42 gallons of crude oil required to make the same quantity, albeit alongside a range of other products) (US EPA, 2015); again recycling rates of over 80% are achievable if EPR schemes are in place (e.g. in BE, FI, IT, DE and PT) (Bio by Deloitte, 2014). For plastics, mechanical recycling is the best waste management option in respect of reducing the depletion of natural resources (WRAP, 2010). Recycled rubber from end-of-life tyres can be processed into rubber shred and crumb of various particulate sizes which can then be incorporated into a wide range of products (e.g. paving, roof tiling, mulch, vehicle parts, new tyres, road surfaces, carpet underlay, footwear etc.) (WRAP, no date), reducing the need for various raw materials.

Table 7.1 GHG emissions avoided per tonne of different types of waste avoided or recycled

<table>
<thead>
<tr>
<th></th>
<th>Glass</th>
<th>Board</th>
<th>Wrapping paper</th>
<th>Dense plastic</th>
<th>Plastic film</th>
<th>Metals</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoided</td>
<td>Kg CO₂e/t</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>920</td>
<td>1,600</td>
<td>1,510</td>
<td>3,320</td>
<td>2,630</td>
<td>12,000</td>
</tr>
<tr>
<td>Recycled</td>
<td>390</td>
<td>1,080</td>
<td>990</td>
<td>1,200</td>
<td>1,080</td>
<td>3,300</td>
</tr>
</tbody>
</table>


**Reuse**

Re-using products may result in an overall offset in environmental effects due to a lower requirement for manufacture of replacement products, thereby reducing raw material inputs, water and energy use associated with manufacture, and the transportation of new products to the point of sale. Design for reparability can facilitate product reuse. Reuse can potentially save almost 100% of materials, energy and water, whilst remanufacturing can save 85% and 80% for materials and energy respectively used in manufacturing (Steinhilper, 2006). However, product re-use is also associated with environmental impacts arising from transport and collection, as well as product cleaning operations. Some disassembly operations may require significant electricity demand, and such operations must also be carefully controlled to minimise leakage of hazardous substances such as refrigerants, oils or PCBs (EC, 2016a).

**Landfill**

The environmental impact of landfilling depends to a large extent on the design and management of the landfill and the type of waste landfilled; well-lined and capped landfills, with landfill gas captured and used for energy generation, or landfills for inert waste, typically have the lowest environmental impacts (EC, 2016a). Approximately 120 m³ of biogas (60% of which is methane, with a global warming potential 25 times that of CO₂) is produced per tonne of MSW (fresh weight) landfilled (Obersteiner et al., 2007), and emissions can be anywhere in the range of 158 to 1,285 kg CO₂e per tonne of MSW deposited, depending on the landfill’s design (EC, 2016a). If MSW is mechanically and biologically treated (MBT) before landfilling, gas production can be reduced by approximately 95 % (JRC, 2006). Whilst landfill leachate is typically collected, removed and treated, in lower-standard landfills it may leak into soils or water courses, causing contamination.

**Incineration**

Incineration, the combustion of waste at high temperatures for a sustained period, leads to a very substantial reduction in the volume of waste and effectively destroys pathogenic biological organisms. By-
products of the combustion process principally comprise emissions to the atmosphere\textsuperscript{53} (which can, in high-
standard incinerators, be largely captured e.g. by catalytic processes) and residual ash, which can account
for something in the region of 20-30 % of the weight, and 10 % of the volume, of inputted MSW. Many
incinerators are now classed as waste-to-energy facilities, since waste combustion can be used to generate
heat and energy. The energy recovery efficiency of incineration plants can vary considerably, from
anywhere between 14-90% thermal efficiency (with the higher rates achievable e.g. where generated heat
is used directly for district heating) (EC, 2016a). It should however be noted that with regards to the waste
hierarchy, waste prevention, reuse and material recycling are all deemed preferable to energy recovery.

Composting/anaerobic digestion

Like landfilling, composting of organic waste generates CH\textsubscript{4}, \textsubscript{NH}3 and N\textsubscript{2}O emissions and nutrient leaching,
although these may be partially compensated by the fertiliser replacement and soil improver (humus)
properties of compost (EC, 2016). Anaerobic digestion (AD) can be an efficient option to recycle nutrients
and recover energy from organic wastes, but overall environmental impacts also depend on emission rates
of CH\textsubscript{4} and NH\textsubscript{3} and methods to store and apply digestate. Larger, centralized AD plants can be more
efficient, whilst emissions may be high from small plants (EC, 2016a).

7.1.3 Sewerage pathways

Recycling sewage sludge for agriculture

The practice of applying sewage sludge to agricultural land is common in Europe, as a way to utilize
nutrients such as phosphorous and nitrogen and organic substances for soil improvement (Milieu et al). In
addition, this would reduce the use of synthetic fertilizers which bring in nutrients from non-renewable
sources (EMF, 2017). Despite the benefits that such practice can deliver for agriculture, the application of
sewage sludge as a fertilizer on land can have negative implications for the environment: decrease in soil
value, ecosystems degradation and decrease in groundwater quality. This is due to the volatilization of
pollutants to air, the release of pollutants to surface water and to soil (Milieu et al, 2010).

In addition to the emission of pollutants, the use of sewage sludge as fertilizer for land represents an
important source of primary microplastic pollution in soil. Sewage sludge is generally contaminated with
micro fibers and microplastics, as these are not filtered out from waste water treatment plants. The
application of sewage sludge to land is common in Europe and North America – approximately 50% of
sewage sludge processed for agricultural use. Annual additions of microplastics to land in Europe are
estimated between 125 and 850 tonnes per million inhabitants (Nizzetto et al., 2017).

The reuse of wastewater is considered a reliable source of water supply, due to its independence from
seasonal variabilities and to the over-abstraction of water as a major water stress. Wastewater reuse can
bring a number of economic, environmental and social benefits, for instance in its application to farming
(e.g. irrigation).

Wastewater reuse for irrigation

In addition to providing a reliable water resource, the reuse of wastewater reduces wastewater disposal,
contributing to the preservation of water quality downstream. Thanks to the nutrients contained in
wastewater, reusing it for irrigation reduces the need for fertilizers. However, irrigation through
wastewater requires appropriate treatment. Potential environmental impacts include: groundwater
contamination due to heavy metals, nitrate and organic matter; soil contamination due to salt accumulation
and acidifications; crop contamination due to potential toxic substances in wastewater. The reuse of
wastewater differs significantly across Europe as the volume of water for irrigation depends on climate,
crop type, irrigation method and on the area to be irrigated (EC, 2013). Currently, annual reuse of treated
urban wastewater is approximately 1 billion cubic meters, well below the EU potential, estimated at 6
billion cubic meters. Initiatives for the reuse of wastewater for irrigation are present in southern Member States such as Spain, Italy, Greece, Malta and Cyprus, as well as in some northern Member states such as Germany and Belgium (EC, 2018b).

Wastewater treatment follows a number of steps which require energy: solid waste removal, biological digestion, disinfection and discharge. Energy recovery from sewage can offset the energy required for its treatment. It is estimated that the energy embedded in wastewater is 14 times higher than what is needed for treatment. The European Commission estimated that if all organic waste was turned into energy, 2% of the EU’s renewable energy target could be met (EMF, 2017).

Sewage sludge to energy
The growing amount of sewage sludge produced has led to the introduction of technologies to produce energy from sewage sludge. Dried sewage sludge is an attractive source of energy. Anaerobic digestion (AD) is one of the most common methods of sludge treatment and energy recovery. AD generates biogas which can be used as fuel.

In 2013, Thames Water saved £15 million on its energy bills through the generation of 14% of energy demand from sewage sludge (EMF, 2017).

7.1.4 Socioeconomic footprint of the sector
According to the NACE rev. 2 statistical classification of economic activities, waste management includes waste collection, treatment and disposal activities as well as materials recovery. Overall, the waste management sector has witnessed substantial growth. Employment increased from 0.8 million (full-time equivalents) in 2000, to 1.1 million in 2014 (Weghmann, 2017; Eurostat54). Among the activities in the sector, waste collection is associated with the highest labour intensity and, together with treatment activities, employment in these stages of waste management is dominated by bigger companies. On the contrary, material recovery remains in the hands of smaller companies (EC, 2016a).

Table 7.2 EU employment in waste collection, treatment and disposal activities, materials recovery and sewerage (Number of persons employed)

<table>
<thead>
<tr>
<th>Year</th>
<th>Waste collection, treatment, disposal activities and material recovery</th>
<th>Waste collection</th>
<th>Waste treatment and disposal</th>
<th>Materials recovery</th>
<th>Sewerage</th>
</tr>
</thead>
<tbody>
<tr>
<td>EU27</td>
<td>2005</td>
<td>729,600</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>760,000</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2007</td>
<td>755,228</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2008</td>
<td>734,300</td>
<td>417,000</td>
<td>152,700</td>
<td>164,600</td>
</tr>
<tr>
<td></td>
<td>2009</td>
<td>733,800</td>
<td>410,800</td>
<td>163,300</td>
<td>159,700</td>
</tr>
<tr>
<td></td>
<td>2010</td>
<td>795,100</td>
<td>438,800</td>
<td>181,500</td>
<td>174,800</td>
</tr>
<tr>
<td>EU28</td>
<td>2011</td>
<td>838,900</td>
<td>187,100</td>
<td>190,000</td>
<td>142,000</td>
</tr>
<tr>
<td></td>
<td>2012</td>
<td>851,500</td>
<td>468,700</td>
<td>190,000</td>
<td>149,300</td>
</tr>
<tr>
<td></td>
<td>2013</td>
<td>877,900</td>
<td>487,700</td>
<td>197,800</td>
<td>193,000</td>
</tr>
<tr>
<td></td>
<td>2014</td>
<td>914,320</td>
<td>515,953</td>
<td>205,565</td>
<td>192,802</td>
</tr>
<tr>
<td></td>
<td>2015</td>
<td>924,424</td>
<td>520,000</td>
<td>212,551</td>
<td>193,000</td>
</tr>
</tbody>
</table>

Source: Eurostat55

The employment potential of the waste management sector is expected to grow further in the future. In particular, a stronger focus on re-use and recycling triggered by the adoption of circular economy principles

55 http://ec.europa.eu/eurostat/web/products-datasets/-/sbs_sc_ind_r2
could bring a significant benefit in terms of additional employment, due to the labour-intensity of re-use and recycling activities compared to disposal (EC, 2016a). Current legislation and stricter targets for recycling also contribute to increasing this potential. It is estimated that by 2020 an additional 400,000 jobs could be created in the waste management industry as a result of full compliance with EU waste policy and an additional annual turnover of €42 billion (EC, 2016b).

As far as recycling is concerned, the enforcement of higher recycling rates in EU Member States is expected to lead to the creation of 50,000 new direct jobs throughout the process’ value chain by 2020. Direct jobs are found in the sorting and separation of materials, collection and recycling. These mainly, but not exclusively, require low-skilled workers, therefore presenting the potential of contributing to social inclusion and poverty alleviation. In addition, an estimated 75,000 new indirect jobs could be created in recycling activities linked to construction and maintenance of facilities, research and innovation, administration and management (PRE, 2016; EEA, 2011). These number are expected to increase further by 2025, reaching 80,000 and 120,000 new direct and indirect jobs respectively (PRE, 2016).

7.1.5 Environmental pollutant releases

The number of E-PRTR releases for the waste management and sewerage sector that has all required data available for the calculation of environmental impacts is around 25% across the period 2007-16.

Human health and environmental impacts follow each other exactly over this period, showing that changes in releases over time do not favour pollutants with a greater impact on one group than the other.

*Figure 7.2 The environmental impacts of pollutant releases from the water treatment and supply sector*

There is a substantial spike in environmental impacts in 2012, which can be attributed to a large release of CO2, and a much smaller release of carbon monoxide, in the UK in this year. CO2 emissions have risen

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56 The key characteristic that is often missing is the CAS pollutant code, which is required in order to assign a human health impact (measured in disability-adjusted life years) and ecosystem health impact (measured in species years lost).
steadily over the time period shown, although the environmental impact of this is mitigated somewhat by the decrease in releases of methane, down from 1.4m tonnes in 2007 to less than 0.9m tonnes in 2016.

7.2 Current direction of travel

Over the last 20 years, a large variety of policies have been introduced in the EU with the aim of increasing recycling. These include targets for the waste management of a variety of materials and products: electrical and electronic equipment, end of life vehicles, packaging, batteries, household waste, and construction and demolition waste (EEA, 2011).

More recently, in 2015, the European Commission adopted the Circular Economy Action Plan (COM/2015/0614 final), including a proposed revision of legislation on waste to support the transition towards a circular economy. These include a range of revised waste management targets put forward in the Waste Framework Directive (2008/98/EC), the Landfill Directive (1999/31/EC) and the Packaging and Packaging Waste Directive (94/62/EC), summarised in the table below.

Table 7.3 Proposed waste management targets

<table>
<thead>
<tr>
<th>Targets</th>
<th>Existing</th>
<th>2025</th>
<th>2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Share of municipal waste prepared for reuse and recycling</td>
<td>50% (2020)</td>
<td>60%*</td>
<td>65%*</td>
</tr>
<tr>
<td>Share of municipal waste landfilled</td>
<td>Reduce landfilled bio-waste to 35% of 1995 production levels (2016)</td>
<td>/</td>
<td>10%^</td>
</tr>
<tr>
<td>Share of packaging waste prepared for reuse and recycling</td>
<td>55-80% (2008)</td>
<td>65%</td>
<td>75%</td>
</tr>
<tr>
<td>Share of plastic packaging prepared for reuse and recycling</td>
<td>22.5% (2008)</td>
<td>55%</td>
<td>/</td>
</tr>
<tr>
<td>Share of wood packaging prepared for reuse and recycling</td>
<td>15% (2008)</td>
<td>60%</td>
<td>75%</td>
</tr>
<tr>
<td>Share of ferrous metal packaging waste prepared for reuse and recycling</td>
<td>50% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
<tr>
<td>Share of aluminum packaging waste prepared for reuse and recycling</td>
<td>50% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
<tr>
<td>Share of glass packaging waste prepared for reuse and recycling</td>
<td>60% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
<tr>
<td>Share of paper and cardboard packaging waste prepared for reuse and recycling</td>
<td>60% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
</tbody>
</table>

Source: European Commission, 2015

A relevant policy which has had implications on waste treatment and disposal is the ban on landfill adopted by several member States following the Landfill Directive. While in some Member States waste is still mostly landfilled, the ban has led to significant reductions in landfilled waste and consequently to increases in alternative options, such as recycling or recovery. As regards plastic waste, the ban has shifted waste treatment primarily towards incineration (EP, 2017). In addition, stricter environmental requirements on incineration plants and landfill sites have also contributed to increasing the competitiveness of recycling (EEA, 2011).

From the four legislative proposals on waste policy, a list of main measures which are relevant to the waste management sector, in addition to the aforementioned revised targets, can be identified (EP, 2016):

- Introducing an early warning system for monitoring compliance with targets

Setting minimum requirements for extended producer responsibility schemes and differentiating the contribution paid by producers on the basis of the costs necessary to treat their products at the end of their life

Promoting prevention (including for food waste) and reuse

Streamlining provisions on by-products and end-of-waste status (the stage at the end of the waste treatment process when materials are no longer considered waste, provided they meet certain conditions)

Aligning definitions, calculation methods for targets, reporting obligations and provisions on delegated and implementing acts.

Application of sewage sludge to land

Farmers’ usage of sewage sludge is regulated by the EU Directive (EU 86/278/EEC) on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. Limits are set on the concentration allowances for 7 heavy metals. Microplastic concentrations are not mentioned (EC, 2015, IEEP, 2018).

Current rates of wastewater reuse are well below EU potential, due to the environmental and human health risks associated with improperly treated water. With the aim of ensuring the safety of water reuse, the European Commission has recently proposed new measures to encourage the safe practice of water reuse for agricultural irrigation (EC, 2018a). These include: harmonized minimum requirements for the reuse of treated wastewater from urban wastewater treatment plants; risk management plans for additional water reuse related hazards; and increased transparency on water reuse practices across the EU. In addition, by improving information and awareness, the regulation can incentivise the adoption of water reuse practices and contribute to water stress alleviation. The introduction of harmonized standards for water reuse can prevent existing obstacles to the free movement of agricultural products which have been irrigated with reused water, increasing confidence in the practice (EC, 2018b). Existing standards are not harmonised at EU level and divergences across Member States can lead to insufficient levels of safety once the agricultural goods have reached the common market (Alcalde-Sanz and Gawlik, 2017). The implementation of the EU legal framework is estimated to increase water reuse in agricultural irrigation from 1.7 billion m$^3$ per year to 6.6 billion m$^3$. The proposed regulation builds on two previous EU legal instruments which already included water reuse as a practice to embrace the circular economy but lacked specified conditions - the Urban Wastewater Framework Directive and the Water Framework Directive (EC, 2018b).

Support for wastewater reuse is provided through funding from the Pillar 2 of the CAP, the EU’s rural development policy, designed to support the Union’s rural areas in overcoming environmental, economic and societal challenges. This is offered on a voluntary basis by Member States through their 7-year Rural Development Programmes based on a European ‘menu of measures’. For water reuse in irrigation, measure 4 ‘Physical investment (processing of farm products, infrastructure, improving the performance and sustainability of farms, etc.)’ is of particular relevance (EP58).

7.2.1 The impact of changing consumer preferences

The design of waste management services can affect their consumption. In particular, differences in payment systems, geographical location, collection strategies and public acceptability and engagement are among the main factors affecting the use and success of waste management services.

The implementation of Pay-as-you-throw systems can lead to a range of benefits in the waste management sector. Given proper infrastructure and collection, a system of this kind can increase the amount of waste being recycled and reduce residual waste (EC, 2016a).

The accessibility of waste collection centres, or the proper implementation of door-to-door collection, also influences the efficiency of waste management services and largely depends on the geographical location – densely populated urban areas or sparsely populated rural areas (EC, 2016a).

The transition to a circular economy requires substantial changes to happen at all stage of products value chain. On the consumer side, growing trends towards circular consumption approaches are leading to an increase in re-use and recycling. Changes in cultural norms have been achieved as a result of policy interventions, such as the integration of taxes/charges and deposit refund schemes. Changes of this kind have implications in the management of waste. While the impact of consumers’ behaviour strongly depends on the different materials and products, an increase in consumers’ awareness on the impacts of an improper management of waste and better knowledge of recycling processes can lead to greater engagement in waste sorting and recycling activities.

Consumers’ preferences represent a significant factor influencing the application of sewage sludge to land for agriculture. Perceptions of the general public towards the practice are generally negative but vary significantly across Member States and have evolved over time. The introduction of the Sewage Sludge Directive has gradually succeeded in encouraging the use of sewage sludge in agriculture and stricter limits have contributed to a higher level of acceptance of the practice. National legislation on sewage sludge have been strengthen in Member States over time, introducing stricter limits on the contaminants already regulated and adding requirements for others. Nevertheless, the sewage sludge applications remain limited in some Member States (Bio et al., 2014).

7.3 Future policy priorities

7.3.1 Links between the sector and the Sustainable Development Goals

The waste management sector has clear links to several of the Sustainable Development Goals and targets contained within them. Regarding Goal 6: clean water and sanitation, the sector has a role to play in helping to improve water quality by reducing pollution, eliminating dumping, minimising release of hazardous chemicals and materials, and substantially increasing recycling. Regarding Goal 11: sustainable cities & communities, the waste management sector is crucial to the target of reducing the adverse per capita environmental impact of cities, including addressing municipal and other waste management. Towards Goal 12: responsible consumption and production, the sector’s role includes contributing to the sustainable management and efficient use of natural resources, significantly reducing food waste, achieving the environmentally sound management of chemicals and wastes and significantly reducing their release to air, water and soil, and substantially reducing waste generation. Regarding Goal 14: life below water, the waste management sector will contribute to preventing and significantly reducing marine pollution from land-based activities, including marine litter. The sector is also crucially relevant to the 7th EAP regarding waste management to increase resource efficiency59 and regarding water management60.

7.3.2 Policy options

The following potential policy options have been explored:

- **Increased reuse** - proper waste sorting and collection systems support improvements in the proportion that is reused. Increasing reuse can reduce demand for manufacture, therefore reducing raw material inputs, water, energy and transportation, reducing environmental impacts. As the potential for product reuse is determined at the design stage, in addition to the availability of proper sorting and collection systems, several market-based instruments have been identified to support ‘design for reuse’. Better harmonisation of extended producer responsibility (EPR) schemes and the introduction of eco-modulation of fees based on reusability, reparability and recyclability criteria can further support the uptake of reuse and recycling, while increasing collection rates.

- **Increased use of wastewater** - the sectors offering the greatest potential to increase the use of wastewater and which have incentives to invest in water reuse infrastructure are those most at risk of water scarcity – power generation and agriculture (CDB, 2014).

- **Sewage-to-energy** - sewage sludge treatment represents a relatively minor fraction of total processed wastewater volume, yet its processing costs can account for up to 50% of wastewater

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59 39, 40, 43(d), & 43(viii) of the Annex of the 7th EAP.

60 See points 8, 17, 28(vii), 41, 43(ix) of the Annex of the 7th EAP.
treatment plant operating expenses (e.g. US EPA, 2008). What is more, older sewage sludge processing operations could also involve high environmental costs. As far as agricultural application is concerned, these are mainly associated with contents of heavy metals, organic compounds and pathogens as well as groundwater leaching of untreated sludge (Lamastra et al., 2018). Given that agricultural application of sewage sludge cannot handle the entire volume of EU production, use of sewage for energy production is of interest (Spinosa et al., 2011; Kacprzak et al., 2017). This option is all the more relevant in view of the EU’s renewable energy & climate goals.

- **Increased wastewater reuse for agriculture** - wastewater re-use is well below the EU potential and has large scope to be increased in many Member States. Where such water can be used as an alternative to freshwater, it has the double benefit of reducing investment costs for (e.g.) agricultural firms, who can access/process wastewater more cheaply than freshwater, while also limiting (or reducing) the environmental impact of the sector. Harmonised and stricter EU-wide standards for wastewater reuse can overcome the current divergences across Member States and further incentivise investments by both the agriculture sector and the water utilities. As the safety of the agricultural products irrigated with wastewater increase, so does the confidence in and the uptake of the practice (EC, 2018b).

### 7.4 Conclusions

The sewerage & waste management sector is an environmental services sector with a heavy dependence on ecosystem services. The sector has strong links to other sectors, which are dependent on water as a resource. There are significant environmental policy challenges relating to wastes and emissions, and well-targeted policy can in fact lead to a reduction in environmental impacts in other sectors (for example where waste can be used for energy generation, or by reducing waste and therefore encouraging greater resource utility). The clearest opportunities for reducing the environmental impact of the sector are from increased re-use and recycling of waste and wastewater; and while some measures require substantial investment (such as for improved waste sorting), others can be achieved at relatively low cost (such as standards for wastewater use).

Very similar analysis (from a macroeconomic modelling perspective) has been carried out in the sector study looking at water supply, investment in water supply and/or sewerage has a small positive economic impact. The investment itself acts as a stimulus, leading to a short-term boost to GDP and employment, including the creation of local jobs in the construction and installation of infrastructure. However, in the longer term, the payback of such investment leads to slightly lower output in the economy than would otherwise be the case.

There are some mild redistributive impacts of investment in water, assuming that it is distributed according to need (and sourced from central pooled funds, rather than funding being structured at the national level).

In addition, improving the state of Europe’s water supply, treatment and sewerage infrastructure has additional benefits beyond these. It boosts water security, reducing reliance upon rainwater and other fresh water sources, as well as improving resource efficiency and helping to reduce the environmental footprint of water supply (directly) and all sectors of the economy which use water as an input.
8 Conclusions

The analysis carried out in this study has highlighted the environmental impact of production across a subset of economic sectors. All studied sectors have some impact on the environment; either directly, through the activities carried out within that sector, e.g. fuel extraction, or indirectly through supply chains, e.g. through the consumption of electricity. Besides bilateral links, the production within these sectors and with the natural environment is also inter-related. This analysis has set out how the environmental impact of these sectors might be expected to increase over time, and it has examined the potential role for policy in shaping the evolution of the environmental footprint of these sectors.

The transition to a circular economy requires substantial changes to happen at all stage of products value chain. Changes in cultural norms have been achieved as a result of policy interventions, such as the integration of taxes/charges and deposit refund schemes. Further consumer behaviour change is key in such a transition, whether products with the greatest environmental impacts are substituted, bringing reduced energy consumption and resource use requirements. Consumers’ role lies in the individual choices of powertrains (electric or internal combustion engine), of plastic products (single use or multiple use or bioplastics), of food and beverages. The study concludes that current consumer-driven market changes to more eco-friendly consumption patterns generated by increasing consumers’ education, awareness and demand on reliable eco-labelling are expected to continue; and will continue to encourage producers to invest in more sustainable practices and to gain competitive advantage.

The study has also analysed the socio-economic impact of potential environmental policy, both within these sectors and across the economy more broadly. It has demonstrated that environmental policy can lead to positive environmental and economic outcomes within Europe. This analysis highlights that comprehensive action across sectors is required to achieve the desired environmental policy goals. The identification of future policy priorities in each of the sector studies show that the action required in each sector is distinct. Nonetheless, there are a number of key themes that emerge from the sector studies:

- There is substantial scope for the environmental impact of sectors with large environmental footprints to be reduced through public policy or through corporate social responsibility.
- Market-based instruments are options in many sectors, such as expanding the coverage of carbon taxes.
- In several sectors, a lack of investment is holding back (or has the potential to hold back) measures to reduce environmental impacts. Public policy interventions should aim to facilitate private sector investment in such cases, for example to fund the development of electric vehicle charging points or improve water supply infrastructure.
- The macroeconomic impact of sector-specific environmental policy is typically small economy-wide, although it can have major effects within the targeted sector(s), such as economic growth and job creation, as well environmental benefits.
- Negative effects of environmental policy are likely to be concentrated in sectors of resource extraction. Policy is required to manage a ‘Just Transition’, as socio-economic risks are most evident where there are geographical concentrations of these sectors.

This project has also highlighted some of the challenges to policy makers in successful implementation of environmental policy. Well-designed environmental policies may ensure economic growth and the environmental benefits of policy measures need careful consideration with regards to potential rebound effects and wider spillover effects – associated with increased consumption. As such, decisions about how revenues from taxation are used can have a major impact upon both the economic and the environmental implications from any policy.
Each of these factors need to be considered when designing environmental policy, to ensure that a policy framework effectively meets the environmental objectives of the 7th EAP, and at the same time facilitates economic growth and the creation of jobs:

- **Rebound effects** – environmental policy (particularly energy efficiency policy) can lead to additional consumption of energy because of increased incomes. Rebound effects need to be considered when determining the true benefits of environmental policy.

- **The role of technology** – technology plays a crucial part in the pathway to a resource efficient economy. However, new technologies face many barriers in their development and implementation. Government intervention is needed to ensure the true economic benefit of these technologies is realised, and firms are incentivised to innovate.

- **Financing of investment** – a key barrier faced by new technologies is inadequate financing from the private sector. Government intervention is required to encourage private sector investment in new clean technology developments.

- **Crowding out and capacity constraints** – low-carbon and circular economy transitions necessitate a significant reallocation of capital and labour resources in the economy, and effective environmental policy should therefore be designed considering current and anticipated future capacity constraints.

With regards to environmental protection expenditure, it can be concluded that better data collection is needed on a country / sector level within the EU. Looking at trends, spending is rather constant on environmental protection.
Appendix A – Studies

The following studies have been produced as part of this project and provide the evidence for this Final Report:

Environmental policy and the EAPSector studies
  - Food, drink & tobacco
  - Plastics
  - Motor Vehicles
  - Water supply
  - Waste & sewerage
Appendix B – E3ME Description

B.1 Overview
E3ME is a computer-based model of the world’s economic and energy systems and the environment. It was originally developed through the European Commission’s research framework programmes and is now widely used in Europe and beyond for policy assessment, for forecasting and for research purposes. The global version of E3ME provides:
• better geographical coverage
• better feedbacks between individual European countries and other world economies
• better treatment of international trade with bilateral trade between regions
• new technology diffusion sub-modules
This model description provides a short summary of the E3ME model. For further details, please read the full model manual available online from www.e3me.com.

B.2 Applications of E3ME
Scenario-based analysis
Although E3ME can be used for forecasting, the model is more commonly used for evaluating the impacts of an input shock through a scenario-based analysis. The shock may be either a change in policy, a change in economic assumptions or another change to a model variable. The analysis can be either forward looking (ex-ante) or evaluating previous developments in an ex-post manner. Scenarios may be used either to assess policy, or to assess sensitivities to key inputs (e.g. international energy prices).
For ex-ante analysis a baseline forecast up to 2050 is required; E3ME is usually calibrated to match a set of projections that are published by the European Commission and the International Energy Agency, but alternative projections may be used. The scenarios represent alternative versions of the future based on a different set of inputs. By comparing the outcomes to the baseline (usually in percentage terms), the effects of the change in inputs can be determined.
Price or tax scenarios
Model-based scenario analyses often focus on changes in price because this is easy to quantify and represent in the model structure. Examples include:
• changes in tax rates including direct, indirect, border, energy and environment taxes
• changes in international energy prices

Regulatory impacts
All of the price changes above can be represented in E3ME’s framework reasonably well, given the level of disaggregation available. However, it is also possible to assess the effects of regulation, albeit with an assumption about effectiveness and cost. For example, an increase in vehicle fuel-efficiency standards could be assessed in the model with an assumption about how efficient vehicles become, and the cost of these measures. This would be entered into the model as a higher price for cars and a reduction in fuel consumption (all other things being equal). E3ME could then be used to determine:
• secondary effects, for example on fuel suppliers
• rebound effects
• overall macroeconomic impacts

B.3 Comparison with CGE models and econometric specification
E3ME is often compared to Computable General Equilibrium (CGE) models. In many ways the modelling approaches are similar; they are used to answer similar questions and use similar inputs and outputs. However, underlying this there are important theoretical differences between the modelling approaches. In a typical CGE framework, optimal behaviour is assumed, output is determined by supply-side constraints and prices adjust fully so that all the available capacity is used. In E3ME the determination of output comes from a post-Keynesian framework and it is possible to have spare capacity. The model is more demand-driven and it is not assumed that prices always adjust to market clearing levels.
The differences have important practical implications, as they mean that in E3ME regulation and other policy may lead to increases in output if they are able to draw upon spare economic capacity. This is described in more detail in the model manual.

The econometric specification of E3ME gives the model a strong empirical grounding. E3ME uses a system of error correction, allowing short-term dynamic (or transition) outcomes, moving towards a long-term trend. The dynamic specification is important when considering short and medium-term analysis (e.g. up to 2020) and rebound effects, which are included as standard in the model's results.

Key strengths of E3ME

In summary the key strengths of E3ME are:

• the close integration of the economy, energy systems and the environment, with two-way linkages between each component
• the detailed sectoral disaggregation in the model’s classifications, allowing for the analysis of similarly detailed scenarios
• its global coverage, while still allowing for analysis at the national level for large economies
• the econometric approach, which provides a strong empirical basis for the model and means it is not reliant on some of the restrictive assumptions common to CGE models
• the econometric specification of the model, making it suitable for short and medium-term assessment, as well as longer-term trends

Limitations of the approach

As with all modelling approaches, E3ME is a simplification of reality and is based on a series of assumptions. Compared to other macroeconomic modelling approaches, the assumptions are relatively non-restrictive as most relationships are determined by the historical data in the model database. This does, however, present its own limitations, for which the model user must be aware:

• The quality of the data used in the modelling is very important. Substantial resources are put into maintaining the E3ME database and filling out gaps in the data. However, particularly in developing countries, there is some uncertainty in results due to the data used.
• Econometric approaches are also sometimes criticised for using the past to explain future trends. In cases where there is large-scale policy change, the ‘Lucas Critique’ that suggests behaviour might change is also applicable. There is no solution to this argument using any modelling approach (as no one can predict the future) but we must always be aware of the uncertainty in the model results.

The other main limitation to the E3ME approach relates to the dimensions of the model. In general, it is very difficult to go into a level of detail beyond that offered by the model classifications. This means that sub-national analysis is difficult and sub-sectoral analysis is also difficult. Similarly, although usually less relevant, attempting to assess impacts on a monthly or quarterly basis would not be possible.

B.4 E3ME basic structure and data

The structure of E3ME is based on the system of national accounts, with further linkages to energy demand and environmental emissions. The labour market is also covered in detail, including both voluntary and involuntary unemployment. In total there are 33 sets of econometrically estimated equations, also including the components of GDP (consumption, investment, international trade), prices, energy demand and materials demand. Each equation set is disaggregated by country and by sector.

E3ME’s historical database covers the period 1970-2014 and the model projects forward annually to 2050. The main data sources for European countries are Eurostat and the IEA, supplemented by the OECD’s STAN database and other sources where appropriate. For regions outside Europe, additional sources for data include the UN, OECD, World Bank, IMF, ILO and national statistics. Gaps in the data are estimated using customised software algorithms.

The main dimensions of the model

The main dimensions of E3ME are:

• 59 countries – all major world economies, the EU28 and candidate countries plus other countries’ economies grouped
• 44 or 70 (Europe) industry sectors, based on standard international classifications
• 28 or 43 (Europe) categories of household expenditure
• 22 different users of 12 different fuel types
• 14 types of air-borne emission (where data are available) including the 6 GHG’s monitored under the Kyoto Protocol

The countries and sectors covered by the model are listed at the end of this document.

Standard outputs from the model

As a general model of the economy, based on the full structure of the national accounts, E3ME is capable of producing a broad range of economic indicators. In addition, there is a range of energy and environment indicators. The following list provides a summary of the most common model outputs:

• GDP and the aggregate components of GDP (household expenditure, investment, government expenditure and international trade)
• sectoral output and GVA, prices, trade and competitiveness effects
• international trade by sector, origin and destination
• consumer prices and expenditures
• sectoral employment, unemployment, sectoral wage rates and labour supply
• energy demand, by sector and by fuel, energy prices
• CO2 emissions by sector and by fuel
• other air-borne emissions
• material demands

This list is by no means exhaustive and the delivered outputs often depend on the requirements of the specific application. In addition to the sectoral dimension mentioned in the list, all indicators are produced at the national and regional level and annually over the period up to 2050.

B.5 E3ME as an E3 model

The E3 interactions

Figure B.1 shows how the three components (modules) of the model - energy, environment and economy - fit together. Each component is shown in its own box. Each data set has been constructed by statistical offices to conform with accounting conventions. Exogenous factors coming from outside the modelling framework are shown on the outside edge of the chart as inputs into each component. For each region’s economy the exogenous factors are economic policies (including tax rates, growth in government expenditures, interest rates and exchange rates). For the energy system, the outside factors are the world oil prices and energy policy (including regulation of the energy industries). For the environment component, exogenous factors include policies such as reduction in SO2 emissions by means of end-of-pipe filters from large combustion plants. The linkages between the components of the model are shown explicitly by the arrows that indicate which values are transmitted between components. The economy module provides measures of economic activity and general price levels to the energy module; the energy module provides measures of emissions of the main air pollutants to the environment module, which in turn can give measures of damage to health and buildings. The energy module provides detailed price levels for energy carriers distinguished in the economy module and the overall price of energy as well as energy use in the economy.
Treatment of international trade
An important part of the modelling concerns international trade. E3ME solves for detailed bilateral trade between regions (similar to a two-tier Armington model). Trade is modelled in three stages:

- econometric estimation of regions’ sectoral import demand
- econometric estimation of regions’ bilateral imports from each partner
- forming exports from other regions’ import demands

Trade volumes are determined by a combination of economic activity indicators, relative prices and technology.

The labour market
Treatment of the labour market is an area that distinguishes E3ME from other macroeconomic models. E3ME includes econometric equation sets for employment, average working hours, wage rates and participation rates. The first three of these are disaggregated by economic sector while participation rates are disaggregated by gender and five-year age band.

The labour force is determined by multiplying labour market participation rates by population. Unemployment (including both voluntary and involuntary unemployment) is determined by taking the difference between the labour force and employment. This is typically a key variable of interest for policy makers.

The role of technology
Technological progress plays an important role in the E3ME model, affecting all three E’s: economy, energy and environment. The model’s endogenous technical progress indicators (TPIs), a function of R&D and gross investment, appear in nine of E3ME’s econometric equation sets including trade, the labour market and prices. Investment and R&D in new technologies also appears in the E3ME’s energy and material demand equations to capture energy/resource savings technologies as well as pollution abatement equipment. In addition, E3ME also captures low carbon technologies in the power sector through the FTT power sector model.

Main dimensions of the E3ME model
<table>
<thead>
<tr>
<th>Regions</th>
<th>Industries (Europe)</th>
<th>Industries (non-Europe)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Belgium</td>
<td>Crops, animals, etc</td>
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<tr>
<td>2</td>
<td>Denmark</td>
<td>Forestry &amp; logging</td>
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<tr>
<td>3</td>
<td>Germany</td>
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<td>4</td>
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<td>6</td>
<td>France</td>
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<td>Ireland</td>
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<tr>
<td>44</td>
<td>Brazil</td>
<td>Financial services</td>
</tr>
<tr>
<td>45</td>
<td>Argentina</td>
<td>Insurance</td>
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<tr>
<td>46</td>
<td>Colombia</td>
<td>Aux to financial services</td>
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<tr>
<td>47</td>
<td>Rest Latin Am.</td>
<td>Real estate</td>
</tr>
<tr>
<td>48</td>
<td>Korea</td>
<td>Imputed rents</td>
</tr>
<tr>
<td>49</td>
<td>Taiwan</td>
<td>Legal, account, consult</td>
</tr>
<tr>
<td>50</td>
<td>Indonesia</td>
<td>Architectural &amp; engineering</td>
</tr>
<tr>
<td>51</td>
<td>Rest of ASEAN</td>
<td>R&amp;D</td>
</tr>
<tr>
<td>52</td>
<td>Rest of OPEC</td>
<td>Advertising</td>
</tr>
<tr>
<td>53</td>
<td>Rest of world</td>
<td>Other professional</td>
</tr>
<tr>
<td>54</td>
<td>Ukraine</td>
<td>Rental &amp; leasing</td>
</tr>
<tr>
<td>55</td>
<td>Saudi Arabia</td>
<td>Employment activities</td>
</tr>
<tr>
<td>56</td>
<td>Nigeria</td>
<td>Travel agency</td>
</tr>
<tr>
<td>57</td>
<td>South Africa</td>
<td>Security &amp; investigation, etc</td>
</tr>
<tr>
<td>58</td>
<td>Rest of Africa</td>
<td>Public admin &amp; defence</td>
</tr>
<tr>
<td>59</td>
<td>Africa OPEC</td>
<td>Education</td>
</tr>
<tr>
<td>60</td>
<td></td>
<td>Human health activities</td>
</tr>
<tr>
<td>61</td>
<td></td>
<td>Residential care</td>
</tr>
<tr>
<td>62</td>
<td></td>
<td>Creative, arts, recreational</td>
</tr>
<tr>
<td>63</td>
<td></td>
<td>Sports activities</td>
</tr>
<tr>
<td>64</td>
<td></td>
<td>Membership orgs</td>
</tr>
<tr>
<td>65</td>
<td></td>
<td>Repair comp. &amp; pers. goods</td>
</tr>
<tr>
<td>66</td>
<td></td>
<td>Other personal serv.</td>
</tr>
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<td>67</td>
<td></td>
<td>Hholds as employers</td>
</tr>
<tr>
<td>68</td>
<td></td>
<td>Extraterritorial orgs</td>
</tr>
<tr>
<td>69</td>
<td></td>
<td>Unallocated/Dwellings</td>
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</table>

*Source(s): Cambridge Econometrics.*
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Links between production, the environment and environmental policy

*Modelling cross-cutting resource efficiency and its relevance to the 7th EAP*
This report is an Annex to the Final Report *Links between production, the environment and environmental policy*, ordered and paid for by the European Commission, Directorate-General for Environment, Contract ENV.F.1/FRA/2014/0063. The information and views set out in this study are those of the authors and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

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1 Introduction

The analysis presented in the sector studies, which form annexes to the main report from this study and are summarised in the final report, explores the environmental impact of specific sectors of the economy, how such impacts are changing over time, and conclude with a number of relevant policy recommendations to drive change in these industries.

In this report, we summarise the results of the modelling carried out for this study, the policy recommendations, their potential socioeconomic impacts (where quantified), and how the policies interlink with each other. We also set out how they link to the goals of the 7th EAP, and implications for the 8th EAP, all in Chapter 1.

In Chapter 2, we consider more broadly the challenges surrounding the implementation of successful environmental policy. The aim of this is to consider issues, some captured within the quantitative analysis and some not, which have the potential to impact the efficacy of environmental policy, based upon modelling work carried out in this project and elsewhere.
2 Sector-specific policy recommendations and their links to the 7th EAP

The 7th Environmental Action Programme (EAP) lists nine priority objectives and sets out how they can be achieved by 2020 (EC, undated). There are three specific key objectives;

- to protect, conserve and enhance the Union’s natural capital
- to turn the Union into a resource-efficient, green, and competitive low-carbon economy
- to safeguard the Union’s citizens from environment-related pressures and risks to health and wellbeing.

These are then supported by four “enablers”;

- to maximise the benefits of the Union’s environment legislation by improving implementation
- to increase knowledge about the environment and widen the evidence base for policy
- to secure investment for environment and climate policy and account for the environmental costs of any societal activities
- to better integrate environmental concerns into other policy areas and ensure coherence when creating new policy

with two “horizontal” priority objectives;

- to make the Union’s cities more sustainable
- to help the Union address international environmental and climate challenges more effectively.

The policies that have been discussed and suggested in the sector studies address, to varying degrees, each of these aims. In the sections that follow, the policy recommendations from each sector are considered in turn, in terms of how they map to the objectives of the 7th EAP, the potential positive impacts of the policy and potential barriers or challenges to successful implementation of the policies.

2.1 Food, drink & tobacco

The potential policies explored in the sector study were;

- Introducing a carbon tax on produce
- Extending producer responsibility for food and drink producers
- Excluding the production of sugar from CAP support
- Minimum harmonised public procurement rules
- Variable charges for sector waste

Of these policies, only excluding the production of sugar from CAP support specifically addresses the goal to protect, conserve and enhance the Union’s natural capital, through reducing incentives to plough grassland for arable use. All policies address the second aim, to turn the Union into a resource-efficient, green, and competitive low-carbon economy, with most explicitly aiming to reduce environmental impacts of production (thereby meeting the greening requirement), while most also address resource efficiency; most notably extending producer responsibility for food and drink producers and introducing variable charges for sector waste, both of which explicitly seek to reduce wastage either within the sector or by households (through consumption).

The sector studies highlight potential benefits from these policies, but they also highlight potential challenges. Table 2.1 summarises these.

| Benefits and challenges from environmental policy in the food, drink & tobacco sector |
|-----------------------------------------------|-----------------------------------------------|
| **Introducing a carbon tax on produce** | **Impacts of policy** | **Challenges** |
| | Incentivises low-carbon production methods | Potential for leading to shift towards imports |
| | GDP increased by 0.04%, employment 0.04% higher in | Positive economic impacts lead to rebound effect |
2.2 Plastics

The potential policies explored in the sector study were:

- Increased recycled content in products
- Introduction of plastics tax
- Increased re-use of plastics

These policies relate to all three broad aims of the EAP, in that they will serve to improve resource efficiency (through reducing production of plastic) and they will improve Europe’s natural capital, and reduce human exposure to plastic and microplastic waste.

The impacts and challenges around these policy recommendations are summarised in Table 2.2.

Table 2.2 Benefits and challenges from environmental policy in the plastics sector

<table>
<thead>
<tr>
<th>Policy Description</th>
<th>Impacts of policy</th>
<th>Challenges</th>
</tr>
</thead>
<tbody>
<tr>
<td>Increased recycled content in products</td>
<td>Reducing consumption of raw materials and energy. Reduced CO₂ emissions. Increased employment from recycling.</td>
<td>Can lead to lower quality output (or needs improvements in technology) Need to design policy to ensure that it targets broad plastics use, not just low-value uses.</td>
</tr>
<tr>
<td>Introduction of plastics tax</td>
<td>Small increase in GDP and employment. Stimulates technological innovation.</td>
<td>Difficult to target. Positive economic impacts will lead to rebound effects. Potential for carbon leakage (both direct and indirect).</td>
</tr>
</tbody>
</table>
Increased re-use of plastics

Reducing consumption of raw materials.
Can have minimal cost implications (e.g. deposit refund schemes).
Potential for increased take-up of collaborative economy business models.

2.3 Motor vehicles

The potential policies explored in the sector study related to measures aimed at shifting demand towards low-carbon mobility options and managing (i.e. reducing, or at least limiting growth in) overall demand for road transport. The kinds of policies considered were;

- Market-based instruments (e.g. production taxes, in-use vehicle taxes, in-use energy taxes, in-use road charges, end-of life charges and deposits)
- Supporting the development of (charging) infrastructure
- Improving the competitiveness of the European battery industry
- Supporting the decarbonisation of the electricity grid
- Encouraging the deployment and use of autonomous vehicles in a way that is environmentally beneficial
- Encouraging take-up of shared ownership models

The two broad aims of policy link particularly to the 2nd and 3rd goal of the 7th EAP, to turn the Union into a resource-efficient, green, and competitive low-carbon economy and to safeguard the Union’s citizens from environment-related pressures and risks to health and wellbeing, in that a shift to EVs would reduce the environmental impact of both the production and use of motor vehicles, while the deployment of shared ownership models, and more broadly policies which limit increases in demand for road transport, will safeguard human health in terms of reducing exposure to harmful particulates (particularly those associated with tyre wear, which are not reduced as part of the shift to EVs).

The impacts and challenges surrounding these policies are explored in Table 2.3.

Table 2.3 Benefits and challenges from environmental policy in the plastics sector

<table>
<thead>
<tr>
<th>Impacts of policy</th>
<th>Challenges</th>
</tr>
</thead>
</table>
| **Deployment of market-based instruments** | **Encourages shift to vehicles with smaller environmental footprint**  
Can mitigate fuel duty losses (if done through taxes rather than incentives)** | **Supply shortages (e.g. of EVs) can undermine policy**  
Reduced cost of mobility can lead to rebound effects**  
Requires continued technology development to replace all use cases** |
| **Supporting development of charging infrastructure** | **Encourages take-up of alternative powertrains**  
Can solve “chicken and egg” problem of lack of infrastructure** | **Requires public sector funding**  
Can lead to inefficient deployment of infrastructure**  
Environmental impact of infrastructure manufacture and installation** |
| **Improve competitiveness of** | **Creates output and jobs in** | **Relies on technological** |
European battery manufacture
Reduces environmental footprint of battery production.
Reduces import dependency in supply chains.
Development in Europe (with associated uncertainty).

Encourage deployment of autonomous vehicles
Reduce manufacturing of vehicles (through lower demand).
Potential rebound effects (increased usage of vehicles).

Encouraging take-up of shared ownership models
Reduces manufacturing of vehicles.
Potential for substantial rebound effects.

2.4 Water treatment and supply
This study looked primarily at the investment required in Europe’s water treatment and supply network in order to meet all current legislation. The main policy recommendations were:
- Increased use of blended finance
- Offering of green bonds
- Working closely with water dependent industries
- Improved supporting systems (e.g. regulations, urban design)

These policies, and their broad aim to improve Europe’s water infrastructure, meet all three broad aims of the 7th EAP; they reduce impacts on the natural environment (and therefore protect natural capital), they improve resource efficiency and they reduce risks to human health.

Since all policies are aimed at improving the state of water treatment and supply infrastructure, the policies share the same broad impacts, although challenges are different. These are explored in more detail in Table 2.4.

<table>
<thead>
<tr>
<th>Impacts of policy</th>
<th>Challenges</th>
</tr>
</thead>
<tbody>
<tr>
<td>Increased use of blended finance</td>
<td>Leverage increased investment from financial markets (i.e. private sector)</td>
</tr>
<tr>
<td>Offering of green bonds</td>
<td>Increase clarity of offering</td>
</tr>
<tr>
<td>Working closely with water dependent industries</td>
<td>Improved awareness of role of water in economic activity</td>
</tr>
<tr>
<td></td>
<td>Allows funding of infrastructure that is not attractive to private finance</td>
</tr>
<tr>
<td>Improved supporting systems</td>
<td>Can reduce risk in water infrastructure investment</td>
</tr>
<tr>
<td></td>
<td>Provides potential to boost European water technology sector</td>
</tr>
</tbody>
</table>
2.5 Sewerage and waste management

The sector study sets out a number of different policy areas that could be addressed to improve waste management in Europe;

- Using waste as an energy source
- Increased capture of microplastics
- Increased recycling targets
- Further bans on landfill
- Improve minimum standards for some waste streams

These policies address primarily the second aim of the EAP, to improve resource efficiency and create a greener and more competitive European economy. However, the improved capture of microplastics addresses all three, in that it also maintains the quality of Europe’s natural capital and have the potential to reduce environmental damage to human health.

Table 2.5 highlights the potential beneficial impacts of these policies, as well as the challenges to implementation.

Table 2.5 Benefits and challenges from environmental policy in the sewerage and waste management sector

<table>
<thead>
<tr>
<th>Impacts of policy</th>
<th>Challenges</th>
</tr>
</thead>
</table>
| Using waste as an energy source | Reduces landfill  
Potential to reduce carbon footprint of energy generation | Requires investment  
Resultant energy could have high cost  
Environmental impacts not clear |
| Increased capture of microplastics | Reduces human and environmental damage from microplastics | Requires investment |
| Increased recycling targets | Reduces consumption of virgin materials  
Reduces landfill | Increases costs to industry |
| Further bans on landfill | Reduces waste generation  
Encourages lowest-cost recycling/re-use | May force take-up of high-cost (but low environmental impact) technologies |
| Improve minimum standards for some waste streams | Facilitates recycling and re-use  
Reduces virgin material consumption | |
| Improve processing | Allows higher rates of recycling and re-use  
Reduces virgin material consumption | |
2.6 Mapping EAP areas to Sector Studies

Table 2.6 Mapping priorities outlined in the 7th EAP to Sector Studies

<table>
<thead>
<tr>
<th></th>
<th>Food, drink, and tobacco</th>
<th>Plastics</th>
<th>Motor vehicles</th>
<th>Water treatment and supply</th>
<th>Sewerage and waste management</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural capital: air</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Natural capital: water</td>
<td></td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Natural capital: land</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Resource efficiency, waste, and the circular economy</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Biodiversity</td>
<td></td>
<td>X</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Low-carbon and climate change</td>
<td>X</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Human health and well-being</td>
<td></td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

Table 2.6 maps the detailed thematic priorities of the 7th Environmental Action Programme to the five sectors assessed in this study. The mapping highlights the fact that for most objectives, comprehensive action across sectors is required. The identification of future policy priorities in each of the sector studies show that the action required in each sector is distinct. Given the distinct nature of policy interventions across sectors, the macroeconomic effects are expected to be largely additive – that is, the macroeconomic impacts from each policy are distinct, and not inter-related; so introducing all policies across all themes simultaneously would lead to an impact on GDP equivalent (approximately) to the sum of the individual impacts.

2.7 Macroeconomic Impacts of Resource Efficiency

Resource efficiency is a key component of the EC’s 7th EAP; managing waste flows and sustainable use of natural resources are needed to live within the means of our planet. This section addresses the recurring research question as to whether pursuing environmental policy, and particularly resource efficiency, is also economically beneficial. Identifying potential economic co-benefits of environmental policy can be used to engage stakeholders, where they would not take action motivated purely by environmental concerns.

Resource productivity at the economy level can be measured by calculating GDP per unit of raw material consumption. Cambridge Econometrics (2014) argues that whilst resource productivity in the EU28 has increased substantially in recent years, 19.6% from 2001 to 2011, raw material consumption has remained relatively coupled with economic growth; before the financial crisis, from 2001 to 2007, average annual resource productivity was +1.2% but GDP growth was on average +2.3%. This study used the material module in E3ME to develop a baseline for future resource productivity to 2030, considering existing EU policy. The baseline projections suggest a relative decoupling of economic growth and raw material consumption; however, given economic growth projections, this still implies significant absolute increases in consumption of metals and minerals. The transition to a low-carbon economy is likely to still require significant raw material consumption in construction of new capital.

There are a number of competing factors which affect the macroeconomic impacts of increasing resource efficiency. Table 2.7 lists the positive and negative effects of resource efficiency policy, identified by

61 Absolute increase in raw material consumption of metals and minerals, from 2012 to 2030, are estimated to be 39% and 26% respectively.
Policy to increase resource productivity imposes costs on industry; market-based instruments would directly increase the cost of material use, and regulation would require industry to invest in abatement measures. Investment in resource efficiency is the key positive economic driver, particularly in the short to medium term.

Table 2.7 Key macroeconomic impacts of resource efficiency policy

<table>
<thead>
<tr>
<th>Positive</th>
<th>Negative</th>
</tr>
</thead>
<tbody>
<tr>
<td>Investment in resource efficiency</td>
<td>Higher costs of resource use, higher domestic prices</td>
</tr>
<tr>
<td>Revenue recycling of market-based instrument revenue (dependent on policy choice)</td>
<td>Lower exports; reduced competitiveness of domestic industry (resource use equivalent of carbon leakage)</td>
</tr>
<tr>
<td>Imports; decrease in imports of resources</td>
<td></td>
</tr>
</tbody>
</table>

Macroeconomic modelling in Cambridge Econometrics (2014) and Meyer (2011) find evidence that ‘win-win’ situations of increased resource productivity and economic benefits are possible, under certain conditions. Meyer (2011) scenario analysis finds that across a balance policy mix (recycling, taxation, and information and consulting) a reduction of total material requirement by 1% is accompanied by an average increase of GDP between 12 and 23 billion € and an increase in employment of between 0.04% and 0.08% across EU27 Member States. Cambridge Econometrics (2014) finds positive GDP effects for the EU28 for scenarios of resource productivity improvement up to 1.25 percentage points more ambitious than baseline forecasts. Negative results, in the long term, are found for a scenario of 2.25 percentage points greater ambition in their reduction of total material requirement; when abatement measures are increasingly expensive, and costs of investment outweigh positive impacts.

Sectoral effects of resource productivity should also be considered. Intermediate sectors which sell raw materials face significant contraction of output. Material resource intensive sectors face higher proportional increases in costs. However, sectors such as manufacturing and construction, which are material-intensive benefit from demand stimulus from investment in resource efficiency. Effects across MSs are likely to be driven by relative domestic importance of resource extraction and processing, manufacturing, and construction industries.

Sectoral-level scenario analysis in this project provides complementary evidence to these studies of economy-wide resource use. Modelling of a plastics tax focused on examining substitution to recycled, rather than absolute reduction of material use: positive GDP and employment effects are found across tax design scenarios, accompanied by a contraction of the plastics sector. Similarly, modelling reductions of food waste, positive GDP and employment effects are accompanied by a contraction of the FDT and agricultural sectors. Positive macroeconomic effects in these scenarios largely rely on the revenue recycling mechanism of any tax revenues.

In the following chapter, the modelling undertaken in Cambridge Econometrics (2014) is updated (to take account of more recent data) and expanded to consider in more detail environmental impacts of the envisaged resource efficiency.

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62 There may be some zero cost improvements in resource efficiency, associated with waste reduction and information improvement (EC, 2014). In this case, industry costs would fall in this part of the abatement cost curve.

63 Meyer (2011) provides a summary of the outcomes of the project “Macroeconomic Modelling of Sustainable Development and the Links between the Economy and the Environment”, which included E3ME modelling by Cambridge Econometrics.

64 Plastics sector output does not contract under a border tax scenario with intermediate demand characterised by perfect inelasticity.
3 The economic and environmental impacts of cross-economy resource efficiency

3.1 Introduction
This analysis provides evidence of the socioeconomic and environmental effects of policy targeted at achieving sustainability in raw material consumption. The scope of the modelling work is economy wide, at a relatively high-level aggregation of material groups. Scenario modelling employs the materials module in E3ME, originally developed for the European Matisse research project. The module incorporates materials modelling, in physical terms, to the E3ME framework. Detailed documentation of the module can be found in Pollitt (2008). This work focuses on targeting increases in resource productivity, defined by GDP per unit of Raw Material Consumption (RMC).

This economy-wide modelling complements the evidence base developed for individual sectors. Assessment of policy at this aggregation does not permit examination of sectoral level economic/technological/political challenges, but it provides evidence of effects of comprehensive policy. Importantly, this includes capturing the impacts of conflicting and complimentary policy measures.

3.2 Methodology

Estimating price elasticities
The materials data from Eurostat’s material flow accounts provides detailed data up to 2016.

The econometric relationships for material use in E3ME have been re-estimated. Specifications of the equations are unchanged from Cambridge Econometrics (2014). Material consumption intensity (DMI (domestic material input) per unit of output) is a function of economic activity (+/-), material prices (-) and measures of technology (-).

As previously, long-run price elasticities for material intensity are estimated at the EU level. Attempts to estimate these elasticities at the national/sectoral output were made but did not produce robust results. These elasticities are a key parameter in this analysis, and therefore identifying robust results is crucial. The EU price elasticities are imposed only on the long-run price parameters for each region, short run price elasticities are estimated at the sectoral/country level.

The functional form of the econometric equations allows the parameter estimates to be interpreted as elasticities. The elasticities give the percentage change in material consumption for a 1% increase in price. Table 1 below details the estimated long-run price elasticities in the previous study, and current work. The magnitude of elasticities is markedly lower across most materials. The relative elasticities across materials is largely consistent, however. The highest elasticities in both estimations are ores, forestry and construction minerals. Food remains the least elastic.

The difference in magnitude of elasticities is likely to be a result of the different economic environment in the period of new data, namely 2011-2016. In particular, the price index for value added of the NACE sector “B Mining and quarrying” decreases substantially and consistently from 2012 to 2016 (price of minerals and ores use is estimated by the price of this sector). Both agriculture and forestry are characterised by more similar time series, and more similar price elasticity estimates.
Price elasticity of animal feed could not be estimated because the production and purchasing sector are the same (i.e. agriculture). For this reason, animal feed was excluded from the calculation of resource productivity targets and modelling of policies.

Resource productivity

Resource productivity is defined as GDP per unit of raw material consumption, where RMC is measured in raw material equivalent. The focus of this study is non-energy materials, and therefore all reported data and calculations represent non-energy materials. The average annual increase in resource productivity from 2000 to 2011 was 2.1%. Resource productivity from 2011 onwards has increased at a similar rate of 1.9% per annum. Raw material consumption remains lower than before the financial crisis.

Conversion of material imports/exports to raw material equivalents follows the same process as in Cambridge Econometrics (2014). Coefficients for conversion were updated using data from Eurostat’s material flow accounts in raw material equivalents modelling estimates.
The E3ME baseline is calibrated to the PRIMES 2016 Reference Scenario and the long-term projections in the 2015 Ageing Report. The forecasts for future material use intensity were taken from Cambridge Econometrics (2014), and extrapolated to 2040. It is notable that the trend of resource productivity from 2020-2030 is substantially lower under new assumptions compared to figures reported in Cambridge Econometrics (2014). The key issues at play here is that the 2014 study focused exclusively on non-energy material use. Resource productivity improvements in that study were significantly higher in energy-carrier materials. The baseline value of 0.85% resource productivity improvement from 2014 to 2030 corresponds to a 0.22% per annum improvement in non-energy material improvements from 2018-2030.

### Table 3.2 Annual average growth rates in the E3ME reference case

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP Growth</td>
<td>1.5%</td>
<td>1.4%</td>
<td>1.6%</td>
</tr>
<tr>
<td>Total RMC (non-energy)</td>
<td>1.5%</td>
<td>1.4%</td>
<td>1.7%</td>
</tr>
<tr>
<td>Resource Productivity</td>
<td>0.0%</td>
<td>0.1%</td>
<td>-0.1%</td>
</tr>
</tbody>
</table>

**Scenario modelling in E3ME**

For each scenario, improvement in resource productivity comes from:

- 1/3 publicly funded investments in the capital stock to improve resource efficiency
- 1/3 privately funded business measures, stimulated by regulatory policy
- 1/3 market-based instruments (MBI) (such as tax)

Marginal abatement cost curves were first developed for each material group and each material user. This process provides an indication of the relative costs/benefits of targeted policy interventions. The curves were calculated as impacts of achieving a one percent reduction in raw material consumption by 2040.

The investment requirement for improvement of material use efficiency uses data for energy efficiency investment, given a lack of more appropriate data (a similar assumption to that used in the 2014 study). Assessing cumulative investment in energy efficiency in the IEA WEO 2018 New Policy Scenario to achieve a 1% improvement in energy use, compared to the Current Policy Scenario, yields a value of annual investment of 3.91bn 2010EUR.
Modelling choices for revenue recycling, investment funding mechanism, and world commodity prices are the same as the 2014 study.

- Each scenario employs revenue recycling. MBI tax revenues are collected by national governments and recycled back at Member State level through lower income taxes and employers’ social security payments.
- In each scenario, it is assumed there is no change to world raw material prices. Changes in the EU alone are not likely to have substantial impacts on global material markets. The exogenous commodity price series uses data from the 2018 World Bank Commodities Price Forecast.
- Investment in material use efficiency is funded from material tax revenues. Any surplus/deficit is recycled/funded through adjustments to income and taxation and employers’ social security contributions.

The process of scenario design was iterative following the calculation of the marginal abatement cost curves. A simple scaling up and aggregation of policies across material users and groups does not yield the material consumption reductions implied by adding abatement effects in the cost curves. There are important interaction effects in introducing policy across the economy.

The investment in material use efficiency increases demand for construction activity and manufacturing equipment. These sectors are key consumers of construction minerals and metal ores. Substitution effects in consumer expenditure are less strong when consumer prices rise across more consumption categories. Positive GDP effects and increased consumer income cause rebound effects. Further, the magnitude of reduction in material use intensity and magnitude required in the scenarios is such that limits of reduction are reached.

The 2% scenario is not a least cost-scenario. Policies are introduced across all material users and material groups. This represents economy wide policy action, and follows the calculations of the 2014 study in estimating ‘potential reductions’. Though not modelled, the 2% scenario could be achieved by removing the most costly policies and scaling up interventions in other sectors. The cost curve was used to remove the most costly polices to achieve the 1.5% and 1% targets.

Further, adjustments were made in the treatment of food. In calculation of the marginal abatement cost curve, the tax rates required to achieve the given 1% reduction in RMC are substantially higher than other materials, given the inelastic nature of food demand. In calculating the MBI curve, the net economic results are positive for food, drink, & tobacco; the reason being that the revenues to be recycled were so substantial given the size of the demand and the high tax rate required to stimulate demand reductions. Whilst GDP results are positive, such a tax causes a very strong inflationary effect, and such a tax is politically unrealistic and highly regressive. Material users “agriculture” and “food, drink and tobacco” are the greatest consumers of food raw material, and are the two most costly options in the regulation curve.

The scenario analysis was carried out to 2040, using cost curve information in the same year. There are difficulties in assessing substantial resource efficiency in the long run post-2040 without strong systemic and technological innovation assumptions especially for the more ambitious scenario. The approach applied in this analysis does not lead to credible results beyond 2040, based as it is upon coefficients estimated from historical data, while such a deep and sustained change in resource efficiency would require much more systemic change. For this reason, the time horizon of our analysis was curtailed to 2040 to ensure more robust results.

Scenarios are defined by a single EU28 resource productivity growth rate. Three levels of ambition are presented, namely 1%, 1.5% and 2% per annum improvement in resource productivity. EU28 average GDP growth rate from 2018-2040 in the baseline is 1.5%. Therefore, the least ambitious scenario achieves a relative reduction in raw material consumption only. The 1.5% RP scenario results in a total RMC in 2040 very similar to the 2018 (i.e. increases in RMC from output growth are counteracted by improved resource efficiency). The 2% RP scenario achieves an absolute reduction of RMC. Both the 1.5% and 2% RP scenarios suggest an absolute decoupling of GDP growth and RMC; however, in practice that is not true; increased economic activity still increases material demand – the modelling simply assumes that such an increase is negated by improvements in resource efficiency.
**Modelling environmental impacts**

E3ME allows for quantifying emissions of greenhouse gases and air pollutants from fuel combustion. However, the impacts of production processes often have associated non-combustion emissions such as emissions of heavy metals to air from metals processing or emissions of volatile organic compounds from certain plastics (e.g. styrene from polystyrene production).

In order to account for a wider range of impacts, E3ME outputs were supplemented with environmental impact multipliers from EXIOBASE v3.3.14 - the most detailed environmentally-extended input-output database currently available (Stadler et al., 2018; latest at time of writing). This allows for assessing a total of 29 types of emissions to air (GHGs, air pollutants, heavy metals and refrigerants) from both combustion and non-combustion processes, as well as water pollution (from N and P emissions to water) and land use (occupation of land only, i.e. no land-use change). What is more, E3ME allows for a detailed breakdown of resource use.

The large range of emissions and resource uses included presents a challenge in itself due to the number of variables under analysis. In order to allow for more streamlined interpretation, plus link use and emissions with environmental impacts, a life-cycle impact assessment (LCIA) step is performed. LCIA allows for aggregation of results from different emissions and resource uses into a smaller number of policy-relevant impacts such as Global Warming Potential and Eutrophication potential (e.g. Guinée, 2002)

The following sections detail the range of impacts included, as well as the procedure of linking E3ME outputs with EXIOBASE impact multipliers, and the procedure for LCIA.

The assessment includes the following emissions and resource uses:

- GHGs – CO2, CH4 and N2O, including from agriculture and waste management
- Air pollution – PM10, PM2.5, SOx, NOx, CO, NMVOCs, PAHs, PCBs and heavy metals
- Water pollution – N and P emissions from agriculture and waste management
- Land use – both agricultural and industrial land uses
- Resource use – ferrous and non-ferrous metals, food feed and forestry products, construction and industrial minerals

Of the above, CO2 emissions and resource use are taken directly from E3ME outputs. For CO2 emissions, this is to ensure that emissions explicitly align with E3ME material uses (some discrepancies may be introduced for EXIOBASE outputs – see below). For resource use, this is in order to explicitly align with economic forecasts, as well as in due to E3ME’s more detailed and policy-relevant disaggregation of material use types.

The rest of emissions (as well as land use) assessed are included via EXIOBASE environmental impact multipliers (see below for method). The assessed indicators are the most extensive possible based on what is available via EXIOBASE. See Appendix 1 for a full list of indicators assessed.

**Estimating future impacts using EXIOBASE**

In order to extend E3ME impacts with additional ones from EXIOBASE, the following procedure has been utilised (for all EXIOBASE indicators Appendix 1):

- EXIOBASE provides total (economy-wide) impacts per industry and per country for all variables in Appendix A.
- Total industry impacts per country for 2011 (latest base year in EXIOBASE) are calculated through coefficients based on E3ME data for the same year as follows:
  - Where environmental impacts are a result of combustion, the coefficient is via total combustible fuel use
  - All other impacts are measured via gross output
  - For electricity supply, which is highly disaggregated in EXIOBASE, generation is tracked via individual disaggregated categories. Transmission and distribution impacts are tracked via the “electricity supply” sector itself.
The EXIOBASE coefficients are multiplied by E3ME output for forecasted years 2030 and 2040, yielding a country and industry-specific impact in E3ME sectoral classification.

In order to derive impacts in E3ME classification, a concordance table between EXIOBASE and E3ME has been used - this is given in Appendix B.

This results in impacts for all variables included in Appendix A, which are further aggregated at the life-cycle impact assessment step.

Life-cycle impact assessment

While comprehensive in scope, the range of impacts derived via EXIOBASE are difficult to interpret due to their breadth. The aggregation of multitudes of emissions and resource uses into a limited set of policy-relevant indicators is a principal aim of life-cycle impact assessment (e.g. Guinee, 2002). Within the development of EXIOBASE v2, LCIA impact factors have also been developed in order to allow for this (Wood et al., 2015). From the LCIA categories given, we use the five categories recommended by the study authors along with an extra category for land use, namely:

- Acidification potential (Huijbregts, 1999a) – in kg SO2 eq.; CML 2001 Problem oriented approach: baseline (Guinee, 2002)
- Eutrophication potential (Heijungs et al. 1992) – in kg PO4 eq.; CML 2001 Problem oriented approach: baseline (Guinee, 2002)
- Photochemical oxidation potential (Jenkin and Hayman, 1999; Derwent et al., 1998) – in kg ethylene eq.; CML 2001 Problem oriented approach: non-baseline (Guinee, 2002)
- Human toxicity potential (infinite time horizon; Huijbregts 1999b; Huijbregts et al., 2000) – kg 1,4-dichlorobenzene eq.; CML 2001 Problem oriented approach: non-baseline (Guinee, 2002)
- Global warming potential (100-year time horizon; IPCC, 2007) – in kg CO2 eq.; CML 2001 Problem oriented approach: non-baseline (Guinee, 2002)
- Land use – in km2, summation of individual EXIOBASE land-use categories

EXIOBASE v2 and v3.3.14 impacts and sectors are identical save for v3.3.14 non-combustion emissions and N and P emission to water disaggregated by applicable technologies and crop types. Due to this, it is straightforward to map EXIOBASE v2 LCIA factors to v3.3.14.

The environmental impacts from Appendix 1 for the base year (2011) and forecast years (2030, 2050) are multiplied by applicable LCIA factors for the 5 categories outlined above (plus land-use is aggregated) in order to arrive at the end-set of life-cycle impact assessment results analysed. LCIA impact assessment results are further supplemented by E3ME material use outputs, which serve as a detailer “resource use” indicator.

3.3 Expected socioeconomic outcomes

The key dynamics in the modelling are the increased cost of material use, and operation of the revenue recycling mechanisms. The below diagram (from Cambridge Econometrics (2014)) illustrates the propagation mechanism of regulation and MBI costs. Both price-based policies and regulation increase the cost of consuming raw materials. Cost increases are passed through to product prices of these industries. The magnitude of cost-pass through is determined by the parameters in the price formation equations in E3ME; parameters are estimated using sectoral-level historical data. Industries operating in a competitive international market are likely to pass on less of the policy cost burden. Final product prices to consumers are ultimately affected by policy costs in the supply chain. The two key impacts of price increases are: 1) reduced industry competitiveness in international markets and 2) reduced real household disposable income.
The choice of revenue recycling mechanism ameliorates these effects:

- public funding for investment – industries do not bear the cost of additional investment
- reducing income tax – to help offset any reduction in real incomes due to higher prices
- reducing employer’s social security contributions – to reduce cost of labour and partially offset higher material input costs

### 3.4 Socioeconomic results

This section presents results for three scenarios, each representing different levels of ambition, namely S2: 1%, S3: 1.5% and S4: 2% yearly improvement in resource productivity from 2018 to 2040. S1 is the baseline to which the socioeconomic results are compared to. These scenarios do not reflect the combination of sectoral policies presented and analysed but these are derived from previous resource efficiency work (Cambridge Econometrics, 2014)

Table 3.3 Socioeconomic impacts in 2030, percentage difference from baseline

<table>
<thead>
<tr>
<th></th>
<th>S2</th>
<th>S3</th>
<th>S4</th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP</td>
<td>0.9%</td>
<td>1.0%</td>
<td>0.7%</td>
</tr>
<tr>
<td>Employment</td>
<td>0.5%</td>
<td>0.5%</td>
<td>0.3%</td>
</tr>
<tr>
<td>Consumer spending</td>
<td>1.2%</td>
<td>1.1%</td>
<td>0.1%</td>
</tr>
<tr>
<td>Investment</td>
<td>1.3%</td>
<td>2.0%</td>
<td>2.5%</td>
</tr>
<tr>
<td>Imports (extra-EU)</td>
<td>-0.2%</td>
<td>-0.1%</td>
<td>-0.4%</td>
</tr>
<tr>
<td>Exports (extra-EU)</td>
<td>-0.3%</td>
<td>-0.3%</td>
<td>-0.6%</td>
</tr>
</tbody>
</table>

Figure 3.3 How cost changes interact with the economy in E3ME
Table 3.4 Socioeconomic impacts in 2040, percentage difference from baseline

<table>
<thead>
<tr>
<th>2040</th>
<th>S2</th>
<th>S3</th>
<th>S4</th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP</td>
<td>1.6%</td>
<td>1.6%</td>
<td>0.3%</td>
</tr>
<tr>
<td>Employment</td>
<td>0.9%</td>
<td>0.9%</td>
<td>0.3%</td>
</tr>
<tr>
<td>Consumer spending</td>
<td>2.3%</td>
<td>2.1%</td>
<td>-0.4%</td>
</tr>
<tr>
<td>Investment</td>
<td>1.2%</td>
<td>1.9%</td>
<td>1.8%</td>
</tr>
<tr>
<td>Imports (extra-EU)</td>
<td>-0.5%</td>
<td>-0.5%</td>
<td>-1.1%</td>
</tr>
<tr>
<td>Exports (extra-EU)</td>
<td>-0.5%</td>
<td>-0.5%</td>
<td>-1.0%</td>
</tr>
</tbody>
</table>

**GDP**

The GDP and employment impact are positive for all scenarios, throughout the period to 2040. These results show that policy targeting resource productivity can have positive macroeconomic impacts on the EU economy.

The main drivers of positive GDP effects in the scenarios are: 1) investment in material efficiency; and 2) revenue recycling effects. The main negative drivers are high material use prices leading to: 1) loss of consumer purchasing power; and 2) loss of competitiveness in extra-EU exports.

Investment in material efficiency provides a demand stimulus through all years. Scenarios 2 and 3 show an increasing positive effect to 2040. This increasing trend is driven by revenue recycling effects from increasing material tax revenues. S4 starts with the largest positive effect in 2019, because the investment requirement is highest in the most ambitious scenario. Positive GDP effects reduce from 2035 in S4, however. The most economically costly policies are included in S4, and these effects start to dominate from 2035. The key driver is inflationary effects reducing consumer purchasing power.

**Investment**

The main impact on economy level investment is that made in material efficiency. This investment is spread evenly (in real terms) throughout the years 2019-2040. Investment in agriculture, forestry, and mining reduce in all scenarios, but this effect is minor compared to the material efficiency investment. Changes throughout the rest of the economy are limited and largely reflect GDP trends.
Trade
Net effects of extra-EU trade are minimal. Across all scenarios the effects of policies is to reduce total trade activity. Imports of raw materials reduce as EU domestic demand falls. Exports fall due to higher input costs and associated competitiveness effects. Reductions in imports are dominated by sectors such as agriculture, mining and metals. Reductions in exports are more widespread however, as higher input costs affect all industries.

Consumer spending and inflation
Consumer expenditure, in real terms, increases in all scenarios by 2030. This is a result of higher employment and lower income taxation rates. Price inflation, propagated through higher industry costs and prices, exerts a downward impact on consumer spending. The aggregate consumer price level is very sensitive to changes in food costs, and the heavily inflationary regulation is only introduced in the most ambitious scenario, as outlined above.

Employment
The pattern of employment changes largely follows the trend of GDP changes, as expected. Other than the stimulus to derived demand for labour from material efficiency investment and consumer expenditure from revenue recycling, there are two key effects increasing employment. First, the increase in cost of material use incentivises substitution of material for labour inputs. Second, revenue recycling directed to reducing employers’ social security contributions reduces labour costs to industry and encourages additional employment.

Impacts on the income distribution
Real income increases are markedly equitable in S2 and S3. Reductions in income tax from revenue recycling benefit across the income distribution; if resolution was higher it might be expected that the lowest income group, who are less likely to draw a substantial portion of their household income from taxable earnings. In contrast, S4 is characterised by increased income inequality. Real income falls across the distribution, but substantially more for the lower quintiles. This effect is driven by increasing food prices.

It should be noted that the specification of revenue recycling is a modelling decision, which provides a transparent and simple mechanism. In reality, more nuanced fiscal policy could be used to ameliorate the negative distributional effects.

Table 3.5 EU28 real income effects by quintile (% difference from baseline)

<table>
<thead>
<tr>
<th>Quintile</th>
<th>S2</th>
<th>S3</th>
<th>S4</th>
</tr>
</thead>
<tbody>
<tr>
<td>All households</td>
<td>2.7%</td>
<td>2.3%</td>
<td>-0.7%</td>
</tr>
<tr>
<td>1st</td>
<td>3.0%</td>
<td>2.6%</td>
<td>-1.7%</td>
</tr>
<tr>
<td>2nd</td>
<td>2.7%</td>
<td>2.4%</td>
<td>-1.2%</td>
</tr>
<tr>
<td>3rd</td>
<td>2.7%</td>
<td>2.3%</td>
<td>-0.9%</td>
</tr>
<tr>
<td>4th</td>
<td>2.7%</td>
<td>2.3%</td>
<td>-0.5%</td>
</tr>
<tr>
<td>5th</td>
<td>2.6%</td>
<td>2.2%</td>
<td>-0.1%</td>
</tr>
</tbody>
</table>

CO2 Effects
EU level CO2 emissions increase in all scenarios, throughout the forecast period. This increase, however, is limited. Material efficiency and reduction of extraction activities reduces related emissions. However, emissions resulting from investment activities in material efficiency, and increased consumer expenditure outweigh this effect. In these scenarios there is no policy targeted at use of energy or fossil fuels.

Table 3.6 EU28 CO2 impacts (% difference from baseline)

<table>
<thead>
<tr>
<th>Year</th>
<th>S2</th>
<th>S3</th>
<th>S4</th>
</tr>
</thead>
<tbody>
<tr>
<td>2030</td>
<td>0.3%</td>
<td>0.4%</td>
<td>0.3%</td>
</tr>
<tr>
<td>2040</td>
<td>0.6%</td>
<td>0.6%</td>
<td>0.3%</td>
</tr>
</tbody>
</table>
Comparison to the 2014 study results
Socioeconomic outcomes are consistent with those reported in Cambridge Econometrics (2014). The most ambitious scenario has the lowest GDP, employment and consumer expenditure effects. GDP is positive throughout the forecast in the present study, however, and more negative effects only dominate later in the scenario.

3.5 Environmental results

Table 3.1 Environmental impacts, percentage change from 2018 baseline

<table>
<thead>
<tr>
<th></th>
<th>2018</th>
<th>BASE</th>
<th>BASE</th>
<th>S2</th>
<th>S2</th>
<th>S3</th>
<th>S3</th>
<th>S4</th>
<th>S4</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>2030</td>
<td>2040</td>
<td>2030</td>
<td>2040</td>
<td>2030</td>
<td>2040</td>
<td>2030</td>
<td>2040</td>
</tr>
<tr>
<td>Acidification potential</td>
<td>th</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
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<td>14.6%</td>
<td>-1.5%</td>
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<tr>
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<td>-9.5%</td>
<td>-6.5%</td>
<td>-9.4%</td>
<td>-6.3%</td>
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<td>-12.7%</td>
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<td>Non-ferrous ores use</td>
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<td>15.3%</td>
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<td>-7.0%</td>
<td>-3.9%</td>
<td>-7.0%</td>
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Acidification potential
The results for acidifying pollutants indicate only a marginal increase in potential impact in the base case toward 2040, with impacts under a 1% and 2% increase in material efficiency yielding similar decreases in impacts of approx. 13% toward 2040, and with a 3% increase in material efficiency resulting in a -15.7% decrease in impacts.

Closer inspection shows that the decrease in acidification potential is due in part to decreases of SOx and NOx emissions from combustion, and primarily due to decreases in NH3 emissions to air from agriculture. The Agriculture, Fishing & Forestry sector contracts most strongly in gross output amongst all sectors – the decreases in combustion emissions of SOx and NOx is thus owed to decreased fuel use in the sector, as combustion emissions in EXIOBASE-E3ME are scaled based on total combustible fuel use. The decrease in NH3 emissions from agriculture - i.e. non-combustion on-farm emissions such as due to fertilisers and animal excreta – is due to the decrease in overall output of the sector.

Eutrophication potential
The results for eutrophication potential indicate an increase of impacts of 8.6% in the base case by 2040, while these decrease by approximately 13% in the S2 and S3 scenarios, and by approximately 18% for S4 by 2040.

The results mostly mirror those for acidification potential and are linked to the projected dynamics of the Agriculture, Fishing & Forestry sector. Baseline results indicate modest decreases in NOx emissions from combustion, but these are outweighed by N emissions to water and NH3 emissions to air owed to fertiliser
application and urea from animal excreta. The base case increase in impacts can be explained by the increased productivity in the Agriculture, Fishing & Forestry sector owed to rising demand for food.

In the cases of imposed increase in material efficiency, the contracting of gross output of the Agriculture, Fishing & Forestry sector leads to impact reductions primarily due to decrease in N and NH3 emissions from fertilisers and urea, as well as in part due to decreased emissions of NOx from combustion.

Photochemical oxidation potential
Photochemical oxidation (i.e. summer smog) potential increases by approx. 4% in the base case and varies only weakly under scenarios of increased resource efficiency. Under these scenarios, some decrease in SOx, CO and N2O occur (summer smog precursors) due to reduction in combustible fuel use and due to reduction of CH4 emissions from enteric fermentation in agriculture, the latter from contracting of the agriculture sector. However, these are outweighed by increased non-combustion emissions of NMVOCs and CO from industry. Careful interpretation is necessary, as combustion emissions in E3ME-EXIOBASE are linked to combustible fuel use, while non-combustion emissions are linked to sectoral gross output. What is more, constant 2011 technology (latest in EXIOBASE) is assumed for EXIOBASE emissions, meaning that efficiencies and reductions from technological progress are not captured. Thus, while combustible fuel use may decrease due to increased material efficiency gains, the coupling of non-combustion emissions and sectoral gross output means that with projected (modest) growth in industrial sectors in E3ME, non-combustion emissions also modestly increase.

Human toxicity potential
Potential impacts from human toxicants (such as air pollution, heavy metals, polyaromatic hydrocarbons and polycyclic biphenyls) are projected to increase by 8.6% in the base case. Emissions of air pollutants from combustion have only a moderate influence and the bulk of impact is due to non-combustion emissions of PAH from industry, owed to the projected increase in industry output.

Potential impacts are projected to further increase in the resource efficiency scenarios; by approximately 10% compared to the baseline with only marginal differences between scenarios. The influence of different emission types mostly mirrors the trends in the base case but the increase in non-combustion emission impacts (due to linkage with gross output) is dampened by the decrease in emissions from reduced combustion emissions from decreased fuel use. As noted, fixed technology assumptions for emissions modelled via EXIOBASE does not take into account any future reductions due to technological advancements.

Global warming potential
Greenhouse gas emissions are projected to decrease in all cases, with the base case having a 23.4% decrease toward 2040. CO2 decreases in line with decarbonisation as calibrated to the PRIMES Reference Scenario. Non-CO2 combustion emissions decrease with decreasing combustible fuel use and though overall GHG emissions decrease, this is slightly dampened by increases in non-combustion emissions, as these are coupled in EXIOBASE-E3ME to overall sectoral gross output growth.

Under the projected resource use efficiency gains, GHG emissions fall further by approx. 26% in the 1% and 2% efficiency gain scenarios, and by approx. 27% in the 3% scenario. GHG emission reductions are largest for the Agriculture, Fishing & Forestry due to its strongest contraction in output, with also a moderate influence of contraction of the Non-metals mining sector as well.

Land use
Land use rises by 12.6% in the base case scenario toward 2040 in line with overall growth in GDP and growth in demand for food and forestry products.
3.6 Conclusions

In this analysis we have used a macroeconomic model, E3ME, and coefficients on environmental damage from EXIOBASE to assess the economic and environmental impacts of potential resource efficiency scenarios.

The modelling shows broadly positive socio-economic impacts, although these are very sensitive to the assumptions made with regards to how the effort to realise resource efficiency is balanced across the economy. In particular, costly resource efficiency in the agricultural and food, drink & tobacco sectors can lead to negative economic outcomes, particularly when looking across the income distribution, reflecting low price elasticities and the higher proportion of household income spent on food in low-income households.

The scenarios show environmental benefits from improved resource efficiency across impacts where combustion or agricultural emissions are prime drivers (global warming potential, acidification and eutrophication potentials), as well as for resource use apart from food use.

The environmental impacts are mixed, although there are sustained decreases in emissions from combustion. The results are partly a result of the methodology applied; links to gross output (rather than to DMC) mean that not all expected impacts from resource efficiency are captured in the environmental measures; in addition, in the absence of clear data on the role of technology in altering production techniques and therefore environmental impacts, EXIOBASE coefficients are held constant over time, leading to the likely over-estimation of environmental impacts.

Nonetheless, these results make clear that in aggregate there are both economic and environmental benefits from some level of resource efficiency; but that as resource efficiency exceeds certain limits dictated by the costs of abatement, the economic impacts start to diminish and ultimately turn to costs (in terms of GDP, employment and equality).
The challenges to the successful implementation of policy

The 7th EAP will guide European policy to 2020, and sets out a vision of where it would like the EU to be in 2050:

*In 2050, we live well, within the planet’s ecological limits. Our prosperity and healthy environment stem from an innovative, circular economy where nothing is wasted and where natural resources are managed sustainably, and biodiversity is protected, valued and restored in ways that enhance our society’s resilience. Our low-carbon growth has long been decoupled from resource use, setting the pace for a safe and sustainable global society.*

The environmental policies described in the previous section detail how the vision of the 7th EAP can be achieved through the decarbonisation of production in key sectors.

However, there are some cross-cutting issues faced by environmental policy, which can in some cases present challenges or barriers to achieving the goals of the 7th EAP:

- **Rebound effects** – environmental policy (particularly energy efficiency policy) can lead to additional consumption of energy because of increased incomes. Rebound effects need to be taken into account when determining the true benefits of environmental policy.

- **The role of technology** – technology plays a crucial part in the pathway to a decarbonised economy. However, low-carbon technologies face many barriers in their development and implementation. Government intervention is needed to ensure the true economic benefit of these technologies is realised, and firms are incentivised to innovate.

- **Financing of investment** – a key barrier faced by low-carbon technologies is inadequate financing from the private sector. Government intervention is required to encourage private sector investment in new clean technology developments.

- **Crowding out and capacity constraints** - low-carbon and circular economy transitions necessitate a significant reallocation of capital and labour resources in the economy, and effective environmental policy should therefore be designed considering current and anticipated future capacity constraints.

Each of these factors need to be considered when designing environmental policy, to ensure that a policy framework effectively meets the environmental objectives of the 7th EAP, and at the same time facilitates economic growth and the creation of jobs.

In the following subsections we discuss each of these cross-cutting issues in turn and consider the implications for environmental policy.

### 3.1 Rebound effects

One of the main objectives of the EAP is to ‘turn the Union into a resource-efficient, green, and competitive low-carbon economy’ and improving energy-efficiency is a key policy consideration in meeting this objective. The potential energy savings and emissions reductions to be realised from energy efficiency measures and improvements in technology are often estimated using complex models and analysis, but in reality, evidence suggests the actual energy savings and emissions reductions achieved by policy are lower than expected.

#### 3.1.1 Direct and indirect rebound effects

There is a theory that the gap between the estimated and actual environmental impacts of environmental policy occurs because of a behavioural response known as the ‘rebound effect’. The rebound effect occurs because decarbonisation measures such as energy efficiency measures lead to lower running costs, and therefore additional income for energy users to spend on other things. While spare income and greater expenditure are good things for society and the economy, they ultimately lead to further demand for energy, which can have a detrimental impact on the environment. Other environmental policy measures, such as the promotion of collaborative and circular economy activities also produce unwanted increases in energy use and emissions, due to positive changes in household incomes and expenditure. Economic agents’ behaviour and attitudes can also have a part to play. For example – while environmental policy may encourage individuals to walk or cycle to work every day, they then may feel that by ‘doing their bit’ for the
environment, their conscience is clear to carry out some other environmentally un-friendly activity such as going on an overseas flight. The existence of rebound effects therefore calls in to question the effectiveness of some environmental policy.

Rebound effects can be both direct and indirect. Direct rebound effects occur when households and/or firms use this additional income to consume a technology more. For example, if household heating systems are more energy efficient, we may use the money saved on bills to simply keep our heating on for longer, or in the case of firms, a reduction in energy costs leads to greater profitability, and potentially greater investment and levels of production.

There are two types of indirect rebound effects. First, indirect rebound effects occur if households instead spend the additional income that results from lower energy costs or from greater income earned through collaborative economic activities on other goods and services in the economy, creating increased demand for energy, and increased emissions in other sectors. For example, households may spend their additional income on an overseas holiday, which includes a flight with a high carbon footprint. Second, an overall reduction in energy use will mean less demand for fossil fuels, but as a result, fossil fuel prices are lowered, and greater consumption of these fuels is encouraged.

3.1.2 The scale of rebound effects

Direct and indirect rebound effects may vary in size depending on the specific policy or intervention in question, the sector, products or services targeted by the policy, and the technological change it encourages. Furthermore, income and productivity levels, price elasticities, geographical location and time scales can all have an impact on the magnitude of rebound effects, making it very difficult to quantify their precise impact (Maxwell et al 2011). A comprehensive review of literature concerning the impacts of rebound effects carried out by Turner (2013) concludes that ‘there is a lack of agreement and clarity in the literature regarding how ‘rebound’ should be measured’. Therefore, within the literature, the magnitude of rebound effects is greatly debated.

Barker et al (2009) is a unique study that considers the global impact of rebound effects. The study estimates the total (direct plus indirect) rebound effect of energy efficiency policies aimed at transport, buildings and industrial sectors of the global economy in the period 2013-2030 using the E3MG model. The research finds that the total rebound effect could be 31% by 2020, reaching 52% by 2030. Rebound effects accumulate over time as the multiplier effects of higher real incomes and greater investment increase.

The scale of the direct rebound effect varies between socioeconomic groups, and therefore differs between developed and developing economies. Rebound effects within the poorest in society are likely to be largest, since these groups are likely to consume more heat or road transport if costs are reduced, enjoying previously unaffordable warmth or the option to drive a car more frequently. The wealthier groups in society are likely to have already reached ‘saturation’ points for some energy consuming activities, and a reduction in costs does not make any difference to how much they heat their homes or use their cars, for example (Grubb 2014).

On this basis, direct rebound effects in developed countries such as the EU Member States are likely to be lower than in poor, developing countries. However, indirect rebound effects can still be substantial within developed countries, making energy efficiency policies less effective at reducing overall energy use. In recent years, direct and indirect rebound effects have been given more attention, as governments of developed countries have questioned why economy-wide energy use has not fallen as much as expected in response to energy efficiency measures.

In 2007 the UK government commissioned a landmark report by the UK Energy Research Centre (UKERC) to investigate evidence of rebound effects (Sorrell 2007). The report finds there is robust evidence of direct rebound effects when considering energy efficiency measures aimed at automotive transport and household heating and cooling in developed countries. The direct rebound effect is estimated to be up to 30% for household heating (i.e. 30% of the energy savings from energy efficiency are offset by households consuming more heating), and around 10% for automotive transportation. Furthermore, when indirect rebound effects are taken into account, the economy-wide reduction in energy use will be smaller still. For other types of consumer energy services and for changes in energy use by producers, the evidence is
weaker, and it is difficult to draw reliable conclusions. However, the report concludes that in the case of other consumer energy use, such as using electrical appliances, changes in energy costs are unlikely to have much influence on operating decisions, and there is unlikely to be much of a direct rebound effect.

Similarly, in 2011 the European Commission commissioned a study to estimate the magnitude of rebound effects that could occur in the EU (Maxwell et al 2011). The study combined a comprehensive literature review with key stakeholder engagement and finds that rebound effects do exist and can be substantial in many cases, but that the effect varies in size depending on the sector, product or service targeted by intervention, the intervention itself and location and timeframe of the intervention. Specifically, direct rebound effects for energy efficiency for heating and cooling, cars and other energy services in OECD countries are estimated to be in the range of 10-30%. For energy efficiency improvements in lighting, the direct rebound effect is estimated to be between 5% and 12% in developed countries for private households and 0-2% for industry and commerce. For commercial road transport, the direct rebound effect of fuel efficiency is 30-80%. Evidence for energy efficiency improvements in industry show a direct rebound effect estimate of 15% in the UK. The study concludes that estimates of indirect and economy wide rebound effects are limited due to little existing research, and that this research has weaknesses in its approach. From the published literature, the economy wide rebound effects of energy efficiency improvements are estimated to be smaller than direct rebound effects; approximately 10%. In some particular cases, however, the economy-wide effect may be greater than 30% and in some extreme cases, it is possible that the rebound effect may be so large, that the energy savings and emissions reductions initially achieved through the implementation of policy or change in technology are in fact cancelled out (known as ‘backfire’, which is discussed in the following paragraphs).

3.1.3 Khazzoom-Brookes Postulate

Examples of large rebound effects exist throughout history, as first highlighted by Jevons (1865) in his work on the consumption of coal in the UK, which increased despite improvements in the efficiency of using this resource. the rebound effect resulting from the development of more efficient lighting is often cited as a historical example of the phenomenon. Although modern-day electric light bulbs are vastly more efficient than gas lights or oil lamps, today we use just as much energy lighting our homes, streets and businesses as in the past, except we now enjoy more of it. It has even been questioned whether energy efficient lightbulbs are in fact bad for the environment. Similar examples exist for material resources. Dardozzi (2008) highlights the case of word processing and it’s use of paper. Before the development of personal computers, producing a typewritten document was time consuming and expensive. While it may have been initially perceived that the widespread use of computers in offices and workplaces could lead to ‘paperless’ offices, it is in fact that case that paper consumption increased due to the ease, speed and cheapness of producing a typewritten document.

The idea that energy efficiency or improvements in technology in fact lead to increased energy consumption overall was initially conceptualised by Jevons (1865). In what has become known as the Jevons Paradox, Jevons described how greater energy efficiency (in this case, of coal) brought about by technological change leads to reduced costs, and therefore greater demand for that form of energy. Jevon’s work was further developed by Khazzoom (1980), (1987) and (1989) and Brookes (1990), and was later referred to as the Khazzoom-Brookes Postulate by Saunders (1992). Saunders takes the earlier work a step further and describes how energy efficiency not only encourages energy users to use technology more, but how energy efficiency also leads to greater economic growth (through greater spending), and therefore greater economy-wide energy use. In this case the energy efficiency policy is said to ‘backfire’. This is particularly the case when money that would otherwise have been spent in one sector is substituted for money spent in other, more energy- and carbon-intensive industries, such as households spending more money on air travel.

The Khazzoom-Brookes Postulate has been a subject of much attention, since if the theory is correct, energy efficiency measures could be seriously flawed in their capacity to make a positive impact on the environment. However, the relevance of the Khazzoom-Brookes Postulate has been questioned, since the

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theory is based on an economic model which makes many assumptions and has various limitations. Furthermore, empirical evidence of the Postulate is often ambiguous. Nonetheless, rebound effects can still have a substantial impact on the effectiveness of energy efficiency policy, whether the effect is greater than 100% or not, and therefore should still be taken into account when making policy decisions.

3.1.4 Technological change

The abovementioned example of lighting represents a case of rebound effects that come about because of technological change determined by market forces, not government intervention. Grubb (2014) points out that in some cases technological change comes about not because of policy measures (such as energy efficiency measures), but because there is a demand for more of the good or service. In the case of lighting, consumers demanded more and better lighting, leading to innovations in more cost-effective technology. In cases such as this, it is inevitable that energy efficiency leads to greater consumption of the technology. However, Grubb also acknowledges that there are many cases where energy efficiency improvements are not led by changes in demand and are instead a result of policy intervention, such as fuel efficiency in cars. Improvements in technology are not necessarily occurring because individuals want to drive more, but because environmental policy encourages the development of more efficient vehicles. Grubb concludes that ‘it is easy to confuse correlation with cause and effect’.

3.1.5 Policy-specific challenges

Rebound effects present challenges when implementing energy efficiency measures in particular since, unlike various other environmental policy measures, energy efficiency measures reduce costs for energy users. For this reason, much of the research and literature that considers rebound effects focuses on rebounds from energy efficiency measures. However, rebound effects are also particularly problematic for resource efficiency policies, since these measures lead to more efficient use of resources such as water, lowering costs and therefore encouraging greater consumption of the resource. Other environmental policy measures and initiatives, such as the promotion of circular or collaborative economic activities, can also lead to rebound effects in energy use and levels of emissions.

Resource efficiency

Improving resource efficiency helps safeguard the environment and the ecosystems it provides, through decreased material use and therefore reduced pressure on the EU’s natural capital. As part of the EU’s aim to improve resource efficiency, the Eco-innovation Initiative supports developments in innovative products, technologies and services that protect the environment, become commercially viable. While eco-innovation is positive for the environment, European Commission (2011) acknowledges that rebound effects often result from increased productivity that improvements in technology bring about, and that this improved efficiency translates into higher demand for resources.

Maxwell et al. (2011) describes rebound effects caused by behavioural responses, not necessarily brought about by changes in price. While price-induced rebound effects typically occur because of energy or resource efficiency measures, behavioural effects can also occur through the psychological perceptions of economic agents, whereby there is a perception of being ‘green’ in one instance, encouraging increased consumption for less environmentally friendly goods and services in another instance. A study by Tiefenbeck et al (2013) presents the case in which a water saving campaign in a building complex in Massachusetts successfully reduced water use by 6%, but at the same time, electricity usage increased by 5.6%. The study suggests that a behavioural response known as ‘moral licensing’ is at play, whereby consumers exchange a positive behaviour for a negative one.

In the absence of a well-designed policy mix, the environmental benefits of resource efficiency measures may be partially offset by rebound effects. To mitigate these rebound effects, adverse behavioural responses can be targeted by policy makers through attempts to influence behaviour and habits, for example through information and awareness campaigns, the use of smart meters or environmental standards for products and appropriate labelling. These kinds of policy initiatives should be combined with appropriate technological and fiscal measures to maximise the potential of environmental policy.
Collaborative economy

The ‘collaborative economy’, also known as the ‘sharing economy’, describes economic activities where collaborative platforms create a marketplace for the temporary usage of goods or services, often provided by private individuals (Trinomics et al. 2017). A collaborative economy can be beneficial for the environment, since it reduces material consumption through the sharing of goods, and can reduce emissions through activities such as ride-sharing in the transport sector. These collaborative activities can help the EU meet various objectives of the 7th EAP such as protecting, conserving and enhance the Union’s natural capital, turning the Union into a resource-efficient, green, and competitive low-carbon economy and making the Union’s cities more sustainable. However, research shows that such activities can also lead to rebound effects. Since households gain additional income from renting out goods, or providing accommodation or transport to others, they then spend this additional income in other sectors of the economy, creating economic growth, but also increasing energy use and emissions.

Circular economy

Circular economy activities emphasise the utilisation of recycled inputs and ensure that the inputs to production can be recycled while preserving or enhancing their economic value (Aldersgate Group, 2012). Similar to collaborative economy activities, circular economy activities also help the EU to meet many of the objectives of the 7th EAP. However, rebound effects in consumer spending occur because of efficiency gains and cost savings from circular economy activities, and changes to the capital-labour ratio as the focus shifts from primary economic sectors to secondary and tertiary sectors and from capital-intensive activities like manufacturing, to labour-intensive activities like repairing or refurbishing. While circular economy activities are beneficial for the environment, rebound effects from greater consumer spending mean the economy-wide impacts on material consumption, energy use and emissions are unclear (Cambridge Econometrics et al. 2018).

3.1.6 Policy considerations

There is much evidence that various environmental policies (particularly those aimed at improving energy efficiency) can lead to both direct and indirect rebound effects, which offset the original reduction in energy use achieved. Quantifying the extent of the rebound effect is difficult, but various studies have shown that there is indeed evidence of both direct and indirect rebound effects existing in developed economies to some degree. Estimates of the effectiveness of environmental policy aimed at improving energy efficiency often do not take rebound effects in to account, and therefore overstate the contribution these policies can make to reductions in energy consumption and GHG emissions. The evidence outlined in this section suggests that although rebound effects can be difficult to recognise and quantify, it is important to acknowledge them in analysis linked to environmental policy decision-making, otherwise environmental goals may not be fully achieved. Furthermore, rebound effects vary between sectors, by technology and by societal groups. Taking specific rebound effects in to account can help policy-makers target policy more effectively, or offset some of the impact of rebound effects by coupling energy efficiency policy with other, targeted environmental policy or carbon pricing.

Combining energy efficiency policy with carbon pricing and taxation is particularly important, as increasing the cost of carbon can help substantially reduce both direct and indirect rebound effects. This is because carbon pricing or taxation can be used to keep the cost of carbon stable (or higher than it otherwise would be), while energy efficiency is simultaneously increased. Greater use of energy, whether through greater use of the technology that is more energy efficient, or greater use of energy elsewhere, will be discouraged because of high energy prices. However, carbon pricing and taxation can create unwanted adverse effects on international competitiveness of firms or income distribution within society. Furthermore, carbon pricing and taxation does not address the problem rebound effects pose for material consumption (i.e. increased demand for goods and therefore greater resource use). Therefore, energy efficiency measures should be complemented by various other forms of environmental policy. As Grubb (2014) recommends, countries should have strong policy across three policy response pillars – standards and regulation (which includes energy efficiency measures), markets and prices (carbon taxes and market-based instruments) and strategic investment (investment in clean and renewable technologies). Combining a mix of policies from all three pillars ensures that any increased demand for energy due to rebound effects is not met by greater
consumption of fossil fuels, while also creating opportunities for innovation and economic growth. Addressing the behaviour of economic agents is also important, and education or regulation that leads to complementary behavioural changes or better business practices can be key to the effectiveness of other environmental policy. For example, in the case of transport, policy aimed at improving vehicle fuel-efficiency and increasing the cost of using fossil fuels can be complemented by regulation to fit all new vehicles with meters providing feedback on how driving behaviour affects fuel use (Barker et al 2009).

There are clear economic benefits of rebound effects from energy efficiency, resource efficiency and other environmental policy measures. Assuming that savings made on energy or increased efficiency of a resource are instead spent on consuming more of the good or service, or on other goods and services in the economy, additional economic activity is generated, boosting GDP and employment. However, the economic benefits of the rebound effect may be somewhat detrimental to the environmental targets the policy first set out to achieve. As with many environmental policy measures, policy-makers are faced with the challenge of ensuring that the policy achieves the double-dividend of both reducing emissions, energy and resource use, while achieving economic growth. In the case of energy efficiency measures, little evidence exists that rebound effects would lead to overall increases in energy use that exceed the original reduction in energy use achieved (i.e. ‘backfire’). Therefore, energy efficiency policy is still effective at reducing energy use and emissions, while also leading to greater expenditure and economic growth. Likewise, the evidence of rebound effects in resource efficiency policy

3.2 Technology

Improvements in technology are a crucial part of the pathway to a resource efficient economy for the EU, helping to meet the targets of policies set forth by the 7th EAP. As well as playing a major role in the improved resource efficiency of the EU economy, innovation and investment in new technologies also benefit the EU economy through increased economic growth and jobs, contributing towards the EU’s innovation, jobs and growth agenda. Policies and initiatives aimed at increasing investment in new technologies therefore satisfy the policy goal of achieving environmentally sustainable economic growth.

3.2.1 Types of technology

Technologies for a low-carbon future can be categorised into four key sectors; power, transport, buildings and industry, and many of these are also relevant when considering key options for improving resource efficiency. In the power sector, renewable energy technologies that generate electricity through wind, solar, hydro or ocean, geothermal or bioenergy, instead of through fossil fuels, are playing a substantial role in the pathway to a greener EU economy. In transport, developments in electric vehicles and the use of alternative fuels such as biofuel in aviation are ongoing, helping to both decarbonise and improve overall resource efficiency. Advancements in heating and cooling systems and greater energy efficiency in appliances will be essential for the buildings sector to reduce its resource efficiency. Finally, the development of new technologies in industry offers the potential to improve both energy and resource efficiency. As previously cited within this report, countries should have a strong policy agenda across three policy response pillars – standards and regulation, markets and prices and strategic investment in clean and renewable technologies (Grubb 2014).

3.2.2 Investment in low-carbon technologies

The EU Emissions Trading System (ETS) creates incentives for firms to invest in and develop alternative low-carbon technologies, due to the relatively high running costs of energy- and carbon-intensive technologies. Environmental taxation tends to be more focussed, but ultimately has a similar aim of encouraging the development of new resource-efficient technologies. However, it is often difficult for such technologies to move past a pilot phase of R&D, to full-scale commercial viability, due to financial constraints.

66 https://ec.europa.eu/clima/policies/lowcarbon_en
Data suggests that energy-related sectors spend a very small proportion of turnover on R&D expenditure (R&D intensity), compared to other sectors of the economy. In the electricity, alternative energy, and oil and gas sectors, the top investing EU firms spent between 0.1% and 7.5% of total sales on R&D expenditure in 2017. - a much lower figure than that of other sectors such as biotechnology (24.0%), pharmaceuticals (13.7%) or software (11.3%). In the automobile sector, in which a large amount of low-carbon technology development is ongoing, and in which the EU outperforms investment in other global regions, R&D intensity was still only 5.5% for EU firms in 2017.

The European Commission has various programmes to help fund the development of low-carbon and resource efficient technologies, such as the NER 300 programme, the European Economic Recovery Plan and the Strategic Energy Technology Plan. However, these funds only go so far in aiding R&D, and investment from the private sector is also required. But developing and commercialising a new technology is expensive, time-consuming and often risky – presenting innovators with many challenges in securing enough investment to bring their products to the marketplace.

### 3.2.3 Barriers to innovation

The low R&D intensity seen in the power, transport and industrial sectors can be explained through the existence of various barriers to investment in new technologies, which make these technologies less attractive investment options compared to well-established conventional technologies.

There are several stages to the innovation chain, and for a new technology to be commercially successful, all stages of the chain must be connected. Schumpeter (1939) first defined the innovation chain as three distinct stages – invention, innovation and diffusion. Other economists (e.g. Grubb 2014) have extended the chain to include more detailed stages, but the underlying process remains. For the chain to be connected, two forces need to work simultaneously – technology push, or the supply of innovation, and market pull, or the demand for innovation. In the case of low-carbon technologies, often there are missing links in the innovation chain, with various barriers creating gaps that mean some technologies never make it to the marketplace, and fall in to what is known as the technology 'valley of death' (Murphy and Edwards 2003).

Grubb (2014) summarizes six distinct barriers to innovation faced by new (specifically, low-carbon) technologies, of which three relate to challenges surrounding ‘technology push’, and three relate to challenges surrounding ‘market pull’. These barriers lead to stages of the innovation chain failing to connect, and the ultimate failure of many low-carbon technologies.

In terms of ‘technology-push’, the barriers are:

- Uncertainties and knowledge divisions around risks and rewards: The rewards of new technologies are uncertain, and the potential for a marketplace is not fully understood. Investors may prefer to wait and see what emerges.
- Transitions from technical to management and commercial skills: Technology researchers may have limited knowledge of potential markets and business practice and may fail to formulate a convincing business plan. Investors’ money becomes easily diverted to firms with inferior technology but better sales pitches.
- High capital costs and long timescales: Are both typical characteristics of new low-carbon and resource efficient technologies. The more investors have to commit up front and the longer they have to wait to see a return on investment, the greater the uncertainties (including around government policies) and risks, deterring investors.

For 'market-pull', the barriers are:

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69 https://ec.europa.eu/clima/policies/lowcarbon/ner300_en
Economics of scale and experience: Innovation requires improvements in the technology through learning-by-doing and economies of scale. But initially there is insufficient volume to fuel enough learning-by-doing to reach a sustainable market.

Misalignment of private and public goals: investors may conclude that technologies backed by governments have been motivated for reasons not aligned with commercial interests and are thus riskier.

Incompatible public policies and understanding of the full innovation chain: establishing new industries requires the coordination of many elements, including associated infrastructures and regulatory frameworks.

Policy must be cleverly designed to both facilitate R&D and to create incentives for private investors to support innovation and bring new technologies to the marketplace. Policy options can be classified according to whether they address the technology push barriers, or whether they address the barriers related to market pull. Technology-push policies support the development of initial ideas (i.e. from the invention stage of the innovation chain) to commercially-viable products. Policies could include public funding of R&D, subsidies and grants and demonstrations of the technology and its application. To address the market-pull barriers, policy needs to be aimed at creating a viable marketplace for new technologies. In this context, policy options include carbon and resource pricing (making existing technologies expensive to run) and standards and regulation, all creating a demand for more resource efficient alternatives.

In the case of new more resource efficient technologies, a substantial barrier to investment exists because market prices for conventional technologies or those that are powered by traditional fossil fuels do not fully reflect the true cost associated with resource extraction and use. These technologies may be cheaper to produce or run than more resource efficient technologies, but in fact they lead to detrimental impacts on natural resources and human health. Resource efficient technologies are often seen as being more expensive and less competitive than traditional solutions, and with no guarantee that these technologies can compete in the open marketplace, investment is more likely to be aimed at well-established technologies. But if the true costs of these technologies were taken in to account, resource efficient alternatives would be more economically viable. Policy makers can attempt to correct the price of resources through market-based instruments such as taxes, making investment in more resource efficient alternatives more attractive.

The EU ETS is the EU’s mechanism for carbon pricing, and is one such example. It helps to encourage the take-up of low-carbon technologies. The impact of the EU ETS on the rate of low-carbon innovation amongst EU firms is demonstrated in a paper by Calel & Dechezlepretre (2016) (see Figure 3.1). Before the introduction of the EU ETS in 2005, similar rates of low-carbon innovation were seen between regulated and unregulated firms. However, after the introduction of the ETS, 30% more patents in low-carbon technologies were filed by firms regulated by the system, particularly in renewable energy, energy storage, energy efficiency and carbon sequestration.
While carbon pricing is an essential policy tool helping to aid the development of low-carbon technologies, other barriers to investment will still exist. As with all technologies, costs of production and running costs of low-carbon solutions are initially high compared to other, well-established technologies powered by conventional fuels or using conventional materials or production methods. Over time, learning factors and economies of scale should bring costs of new technologies down, and the economic benefits of these technologies should out-strip those offered by conventional technologies. However, for this process to occur, the technology must move past the innovation stage of development, and widespread diffusion of the technology is required. But diffusion can be a very time-consuming and uncertain process, making new technologies less attractive and riskier for investors. While uncertainty is always a factor in investment decisions, in the case of low-carbon technologies uncertainty is high due to the lack of full information on future government policy, and future carbon pricing. Difficulties in securing investment for low-carbon technologies are further exacerbated by the typically slow returns to investment offered by these technologies. These risks are particularly relevant to resource efficient technologies, where market-based instruments are more focussed, and therefore perceived as more subject to change than even the EU ETS.

Investment in new technology R&D (of any kind) often carries the risk that investors and innovators will not profit from the new technology before knowledge starts to spill over to other firms or sectors. This risk creates further barriers to investment in new technologies, and research suggests that the spillovers from ‘clean’ technologies (that avoid GHG emissions) are consistently up to 40% higher than spillovers from ‘dirty’ technologies (that produce GHG emissions) (Dechezleprêtre et al 2017). Furthermore, the marginal economic value of spillovers from new technologies is also higher. Combined, these factors can dissuade investors supporting new technologies if they believe others will profit from their investment. Government intervention can help overcome this market failure through strong property rights legislation. However, the
pros and cons of intervention of this kind should be carefully considered, first since strong property rights can lead to monopolies in the marketplace, and second since spillovers are beneficial for the wider economy, creating economic growth.

3.2.4 Policy considerations

Together, the barriers to investment discussed above can lead to resource efficient technologies becoming trapped in what is known as the technology ‘valley of death’ whereby high costs and low volumes prevent the technology becoming commercially viable, and many technologies can fail to reach the marketplace. Aiding the innovation of such technologies is a key priority for policy-makers since they provide clear economic and environmental benefits. Alongside market-based instruments to internalise external costs, and potentially strong property rights, additional government support is required to pull technologies through this potential trap. Government support may come in the form of public funding programmes (such as the examples as outlined above), subsidies, supporting better access to finance, through regulations and standards.

3.3 Financing of investment

The previous section focussed on the role of technology in meeting environmental targets, and highlighted that the capital expenditure required to substitute new technologies for resource use may not be entirely met by the private sector without adequate government intervention. Various barriers to investment exist, and a combination of various policies is required to overcome these obstacles.

Cambridge Econometrics et al (2017c) highlights that returns to low-carbon investments are typically lower than the returns from fossil fuel projects, and that returns are slower to materialise. Finance for low-carbon projects is difficult to secure since these projects usually take a long time to develop and in the initial stages of operation the technology may operate at a loss. Profitability may only develop in the medium-term, ‘beyond the usual grace period provided by banks’. Furthermore, the technology may require substantial supporting infrastructure, which requires additional investment from investors, but also entails greater risk for investors. The same is true of investments in resource efficient technologies.

Cambridge Econometrics et al (2017c) identifies various barriers to investment that relate explicitly to the availability of finance. The barriers can be separated in to two distinct categories – barriers that arise from the nature of the low-carbon project, and barriers that arise from the decision-making framework of lenders. Table 4.1 below summarises the identified barriers to investment within each of these two categories.

Table 4.1 Barriers to the availability of finance

<table>
<thead>
<tr>
<th>Financial barriers typical for innovative clean energy technologies/projects</th>
<th>Specifics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lack of a clear market opportunity</td>
<td>Low-carbon R&amp;D is driven by environmental policy. For a market opportunity to exist, environmental policy must require action in the short-medium term.</td>
</tr>
<tr>
<td>R&amp;D uncertainty</td>
<td>As with all technology development, the development of low-carbon technologies is subject to the same uncertainties in terms of costs, and the time it will take to become a commercially-viable and profitable technology. Furthermore, in that time, market conditions and policy setting can change.</td>
</tr>
<tr>
<td>High capital costs and long-lived assets</td>
<td>Renewable technologies have relatively high up-front capital costs and low operating costs compared to conventional technologies, requiring long-term finance. Investment decisions may be biased towards conventional technologies.</td>
</tr>
<tr>
<td>Availability and volume</td>
<td>Financial instruments/investment products (e.g. green bonds), are not readily available for low-carbon technologies.</td>
</tr>
<tr>
<td>Risk-return related to regulatory (political)</td>
<td>Many types of clean energy investment opportunities often have lower returns and higher risk compared to carbon-intensive projects in more</td>
</tr>
</tbody>
</table>
stability established sectors. Low-carbon policy support is required such as carbon pricing or taxes or other intervention such as subsidies. Policy risk therefore becomes an issue since a change in future policy can lead to reduced returns for investors of low-carbon projects in future.

The high-risk nature of First-Of-A-Kind projects First-Of-A-Kind projects can be considered too risky since they involve unproven technologies. The risks of these projects are difficult to determine and quantify, and institutional investors and most commercial banks generally do not have the internal expertise to address these risks.

Lack of coordination and complementarity Complications arise from the lack of coordination and complementarity between financing instruments from the EU, Member States, and technology promoters.

Lack of financial and technical advice First-Of-A-Kind low-carbon energy projects are often too capital-intensive for venture capital investments and too risky for private equity financing. Furthermore, the lack of historical data in the sector prevents the insurance industry from designing products which could contribute to the de-risking of such projects.

Capital intensive Low-carbon technology projects are often small in nature, and transaction costs are higher when investing in smaller assets.

High transaction costs

Financial barriers arising from investors’ decision-making framework

<table>
<thead>
<tr>
<th>Barrier</th>
<th>Specifics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Time horizon of decision-making:</td>
<td>Most fund managers have a decision-making time horizon of maximum three years (and rely on the liquidity of their investments to adjust their portfolio to meet these short-term targets).</td>
</tr>
<tr>
<td>Lack of integration of climate risks into fiduciary duty and engagement practices:</td>
<td>The relationship between climate-related risks and benefits and institutional investors’ fiduciary duty is not clearly established. Additionally, assessments of investment managers’ engagement practices with companies do not sufficiently include climate-related concerns.</td>
</tr>
<tr>
<td>Lack of relevant climate-related risk and performance methodologies:</td>
<td>Current climate risks (physical impacts) and carbon risks (structural policy changes) face methodological shortcomings. The assessments done to date are not easily integrated into mainstream investment tools and practices. Investors currently cannot easily measure the climate and carbon performance of their portfolios.</td>
</tr>
</tbody>
</table>

Source: Cambridge Econometrics et al (2017c)

Different investors typically support different stages of the innovation chain.

4. Investment for basic research that develops a technology concept and its feasibility
5. Investments in first-of-a-kind low-carbon projects are required to prove feasibility
6. Investment brings clean technologies to the marketplace
7. Investment in proven technologies increase their market uptake
8. Investment in well-established, mature technologies

In stages 1 and 2, public finance is particularly important because of the high degree of uncertainty surrounding the outcome of early R&D, and the potential for spillovers in knowledge to occur. The risks involved in stage 3 of the innovation chain remain high, and only high-risk investors are likely to provide private funding. Only in stages 4 and 5, once low-carbon technologies reach commercial success will a range of investors be attracted to providing finance.

The impacts of financial constraints are not typically quantified in economic models. Cambridge Econometrics (2017) recognises that ‘It is important to represent the factors determining the availability of
finance explicitly in economic modelling’. Using a post-Keynesian model, it is assumed that money is created endogenously by commercial banks rather than controlled by central banks or constrained by the availability of savings, thus removing the constraints typically imposed by other models. The study specifies new investment equations within the E3ME model, increasing the costs of finance as cumulative investment in a sector go up. This reflects a declining interest from investors to finance projects within one sector since they become wary about being over-exposed to a specific set of risks. Early developers of low-carbon technologies may therefore receive more financing options, at lower costs, than those who enter the domain later on.

3.3.1 Policy considerations

Various policy considerations were detailed in the previous section related to overcoming the barriers for new technologies to enter the marketplace. Many of these barriers and the government intervention to overcome them are related to inadequate financing of new technology developments. However, the government interventions detailed in the previous section may not be enough to alter the behaviour of investors. Further policy considerations may be needed to address the specific financial barriers outlined in Table 4.1 above, through interventions in the credit market.

3.4 Crowding out and capacity constraints

Capacity constraints in the economy are an important consideration when designing policy; the effect of a given policy may be dependent on the state of the economy within business cycle. A wider discussion of capacity constraints is relevant, rather than focusing only ‘crowding out’\(^\text{72}\). Three key categories of supply constraints can be identified in this area of policy (Cambridge Econometrics and E3-Modelling, 2017a):

- financial capital markets
- labour markets
- product markets

Both the low-carbon and circular economy transitions necessitate a significant reallocation of resources in the economy (Cambridge Econometrics and E3-Modelling, 2017a):

- intertemporal, usually more up-front investments
- across sectors, with construction, engineering and waste management/recycling sectors usually benefiting
- between geographical regions, with fossil fuel and primary material producers and exporters losing out

Where policy increases demand in a market to the level of potential supply capacity, or beyond, displacement of resources will result. The typical mechanism of displacement is price effects.

Previous research has shown that capacity constraints in transitions are not insignificant. Cambridge Econometrics, EY, and SQ Consult (2017) examines capacity constraint sensitivities in the macroeconomic assessment of EU CO scenarios. Little impact is found in the least ambitious EU CO scenarios, where redirection of resources demanded is limited. In the ambitious EU CO40 scenario, however, GDP gains are 46% lower with partial constraints imposed. And positive employment results are 33% lower. IRENA (2016) examines the macroeconomic impacts of the increased deployment of renewable energy generation: under the assumption of no financial sector constraints, global GDP results are 0.6% higher than baseline in the REmap scenario\(^\text{73}\). Under full crowding out, no additional lending from the financial sector, GDP results of the transition are negative, though practically zero.

Environmental policy should therefore be designed considering current and anticipated future capacity constraints in the economy. Capacity constraints are of particular concern in cases where demand increases rapidly, when the private sector has little time to prepare. Effect of policy promoting the low carbon and

\(^{72}\) Crowding out generally refers to a specific case of capacity constraints: when an increase in public sector activity results in a reduction in private sector activity, particularly investment activity.

\(^{73}\) The REmap scenario is characterised by the global share of renewables doubling by 2030 compared to 2010, reaching 36% in total final energy consumption. See IRENA (2016) for more details.
circular economy transitions may be limited in effect by capacity constraints in key sectors, where the largest proportional redistribution of resources and economic activity is directed to.

Risks can be mitigated through engaging the private sector in policy decisions and through communicating timing of new policies. Assessment of future labour and production requirements of environmental policy is a key element of policy design. For example, circular economy objectives are likely to require a significant expansion of the domestic waste management and recycling sector: investment in production capacity and skills should be planned. Directing policy interventions across economic sectors, rather than focusing on a small number of sectors, is likely to minimise risk of product market, localised skills, and sectoral level financial capacity constraints.

The policy recommendations in this report should be considered in the context of the current European policy and economic environment, and its outlook. Given extant unemployment rates across Europe, it is unlikely that labour will be a capacity constraint at the macro level in the short term. Policy initiatives in green finance, crucially the European Regional Development Fund, alleviate the issue of financial capacity constraints in the transition: such funding mechanisms may ‘crowd-in’ private investment through signalling and confidence effects. Product market and sector-level skills capacity constraints, however, should be anticipated.
5 Conclusions

The key objectives of the 7th EAP are:

- to protect, conserve and enhance the Union’s natural capital
- to turn the Union into a resource-efficient, green, and competitive low-carbon economy
- to safeguard the Union’s citizens from environment-related pressures and risks to health and wellbeing

For the five sectors analysed in this study, a number of potential policy pathways have been identified that could reduce the environmental impact of production and use within each sector. These policies all contribute to achieving the broad objectives of the 7th EAP set out above.

We have also carried out economic and environmental analysis of the potential impacts of economy-wide resource efficiency; demonstrating the potential for beneficial socioeconomic and environmental outcomes as a result of the successful introduction of such policy. However, it is clear that the level of ambition, and the way in which measures are targeted, plays a major role in determining the economic impacts. The modelling suggests that a form of absolute decoupling of resource use and economic growth can be achieved in Europe with mild positive economic outcomes.

However, in this chapter, we have identified various cross-cutting challenges to the successful implementation of the policy pathways identified, and to achieving the environmental targets of the EU’s nationally determined contributions. In some cases, these challenges create the need for additional policy to overcome barriers to successful decarbonisation, such as improving accessibility to finance or providing more support to low-carbon innovation. In other cases, policy challenges are encountered because the policy leads to other effects within the economy, such as rebound effects or the reallocation of resources with sub-optimal GDP and employment outcomes.

The sector policies identified in this study all fall within three pillars of environmental policy response (Grubb 2014):

- Standards and engagement for smarter choices – includes regulations, standards, labelling and engagement to facilitate better, more environmentally-friendly choices.
- Markets and prices for cleaner products and processes – including carbon pricing and taxation plus subsidies.
- Strategic investment for innovation and infrastructure – price incentives only partly guide the market for innovation and infrastructure. Public intervention is required to encourage investment which looks beyond short-term returns.

For the five sectors considered in this study, the sector-specific policy pathways may not include elements of all three policy pillars. However, the policies should be combined with policy pathways aimed at the wider economy, addressing all three policy pillars and overcoming the potential challenges identified within this study.
## Appendix A Emissions and resource uses assessed in the cross-sectoral modelling

<table>
<thead>
<tr>
<th>Category</th>
<th>Emission or resource use type</th>
<th>Units</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>GHGs</td>
<td>CO₂ - combustion - air</td>
<td>th tonnes CO2eq</td>
<td>E3ME output</td>
</tr>
<tr>
<td>GHGs</td>
<td>CH₄ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>GHGs</td>
<td>N₂O - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>SOₓ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>NOₓ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>NH₃ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>CO - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>Benzo(a)pyrene - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>Benzo(b)fluoranthene - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>Benzo(k)fluoranthene - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>Indeno(123-cd)pyrene - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>PCBs - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>PCDD_F - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>HCB - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
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<td>NMVOC - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>PM₁₀ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>PM₂.₅ - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>TSP - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>As - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>Cd - combustion - air</td>
<td>kg</td>
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<tr>
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<td>Ni - combustion - air</td>
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<td>Pb - combustion - air</td>
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<td>Air pollution</td>
<td>Se - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>Zn - combustion - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Agglomeration plant - pellets - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Agglomeration plant - sinter - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Glass production - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Production of coke oven coke - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Production of gascoke - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Steel production: basic oxygen furnace - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
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<td>Air pollution</td>
<td>As - non combustion - Steel production: electric arc furnace - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>As - non combustion - Steel production: open hearth furnace - air</td>
<td>kg</td>
<td>EXIOBASE</td>
</tr>
<tr>
<td>Air pollution</td>
<td>B(a)P - non combustion - Primary aluminium production - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>B(a)P - non combustion - Production of coke oven coke - air</td>
<td>kg</td>
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</tr>
<tr>
<td>Air pollution</td>
<td>B(a)P - non combustion - Production of gascoke - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>B(b)F - non combustion - Primary aluminium production - air</td>
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<td>B(b)F - non combustion - Production of coke oven coke - air</td>
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<td>B(b)F - non combustion - Production of gas coke - air</td>
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<td>B(k)F - non combustion - Primary aluminium production - air</td>
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<td>B(k)F - non combustion - Production of coke oven coke - air</td>
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<td>Air pollution</td>
<td>B(k)F - non combustion - Production of gas coke - air</td>
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<tr>
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<td>CH4 - non combustion - Extraction/production of (natural) gas - air</td>
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<td>Air pollution</td>
<td>CH4 - non combustion - Extraction/production of crude oil - air</td>
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<tr>
<td>Air pollution</td>
<td>CH4 - non combustion - Mining of anthracite - air</td>
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<tr>
<td>Air pollution</td>
<td>CH4 - non combustion - Mining of bituminous coal - air</td>
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<td>Air pollution</td>
<td>CH4 - non combustion - Mining of coking coal - air</td>
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<td>Air pollution</td>
<td>CH4 - non combustion - Mining of lignite (brown coal) - air</td>
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<td>Air pollution</td>
<td>CH4 - non combustion - Mining of sub-bituminous coal - air</td>
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<tr>
<td>Air pollution</td>
<td>CH4 - non combustion - Oil refinery - air</td>
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<tr>
<td>Air pollution</td>
<td>CD - non combustion - Agglomeration plant - sinter - air</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Bricks production - air</td>
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<td>Air pollution</td>
<td>CO - non combustion - Carbon black production - air</td>
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<td>Air pollution</td>
<td>CO - non combustion - Cement production - air</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Chemical wood pulp dissolving grades - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>CO - non combustion - Chemical wood pulp soda and sulphate other than dissolving grades - air</td>
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<td>Air pollution</td>
<td>CO - non combustion - Chemical wood pulp sulphite other than dissolving grades - air</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Glass production - air</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Lime production - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Oil refinery - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Pig iron production blast furnace - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Primary aluminium production - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Production of coke oven coke - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Production of gas coke - air</td>
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<td>Air pollution</td>
<td>CO - non combustion - Semi-chemical wood pulp pulp of fibers other than wood - air</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Steel production: basic oxygen furnace - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>CO - non combustion - Steel production: electric arc furnace - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>CO2 - non combustion - Cement production - air</td>
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<td>Air pollution</td>
<td>CO2 - non combustion - Lime production - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Agglomeration plant - pellets - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>Cd - non combustion - Agglomeration plant - sinter - air</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Glass production - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Nickel unwrought - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>Cd - non combustion - Production of coke oven coke - air</td>
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<td>Air pollution</td>
<td>Cd - non combustion - Production of gas coke - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Refined copper; unwrought not alloyed - air</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Refined lead unwrought - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Steel production: basic oxygen furnace - air</td>
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<tr>
<td>Activity</td>
<td>Description</td>
<td>Unit</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Steel production: open hearth furnace - air</td>
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<tr>
<td>Air pollution</td>
<td>Cd - non combustion - Unrefined copper; copper anodes for electrolytic refining - air</td>
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<td>Air pollution</td>
<td>Cd - non combustion - Zinc unwrought not alloyed - air</td>
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<td>Air pollution</td>
<td>Cr - non combustion - Agglomeration plant - pellets - air</td>
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<td>Air pollution</td>
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<td>Air pollution</td>
<td>Cr - non combustion - Glass production - air</td>
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<tr>
<td>Air pollution</td>
<td>Cr - non combustion - Pig iron production blast furnace - air</td>
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<td>Air pollution</td>
<td>Cr - non combustion - Steel production: basic oxygen furnace - air</td>
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<td>Air pollution</td>
<td>Cr - non combustion - Steel production: electric arc furnace - air</td>
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<td>Air pollution</td>
<td>Cr - non combustion - Steel production: open hearth furnace - air</td>
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<tr>
<td>Air pollution</td>
<td>Cu - non combustion - Agglomeration plant - pellets - air</td>
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<td>Cu - non combustion - Agglomeration plant - sinter - air</td>
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<td>Air pollution</td>
<td>Cu - non combustion - Glass production - air</td>
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<td>Air pollution</td>
<td>Cu - non combustion - Pig iron production blast furnace - air</td>
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<td>Air pollution</td>
<td>Cu - non combustion - Steel production: basic oxygen furnace - air</td>
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<td>Air pollution</td>
<td>Cu - non combustion - Steel production: electric arc furnace - air</td>
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<td>Air pollution</td>
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<td>Air pollution</td>
<td>HCB - non combustion - Agglomeration plant - pellets - air</td>
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<td>HCB - non combustion - Agglomeration plant - sinter - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Agglomeration plant - pellets - air</td>
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<td>Air pollution</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Nickel unwrought - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Pig iron production blast furnace - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Production of coke oven coke - air</td>
<td>kg</td>
<td>EXIOBASE</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Production of gas coke - air</td>
<td>kg</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Refined copper; unwrought not alloyed - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Refined lead unwrought - air</td>
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<tr>
<td>Air pollution</td>
<td>HCB - non combustion - Secondary aluminium production - air</td>
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<td>Air pollution</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Steel production: electric arc furnace - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Steel production: open hearth furnace - air</td>
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<tr>
<td>Air pollution</td>
<td>Hg - non combustion - Steel production: transport and depots (used in mobile sources) - air</td>
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<tr>
<td>Air pollution</td>
<td>Indeno - non combustion - Primary aluminium production - air</td>
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<td>Air pollution</td>
<td>Indeno - non combustion - Production of coke oven coke - air</td>
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<td>Air pollution</td>
<td>NH3 - non combustion - N- Fertilizer production - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Beef and veal - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Coils coating (coating of aluminum and steel) - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Decorative paint applicatoin - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Degreasing - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Diesel distribution - transport and depots (used in mobile sources) - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Diesel distribution - transport and depots (used in stationary sources) - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Dry cleaning - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Extraction proc. and distribution of gaseous fuels - air</td>
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<td>NMVOC - non combustion - Extraction proc. and distribution of liquid fuels - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Extraction/production of (natural) gas - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Extraction/production of crude oil - air</td>
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<td>NMVOC - non combustion - Fat edible and non-edible oil extraction - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Fish dried salted or in brine; smoked fish; edible fish meal - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Fish fish fillets other fish meat and fish livers and roes frozen - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Fish otherwise prepared or preserved; caviar - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Flexography and rotogravure in packaging - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Gasoline distribution - service stations - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Gasoline distribution - transport and depots (used in mobile sources) - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Gasoline distribution - transport and depots (used in stationary sources) - air</td>
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<tr>
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<td>NMVOC - non combustion - Industrial application of adhesives (use of high performance solvent based adhesives) - air</td>
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<td>NMVOC - non combustion - Industrial application of adhesives (use of traditional solvent based adhesives) - air</td>
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<td>NMVOC - non combustion - Industrial paint application general industry (continuous processes) - air</td>
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<td>NMVOC - non combustion - Industrial paint application general industry (plastic parts) - air</td>
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<td>NMVOC - non combustion - Industrial paint application general industry - air</td>
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<td>NMVOC - non combustion - Inorganic chemical industry fertilizers and other - air</td>
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<td>NMVOC - non combustion - Leather coating - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Manufacture of automobiles - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Mutton and lamb - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Oil refinery - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Organic chemical industry - downstream units - air</td>
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<td>NMVOC - non combustion - Organic chemical industry storage - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Other industrial use of solvents - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Pharmaceutical industry - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Polystyrene processing - air</td>
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<td>NMVOC - non combustion - Polyvinylchloride produceduction by suspension process - air</td>
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<td>NMVOC - non combustion - Pork - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Poultry dressed - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Printing offset - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Products incorporating solvents - air</td>
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<tr>
<td>Air pollution</td>
<td>NMVOC - non combustion - Raw sugar - air</td>
<td>kg</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Rotogravure in publication - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Screen printing - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Steam cracking (ethylene and propylene production) - air</td>
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<td>NMVOC - non combustion - Synthetic rubber - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Tyre production - air</td>
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<td>Air pollution</td>
<td>NMVOC - non combustion - Vehicle refinishing - air</td>
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<td>SOx - non combustion - Production of gas coke - air</td>
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## Appendix B Concordance table between E3ME and EXIOBASE sector classifications

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<td>Pigs farming</td>
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<tr>
<td>Poultry farming</td>
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</tr>
<tr>
<td>Meat animals nec</td>
<td>Crops, animals, etc</td>
</tr>
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<td>Animal products nec</td>
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<tr>
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</tr>
<tr>
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<tr>
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Huijbregts, M.A.J., 1999b. Priority Assessment of Toxic Substances in the frame of LCA. Development and application of the multi-media fate, exposure and effect model USES-LCA.


Links between production, the environment and environmental policy

Analysis of environmental protection expenditures across Europe in key industries
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1 Introduction

Sustainable growth is one of the key and increasingly important values of the European Union. In the recent years, several action programmes, Directives and strategies were adopted to improve the quality of environment and to achieve a balanced development (including the 7th Environment Action Programme). Environmental Protection Expenditure (EPE) measures the economic resources devoted to prevention, reduction and elimination of pollution or any other degradation of the environment. Moving towards a sustainable development, it is important to see how the EU, its member states and industrial sectors invest in these activities.

This report analyses the Environmental Protection Expenditure (EPE) of EU28 countries using Eurostat Structural Business Statistics (SBS) data over time. It covers data on spending on environmental protection activities at a country and industry level, on an annual basis, from 1995. Some data sources report environmental protection expenditure decomposed into capital investment and operating expenses separately. Despite the considerable gaps in the country level time series it is possible to analyse aggregate expenditure trends. The main questions addressed in this report are:

- How did total EPE change over time in the EU28?
- How did the proportion of capital investments and operating expenses change within the total spending in the EU28?
- Did certain key industries have a different spending pattern over time?
- Did certain regions, EU15 and EU13 countries, show different spending patterns?
- How did the spending of the largest EPE spender countries evolve?

The rest of the report is structured in the following way: Section 2 Methodology explains the methodological framework of our analysis. The datasets, their key variables and the main analytical framework is explained. Section 3 Industry Case Studies presents Environmental Protection Expenditure patterns for selected key sectors of the economy. The same industries are chosen for analysis as in the rest of the Links between production, the environment and environmental policy report, to gain a full picture of the environmental protection efforts of these sectors. Expenditure patterns are examined separately for the EU28, 15, 13 and the largest spender countries. Whenever data availability allows, capital investments and operating expenses are separated, and their ratio is analysed. Data is handled at the highest sectoral disaggregation to focus on the relevant industries. To gain the fullest picture possible, the report analyses a combination of Eurostat data sources on the topic, harmonizing different time frames and variable classifications. (Figures showing the regionally disaggregated figures can be found in Appendix C Regional Disaggregation). Section 4 Total Economy Environmental Protection Expenditure presents economy-wide spending trends using the most recent national environmental protection spending data. The last section summarizes the report’s main conclusions.
2 Methodology

This chapter explains the methodology used throughout this report. First, it introduces the Eurostat datasets used. Second, it lists the key variables and their classifications. Third, it explains how the information present in the main data sources was analysed.

2.1 Data

For the sectoral EPE analysis, information was combined from two Eurostat datasets: Environmental protection expenditure - million euro (env_ac_exp1r2) and Environmental protection expenditure (NACE Rev. 2, B-E) (sbs_env_dom_r2). Both datasets contain EPE spending at a sectoral level, across all environmental protection activities for EU28 countries. The first data set reports capital and operating expenditure separately, but only covers the period 1995-2013. The second data set provides more recent information, from 2008-2016, but only reports capital expenditure. To gain the fullest picture on EPE patterns, information from both sources was used.\(^{74}\)

For the economy-wide EPE analysis National expenditure on environmental protection by institutional sector (env_ac_epneis) data set was used. This source covers the period 2006-2017 showing the most recent data on EPE. This data is not disaggregated by sector or expenditure type and has large gaps in its series. Thus, it is presented as an approximate picture on main and recent trends.

Additionally, data on sectoral gross value added (GVA) and overall economy GVA was used to show the relative importance of EPE: National accounts aggregates by industry (up to NACE A*64) (nama_10_a64).

<table>
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<th>Countries</th>
<th>Sectors</th>
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<td>Total, Capital Operating</td>
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<td>NACE-2</td>
<td>-</td>
<td>Current prices, million €</td>
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Table 2.1 Summary of Data Sources

\(^{74}\) Note: Production of environmental protection services of corporations other than specialist producers by economic characteristics and NACE Rev. 2 activity (env_ac_pepsnsp) was also considered as a source.\(^{32}\)

\(^{32}\) Note: \(\text{“sectoral EPE to total was not straightforward, therefore we decided not to use it.”}\)
2.2 Variables

This section describes the key variables across the different data sources:

- **Classification of Environmental Protection Activities (CEPA):** For most sectoral analysis the total spending across all environmental activities is used which aggregates environmental spending at different domains (TOTCEPA). For analysing waste and wastewater management, we show the whole industry’s spending on these specific activities: CEPA2 and CEPA3.

- **Industry (NACE_R2):** Both sectoral datasets are in principle disaggregated at the NACE-2 level and cover the whole industry sector. However, the first source groups some of the industries together and the second does not include all sectors which the first does. To combine information from both sources, most analysis uses the higher level of sectoral aggregation in the EPE spending based on the first source. Whenever the more disaggregated series can be integrated to the analysis we present them too. The sectors used in the industry case studies consistent with rest of the Links between production, the environment and environmental policy report:

  1. **Food, Drink and Tobacco:** C10-C12 Manufacture of food products; beverages and tobacco products
  2. **Rubber and Plastics:** C20-C22 Manufacture of chemical, pharmaceutical, rubber and plastic products and C22 Manufacture of rubber and plastic products.
  3. **Motor Vehicles:** C25-C33 Manufacture of metal products and other equipment and C29 Manufacture of motor vehicles, trailers and semi-trailers
  4. **Electricity, Gas, Water Supply & Treatment:** D35-E36 Electricity, gas, steam and air conditioning supply; water collection, treatment and supply and E36 Water collection, treatment and supply
  5. **Waste management and Sewerage:** CEPA2 Waste and CEPA3 Wastewater management expenditure across all industries present in the data (NACE B-E)
  6. **Manufacturing:** C Manufacturing

- **Total Spending, Capital Expenditure, Operating Expenses:**

  1. From the first sectoral dataset (env_ac_exp1r2), Environmental protection expenditure (EE1000) is used as total EPE. Total environmental investments (EE1100) is taken as capital expenditure and Total environmental current expenditure (EE1200) as operating expenses.
  2. In the second sectoral data set (sbs_env_dom_r2) there is no reported information on the operating expenses. The sum of Investment in equipment and plant for pollution control (V21110) and Investment in equipment and plant linked to cleaner technology (V21120) is used as a capital expenditure measure.
  3. In the economy wide dataset (env_ac_epneis) there is no separation of different expenditure types. Therefore, it is treated it as a total expenditure measure

- **Years:** The first sectoral dataset (env_ac_exp1r2) shows the period 1995-2013, the second (sbs_env_dom_r2) covers 2008-2016. The economy-wide dataset has data from 2006-17 (env_ac_epneis). However, all series have many missing values, gaps and the first/last years are especially affected.

- **Countries:** In principle all datasets should include all EU-28 countries. However, the second sectoral dataset (sbs_env_dom_r2) has no data on Luxembourg and Latvia. The economy-wide dataset (env_ac_epneis) does not include Greece, Cyprus, Hungary, Lithuania, Finland and Romania. Therefore, caution should be applied when aggregate series from different sources are compared. Note, that even if a country is included in the dataset, it may have gaps in the EPE series.

- **Unit of Measurement:** Millions of Euros, current prices.
2.3 Definitions and methods

This section sets out the methods used in the analysis to allow the most information to be extracted from the data, which has large gaps and potential reporting inconsistencies.

Adjusted total expenditure

Adjusted total expenditure is used instead of the raw total expenditure series reported in the Eurostat sources. It is formed as the sum of capital expenditure and operating expenses. It is created to use the most information the data provides (to fill-in missing values) and to correct for inconsistencies. Missing data is the most serious constraint in analysing the environmental protection expenditures. It becomes even more pressing as we focus on disaggregated data at country or industry level. Often, countries provide data only for capital expenditure or only for operating expenses. In such cases Eurostat reports the total expenditure data as missing. To form the adjusted total expenditure, the information on the non-missing expenditure part is taken as total expenditure. This is done to make use of country-year observations when at least some information is reported on the spending. The adjusted value also aims to correct cases when there is no missing data for a given country-year, yet the reported capital expenditure and operating expenses do not sum up to the total expenditure.\(^{75}\)

EPE as a percentage of Gross Value Added (GVA)

A normalised value of EPE is calculated as the aggregate EPE divided by aggregate gross value added (GVA) and multiplied by 100 (i.e. to calculate a percentage), for EU28 (the whole of the EU), EU15 (western Europe) or EU13 (central and eastern Europe) countries. This normalised measure of EPE is important, as interpreting trends based on a nominal EPE spending series with many missing values is difficult for a number of reasons. First, changes in the total expenditure can be driven by actual changes in countries’ spending but also by the number of countries who report their EPE spending each year. Missing EPE of large economies such as Germany or the UK can move the total spending considerably. Second, it is difficult to sense the magnitude of the EPE presented in millions of euros without a reference point. Third, spending measured in current prices can show different trends and patterns than real spending, depending on underlying inflation. The normalised EPE helps overcoming these difficulties. It illustrates the relative importance of EPE and adjusts to account for non-reported spending. This is achieved as only the GVA of those countries who provided spending data for each EPE category is included in the calculation.\(^{76}\) A ratio of two series, both of which are measured in current prices is also free from inflationary effects.

Opex/Capex Ratios

The ratio of operating expenses and capital expenditure within total EPE, provides insight into expenditure structures, and has been used in previous analyses (such as TME (2015)). This ratio can only be formed if both capital and operating expenditure data is available for a given country-year. Thus, it suffers from missing data problems more than the previous measures do, especially at the country level.\(^{77}\)

\(^{75}\) To illustrate the magnitude of adjustment, take the Food, Drink and Tobacco sector: from the total 532 country-year observations (EU28, 1995-2013), 59% of the data points is missing completely (313 observations). Additional 5% of the country-year observations has missing value only the capital or only operating expenses (24 obs.). In two country-years the two expenditure parts sum up to different value than the total expenditure is. In the latter two cases, the adjusted expenses will be different from the raw total expenditure data and provides us with more information.

\(^{76}\) Note that this measure provides a weighted average of the EPE spending, where weights depend on the GVA size, or the industry GVA size of a given country.

\(^{77}\) The EPE series of the largest spenders was filled-in with estimated data in the following way: In cases of one-year long data gaps, gaps are filled with the average of the values in the years before and after the gap. This gap-filling is a conservative smoothing of the data yet provides cleaner looking graphs in which trends can be inferred more easily.
2.4 Missing Data

The difficulty of analysing current trends in EPE lies in the extent of missing data patterns. Table 2.2, Table 2.3 and Table 2.4 show the missing information based on the three different data sources used in this study. The main conclusions from the tables are the following:

- **Expenditure parts:** Across the three data sources used in this study, which are the most extensive data series available on environmental protection expenditure, only one (env_ac_exp1r2) contains data for all expenditure parts (total, capital, operating) reported at an industry level. The other two series only include capital expenses (sbs_env_dom_r2) and total expenditure (env_ac_epneis). Table 2.2 shows that even the source covering different expenditure types often has missing data for one of the parts or both.

- **Time frame:** The source data which covers all expenditure parts (env_ac_exp1r2) was discontinued in 2013, thus only the other two sources (sbs_env_dom_r2, env_ac_epneis) could provide more recent insights. However, Table 2.2 and 2.4 show that data for the start and end years of the series have frequent missing values. By comparing the overlapping period of different sources, the last years of Table 2.2 and the first years of Table 2.4, it is visible that data is not always missing for the same years in the same countries across sources. This is due to the difference in methodologies for the original data gathering.

- **Country coverage:** Table 2.2, 2.3 and Table 2.4 show that all datasets have countries with completely missing time series and many of the non-missing country series have considerable gaps. The missing countries are not uniformly missing across the different sources in given years, again due to data processing methodology. From the 2000s, there is less missing data in Table 2.2 for EU13 countries, while EU15 member states have more stable data provision. This pattern influences the conclusions drawn for the two groups based on the whole time frame.

- **Industry coverage:** Only the first two datasets (env_ac_exp1r2, sbs_env_dom_r2) have data on industries where the EPE expenditure is realized. However, across the six industries analysed (see Section 2.2) data is missing for different industries in different countries. This pattern depends on the sectoral aggregations used to collect data on their EPE spending. Sector specific missing values are illustrated more clearly in the graphs in the next chapters.

- **Missing parts, missing total:** The source data, which covers all expenditure parts (env_ac_exp1r2) is not reported if either the capital expenditure or the operating expenses are missing for a given country year. Therefore, when summed up to EU28, EU15 or EU 13 level, the total series could be lower than its parts if data is missing for a large economy. Replacing the total EPE with one part when the other is missing, would bias the total measure and omitting country years due to partially missing information with an already scarce dataset is wasteful. Therefore, all country-year observations are used in the graphs presented in this report when there is any information on Capex or Opex. However, in the normalized graphs the incomplete total expenditure is not shown to avoid confusion.

Overall, this section aims to highlight the extent and dimensions of the missing value problem in the data set used. The prevalence of missing information affects the strength of the conclusions and suggests using caution when interpreting results.
Table 2.2 Missing data in the Eurostat source: env_ac_exp1r2. T, C, O reflects if data is missing for at least half of the six industries analysed in the given country years for Total, Capital and Operating Environmental Protection Expenditure.

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Table 2.3 Missing data the Eurostat source: sbs_env_dom_r2. The percentage of missing Capital Expenditure data for the six industries analysed across all country years.

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3 Total Economy EPE

This chapter seeks to provide insights on recent economy-wide Environmental Protection Expenditure trends. Sectoral data sets do not include all sectors of the economy; thus, their use is limited for analysing economy-level trends. For this purpose, the National expenditure on environmental protection by institutional sector (env_ac_epneis) data set is used. This source covers the period 2006-17, showing the most recent data on national EPE spending. This data is not disaggregated by sector or expenditure type (capital/operating) and has large gaps in its time series. It is presented to give an approximate picture on main and recent aggregate trends.

Figure 3.1 illustrates EPE expenditure for the EU28, EU15 and EU13 over time. These graphs show a very clear pattern: aggregate environmental protection expenditure increases with more extensive data coverage over time. This pattern shows the need of normalisation of the data. The figures also show that aggregate EU15 expenditure is still significantly larger in magnitude than EU13 spending (EU15 and EU13 graphs can be found in Appendix B).

Country-specific series can be found in Appendix B. The largest EPE spenders are the countries with the largest GVA in the sample.

Figure 3.1 National Environmental Protection Expenditure of EU28 countries

![Graph showing national expenditure over time](image)

Figure 3.2 shows the national expenditure normalised by economy-wide GVA. This illustrates that the increasing trends of the previous figures were driven primarily by missing data patterns, and by increases in the GVA for reporting countries. The normalised expenditure series exhibit stability over time. For EU28 and EU15 countries the EPE is approximately 2.0-2.2% of overall GVA. For the EU13 it is between 2.1-2.5%.

78 Note that data is not presented for 2016 and 2017, as for 2016 only Spain and for 2017 no country has reported EPE data.
Overall, the economy-wide expenditure data reveals that for EU28 and EU15 countries the EPE is between 2.0-2.2% of overall GVA. For the EU13 it is in the range of 2.1-2.5%. The aggregate nominal spending patterns are heavily influenced by missing data, while normalised series show a high stability in EPE relative to GVA.
4 Industry Case Studies

4.1 Introduction

This section presents Environmental Protection Expenditures (EPE) for selected industries. The same sectors are analysed as in the rest of the *Links between production, the environment and environmental policy* analysis, to gain a comparable picture of their environmental protection efforts. EPE investment in these sectors is crucial to move EU’s production towards a more sustainable path.

For each sector a series of data are presented on EU28 EPE from different perspectives. Aggregating across countries provides insight into EU-wide trends and helps to overcome the gaps in country-level series. For each sector, three graphs are presented:

- **Environmental Protection Expenditure, Capital Expenditure and Operating Expenses for EU28** - this presents nominal EPE for the EU28 measured in million Euros, current prices. It shows total adjusted, capital and operating EPE for 1995-2013 based on the first sectoral data set. To extend the coverage recent data on capital expenditure for 2008-2016 is added from the second sectoral data set, with higher sectoral aggregation whenever possible. This graph illustrates the EPE which the raw data shows. However, as missing data influences is a strong driver of the fluctuations the pattern on this figure could be misleading. To illustrate this, the number of missing country-years is shown in the background of the chart (and measured on the right-hand axis).

- **Capital Expenditure and Operating Expenses, Normalised with Industry Gross Value Added (GVA) of Countries Providing Expenditure Data for EU28** - This shows GVA-normalised expenditure by type. It shows the relative importance of EPE and takes better account of missing data and inflation effects.

- **Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time for EU28, EU15 and EU13**. This graph shows the ratio of operating and capital expenses to illustrate expenditure structure of sectors, whether they invest in new capital projects or spend more on operating existing production in a cleaner way. This graph uses information only on countries who provide data for both expenditure parts each year.

These three types of data complement each other and show how insights on the main trends can be gained from the raw data by different adjustments.

Further data are presented in the appendix: regionally disaggregated graphs for EU15 and EU13, and EPE series for the largest spender countries. For all industries the same set of graphs are presented.

- Regionally disaggregated graphs of EPE trends to EU15 and EU13 spending (*Appendix B Regional Disaggregation Figure B.i.1.-4.*) have the same conclusion for all industries, that aggregate EPE is dominated by EU15 expenditures, with EU13 catching up in both spending and data coverage.

- **Figures showing the EPE and Opex/Capex ratios for the largest spender countries** (*Appendix B Regional Disaggregation Figure B.i.5.-7.*) show the same: usually countries with the largest sectoral GVA are responsible for most of the nominal EPE spending in each sector. This pattern mostly breaks down if the large producers fail to provide EPE data. In a small number of cases some countries spend more on EPE than it would be expected given their industry GVA. The country-level Opex/Capex ratios are presented in the appendix as in most cases their time series have too many missing values to provide clear insights.

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79 *All figures are based on the abovementioned Eurostat datasets and own calculations.*
4.2 Manufacturing

Analysing the aggregate Manufacturing sector gives us insights about EPE in the broad industry sector. Two of the EPE datasets used do not contain any data for the economy as a whole. In these cases, manufacturing provides the broadest sectoral coverage.

Figure 4.1 shows nominal EPE spending in the manufacturing sector for EU28. In the 2000s, when the data coverage is the most extensive, EU28 countries have stable EPE expenditure driven by the operating expenses and the spending of EU15 Member States. The capital expenditure series based on the 2008-2016 data suggests that the expenditure drop in the 2010s is driven by missing observations, as there is no sign of decrease in the capital expenditure based on the more complete data.

Country-level series show the high weight of the largest spenders in the total EPE. For 1995-2013, France and the Netherlands are among the economies with the largest GVA, but their EPE data is often missing. Over 2008-2016, the list of largest spenders corresponds to the list of economies with the highest GVA in this period, except for the UK which does not report EPE data after 2008.

Figure 4.1 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU28: Manufacturing

Figure 4.2 shows the GVA normalised expenditure in the Manufacturing sector for the EU28. The normalised series show a stable pattern. Capital expenditure is between 1.5-2%, while operating expenses are 0.4-0.6% of GVA in the manufacturing sector in both EU28 and EU15. The EU13 series exhibits some volatility in the beginning of the timeframe when data coverage is not very extensive. Then, the relative expenditures stabilize at similar values as EU15 has, with slightly higher normalised operating expenses (0.5-0.6%).
Figure 4.3 shows the Opex/Capex ratios of the EU28, 15 and 13 countries. The ratios show some stability when data availability is maintained; however, their pattern correlates with the number of observation and has considerable fluctuations. The country-specific series show a mixed pattern: countries with highly stable Opex/Capex ratios and large outliers.

Figure 4.3 Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time: Manufacturing

Overall, the Manufacturing sector has relatively stable nominal and normalised EPE, when the data coverage is extensive enough to draw conclusions. Capital expenditure is between 1.5-2%, while operating expenses is 0.4-0.6% of the GVA in EU28 and EU15 countries. Opex/Capex ratios and country-level expenditures show too much volatility for strong conclusions.
4.3 Manufacturing of food products, beverages and tobacco products

The nominal EPE of the Manufacturing of food products, beverages and tobacco products industry for EU28 countries exhibit an increasing trend over time (see Figure 4.4). However, this pattern is mostly driven by the increasing frequency of data provision by the member states. The number of missing country-year observations is the lowest when aggregate spending reaches its maximum, almost 3.2 billion euros. This rise is driven by increasing operating expenses which accompany a steady capital expenditure spending. The decomposition of the EPE to EU15 and EU13 spending show the dominance of the EU15, with EU13 gaining data coverage and increasing expenditure (see Appendix B for more detail).

Figure 4.4 include recent capital expenditure series using the second sectoral dataset. The two capital expenditure series differ quite significantly for the overlapping period. However, this is caused by different missing data patterns. While the first data set has high number of missing values for its last years, the second data set has relatively strong coverage. Their difference clearly illustrates the extent to which spending drops in the graphs can be caused by non-reporting of country level data.

Country-level series for the largest EPE spenders can be found in Appendix B. EPE expenditures show a strong correlation with industry size, but missing expenditure data determines largely whether given countries are among the largest five spenders. For example, Germany has the second largest total GVA for 1995-2013, but due to reporting EPE only for three years is not among the largest five spenders.

Figure 4.4 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU28: Manufacture of food products; beverages and tobacco products

Figure 4.5 shows the capital and operating expenditure of EU28 normalised with the sectoral GVA. This reduces fluctuations from Figure 4.4 which were caused by missing data. This normalisation also filters out the impact of changing price levels. As expected, the normalised series shows a smoother picture than the nominal expenditure series. By the 2000s as the data coverage is getting better the relative spending ratios stabilize. In the early years, too few countries provide data and their reported values move the entire series. Normalised capital expenditure stabilizes around 1.5-2%, and operating expenses around 0.25-0.75% of industry GVA for the EU28. Note that the sectoral GVA exhibited a steady increase over the observed period. The normalised expenditure series for EU15 and EU13 show similar values and stability (see Appendix B). It is noticeable how much of the fluctuation shown in Figure 4.4 is smoothed with the
normalisation. This demonstrates how missing data patterns can move the nominal spending series and how much caution is needed to interpret its trends.

Figure 4.5 Capital Expenditure and Operating Expenses of EU28, Normalised with Industry GVA of Countries Providing Expenditure Data: Manufacture of food products; beverages and tobacco products

Figure 4.6 shows the Opex/Capex ratios, calculated based on those country-years when both values are reported for EU28, EU15 and EU13. The series shows very large fluctuations, which makes it difficult to say anything about long term trends. Operating expenses are consistently larger and drive most of the fluctuation. The Opex/Capex series of the largest spenders shows a mixed pattern.
Overall, the manufacture of food products; beverages and tobacco products industry shows mostly stable capital and operating expenditures relative to industry gross value added over time. The normalised capital expenditure stabilizes around 1.5-2%, operating expenses around 0.25-0.75% of the industry GVA for the EU28. Missing data influences data patterns in other graphs too much to draw any stronger conclusions.
4.4 Manufacturing of chemical, pharmaceutical, rubber and plastic products

Figure 4.7 shows the nominal EPE in the manufacturing of chemical, pharmaceutical, rubber and plastic products sector for the EU28. The series does not exhibit clear trends, but has large peaks related to increases in the number of countries with reported spending. The aggregate EPE is distributed between the EU15 and EU13. As expected, EU15 drives the aggregate spending patterns with EU13 achieving higher coverage and importance over time (see Appendix B for more detail).

Figure 4.7 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU28: Manufacture of chemical, pharmaceutical, rubber and plastic products

Capital expenditure data is shown in all graphs from the second sectoral data set. As this data allows a higher sectoral disaggregation, expenditure is plotted for the Rubber and Plastics sector separately (to stay as consistent with other parts of the study as much as possible). This shows that the majority of the capital investment in the more aggregated sector comes from the chemical and pharmaceutical industries.

The country-level expenditure figures (charts are available in Appendix B) show that the largest spenders are those with the highest GVA. Germany has large reported spending but provides data only for three years in the first data set. UK is missing from the largest spenders over 2008-2016 as a whole, as it provides data only for a single year. These illustrate clearly how much missing data influences nominal spending trends.

Figure 4.8 shows the GVA normalised EPE series for EU28 countries. For the period 2000-2010, when data coverage is most extensive, the relative expenditure series become fairly stable. The normalised operating expense ratios are in the range 1.5-3%, while the capital expenses are 0.5-1% of the industry GVA. The ratios are slightly higher in the EU13 countries and less stable. The disaggregated Rubber and Plastics sector seems to invest less in environmental protection capital expenditure relative to its own sectoral GVA than the more aggregate sector does. This is especially true for the EU13, where chemicals and pharmaceuticals industry seem to reinvest about five times more in EPE activities than the rubber and plastics sector, relative to their GVA (see Appendix B for more details).
Figure 4.8 Capital Expenditure and Operating Expenses of EU28, Normalised with Industry GVA of Countries Providing Expenditure Data: Manufacture of chemical, pharmaceutical, rubber and plastic products

Figure 4.9 illustrates the Opex/Capex ratios for EU28, 15 and 13 countries. It is difficult to see clear trends due to the fluctuations. Country level trends are also too noisy to infer more than Opex’s dominance and substantial volatility (see Appendix B Regional Disaggregation (Figure B.3.7.)).

Figure 4.9 Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time: Manufacture of chemical, pharmaceutical, rubber and plastic products

Overall, for the Manufacturing of chemical, pharmaceutical, rubber and plastic products sector the normalised EPE ratios seem to be stable in the observed period. For the EU28, normalised operating expenses are in the range 1.5-3%, while the capital expenses are 0.5-1% of the industry GVA. The ratios are higher for the chemical and pharmaceutical sectors than for plastics. Due to the noise and missing value problem, nominal spending trends and Opex/Capex ratio trends should be assessed with caution.
4.5 Manufacturing of metal products and other equipment

Figure 4.10 shows total EPE for the manufacture of metal products and other equipment sector. As with the previous industries, this raw spending series exhibits an increasing trend, but the increase correlates with a larger number of countries providing data on EPE over time. The EU15 dominates nominal spending ahead of the EU13, although the gap narrows over time as more EU13 data is available (see Appendix B for more details).

To all three figures below, capital expenditure data on the sector is shown along with the limited data on spending from the manufacture of motor vehicles, trailers and semi-trailers sector industry. This shows that the non-motor vehicle components have higher capital expenditure in nominal terms.

The country-level series, again, illustrate the magnitude of missing data. In the 1995-2013 series, Germany, that has the largest metals, and motor vehicle industry of any Member State, only provided EPE data for two years. Its EPE for those two years accounts for almost half of the total expenditure by all EU28. It is not clear how much Germany spent on EPE in the years missing from the data, but it is possible that knowing that information would change the sectoral expenditure graphs significantly. France and Italy are also among the countries with the highest sectoral GVA, but they scarcely report EPE expenditures. Sweden and Poland on the other hand have almost complete EPE time series, thus they appear as large spenders on the graph. Overall, this observation serves as reminder to cautiously interpret EPE patterns, as missing data can distort the conclusions significantly.

Figure 4.8 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU28: Manufacture of metal products and other equipment
Figure 4.11 shows the normalised EPE series. For the EU28, operating expenses are in the range 0.4-0.8%, while capital costs stay around 0.1-0.3% of the industry GVA. These values are more volatile and lower than in most of the other sectors studied. This suggests that the metal sector places less weight on mitigating environmental damages, relative to its production added value. The analysis show that the vehicles industry spends more on EPE relative to its sectoral GVA than the rest of the metal sector does. EU15 and EU13 graphs show similar trends, with slightly higher ratios across the EU13 countries.

Figure 4.12 presents the Opex/Capex ratios of the EU28, EU15 and EU13 countries. All series exhibit an
upward trend, with large fluctuations and increasing data coverage over time. Thus, it is difficult to
determine whether this is simply a function of the inclusion (or otherwise) of specific countries. Country-
level series also exhibit slight upward trends, but the data is very noisy. (see Appendix B for more detail).

Overall, the manufacture of metal products and other equipment sector has a stable normalised
expenditure, which is lower than the value for the previous two industries. Operating expenses are in the
range 0.4-0.8%, while capital costs hover around 0.1-0.3% of the industry GVA of EU28 countries. The
Opex/Capex ratios seem to exhibit a slight upward trend, yet the data has too many gaps to infer
anything stronger.
4.6 Electricity, Gas, Water Supply & Treatment

Figure 4.13 shows the nominal EPE spending of the electricity, gas, water supply & treatment sector for the EU28 countries. The data shows clear growth in expenditure until 2010-2011, with a relatively good data coverage. Both the EU15 and EU13 exhibit strong growth in spending. Compared to other industries assessed, EPE in the EU13 plays a larger role.

The 2008-2016 capital expenditure series includes more detailed data on water supply and treatment. The expenditure series show that electricity and gas sectors have higher spending shares in total broad sector EPE.

Figure 4.11 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU28: Electricity, gas, steam and air conditioning supply; water collection, treatment and supply

Figure 4.14 shows the GVA normalised expenditure series for the sector. The ratios are highly stable over the observed period. The operating expenses are between 3.5-4% of the industry GVA, while capital expenses range between 1.5-2%. These ratios are higher than in any other of the previous industries suggesting that environmental protection activities are higher priorities for the energy and water sectors. Operating and capital expenses also seem to move more closely together in magnitude than in other sectors.

The normalised graphs reveal unusual trends in the 2008-2016 capital expenditure series, especially in the water treatment and supply (E36) sector in EU13 countries (see Appendix B for more details). High capital expenditure over GVA ratios are driven by the reported expenditure of specific countries. For example, Romania reports higher EPE spending than its industry GVA for several years, which is the highest EPE spending series in the sector following Germany. Its high spending implies that there could be a difference in definitions across countries in what classifies as and is reported as EPE. For the water sector this could be more problematic than in others, as a large part of general investment in the industry can be broadly classified as EPE. As there is no available information to learn how large fraction of the data is affected by this inconsistency, conclusions for the water sector based on the 2008-2016 data should be drawn with great caution.
The largest spenders at Member State level differs substantially from the list of countries with the highest sectoral GVA. Over 1995-2013, France and Spain have higher sectoral GVA than Poland and the Czech Republic. However, the latter two countries have more complete reported EPE series and higher values. As outlined above, there is a chance that different countries classify different expenditures as EPE.

Figure 4.12 Capital Expenditure and Operating Expenses of EU28, Normalised with Industry GVA of Countries Providing Expenditure Data: Electricity, gas, steam and air conditioning supply; water collection, treatment and supply

Figure 4.15 shows the Opex/Capex ratios for the EU28, EU15 and EU13 countries. The ratios seem to exhibit some convergence between EU15 and EU13 countries, with a slight decrease for the first group and a stable ratio for the second group of countries. However, the data is very noisy, and country specific series show a very mixed pattern (Appendix B Regional Disaggregation (Figure B.5.7)).
Overall, Electricity, gas, steam and air conditioning supply; water collection, treatment and supply sector shows relatively stable environmental protection expenditure relative to GVA. The operating expenses are between 3.5-4% of the industry GVA, while capital expenses range between 1.5-2%. These percentages are slightly higher than in other industries, particularly in the EU13 countries. However, some unusually high spending patterns suggest that there may be consistency issues in the underlying data. Therefore, conclusions drawn for this sector must be treated with caution.

Figure 4.13 Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time: Electricity, gas, steam and air conditioning supply; water collection, treatment and supply
Waste and Wastewater management

Figure 4.16 shows the EPE on Waste and Wastewater management. The graphs show the expenditure of all industries present in the data on this specific environmental protection activity. Over 2000-2010, when data coverage is best, the spending of EU28 seems to be stable and is dominated by the operating expenses. The aggregate values are driven by the EU15 member states, with EU13 catching up both in terms of magnitude and data coverage. The capital expenditure series based on the 2008-2016 data suggests that the expenditure drop in the 2010s is likely to be driven by missing observations (see Appendix B for more details).

Country-level series show that total expenditures are dominated by a few countries. As in other sectors, the list of countries presented as largest spenders is largely affected by missing data. For 1995-2013 France and Italy both have high GVA in the sectors analysed, but they do not report EPE data for this activity (Italy) or report it for only a couple of years (France). Poland and Austria on the other hand have long and consistent EPE series for the observed period. For 2008-2016 UK and Spain are among the countries with the largest sectoral GVA, yet they are not among the largest EPE spenders. The UK only reports data for a single year and Spain has relatively low expenditure in waste and wastewater management despite the size of the domestic industry.

Figure 4.17 presents the normalised EPE, which show is relatively stable over time. The normalised capital expenditure is between 0.8-1.2% and the operating expenses is between 0.2 and 0.4% for both EU28 and EU15. In the EU13 the data is more volatile and slightly higher ratios 1.2-1.6% and 0.3-0.5% are seen in the 2000s, when the data coverage is the best.
Figure 4.15 Capital Expenditure and Operating Expenses of EU28, Normalised with Industry GVA of Countries Providing Expenditure Data: Waste and Wastewater Management

Figure 4.16 Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time: Waste and Wastewater Management

Figure 4.18 shows the Opex/Capex ratios for EU28, EU15 and EU13 countries. The ratios seem to be more stable than for most other industries. In the EU28 and EU15 the series move between 3-4 in the 2000s, when coverage is the best. Similarly, the Opex/Capex ratio of the EU13 stabilize around 3. This graph suggests that there could be a stable ratio between the two expenditure parts in Waste and Wastewater.
EPE. Country specific ratios show a mixed pattern. However, for the countries with more complete EPE series there seem to be some stability in the Opex/Capex ratios.

Overall, industry spending on Waste and Wastewater Management environment protection activities seem to be stable over time relative to industry GVA. The normalised capital expenditure is between 0.8-1.2% and the operating expenses is between 0.2 and 0.4% for both EU28 and EU15. The Opex/Capex ratios show higher stability than in other industries even at a country level.
5 Conclusions

The aim of this report was to analyse the Environmental Protection Expenditure (EPE) of EU Member States over time. EPE measures the economic resources countries and sectors devote to prevention, reduction and elimination of pollution or any other degradation of the environment. Therefore, knowing the main trends in EPE helps to understand whether countries and sectors invest enough in environmental protection to achieve environmental sustainability.

This report analysed EPE for key industries used in the Links between production, the environment and environmental policy report. Capital expenditure and operating expenses are analysed separately at an EU28, EU15, EU13 and country level. Data is presented from 1995-2016. No available data set has information for the whole time period at the level of disaggregation needed for this analysis. Thus, different Eurostat data sources were combined. The main conclusions from this report are the following:

- It is difficult to draw strong conclusions based on the available EPE data as country-level time series have considerably gaps and aggregation can heighten data inconsistencies across countries. Reported country-level EPE series have so many missing observations that it requires heavy interpolation to infer spending trends. On the other hand, aggregate analysis often suffers from inconsistencies caused by different EPE definitions and reporting practices used by member states.

- From the nominal aggregate EPE series it can be inferred that EU15 countries still dominate EU28 spending, but EU13 is catching up fast in the recent years. Most of the visible trends and fluctuations in the nominal spending graphs can be explained by missing observations.

- Capital and operating expenses are presented as percent of GVA for the sectoral analysis. This normalisation mitigates the fluctuations caused by missing data and inflation and show the relative magnitude of EPE spending. The normalised EPE series are much more stable over time, suggesting that sectoral EPE spending relative to industry performance has not changed much over the observed period. Operating expenses are approximately 0.8-3.0%, while capital expenditure is about 0.1-0.8% of industry GVA in the sectors analysed for EU28 countries in the 2000s, when data coverage is the most extensive. Naturally, there are sectoral differences with the metal products and other equipment sector showing lower, and the electricity, gas, water supply & treatment sector exhibiting higher normalised expenditures.

- Sectoral OPEX/CAPEX ratios exhibit large fluctuations ranging approximately between 2-6 in the 2000s for most industries. However, country-level series are so noisy that aggregate trends should be interpreted with great caution.
Appendix A References


Appendix B Regional Disaggregation

This appendix shows regionally disaggregated graphs for EU15 and EU13 regions, and EPE series for the largest spender countries. For all industries the same seven graphs are presented:

*Environmental Protection Expenditure, Capital Expenditure and Operating Expenses for EU15 and EU13:* This presents nominal EPE expenditure for the EU15 and EU13 measured in million Euros, current prices. It shows total adjusted, capital and operating EPE for 1995-2013 based on the first sectoral data set. To extend the coverage recent data on capital expenditure for 2008-2016 is added from the second sectoral data set, with higher sectoral aggregation whenever possible.

*Capital Expenditure and Operating Expenses, Normalised with Industry Gross Value Added (GVA) of Countries Providing Expenditure Data for EU15 and EU13:* This shows the GVA-normalised expenditure series by type. It shows the relative importance of EPE and cleans the pattern of the first graph from missing data and inflation effects.

*Total Environmental Expenditure of the 5 Largest EPE Spender Countries for 1995-2013, 2008-2016:* These graphs select the largest spender countries based on the nominal spending in the two data sets over the timeframe, based on total adjusted spending in the first and based on total capital expenditure in the second series. These graphs provide information on country-specific trends for those member states who determine EU-wide spending patterns the most.

*Operating Expenditure to Capital Expenditure - OPEX/CAPEX ratios over time for the 5 largest EPE spenders.* This presents how the two expenditure parts, capital investments and operating costs, are divided within the total expenditure for the largest spenders.
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Figure B.1 National Environmental Protection Expenditure of EU15 countries

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Note: Adjusted country-years: United Kingdom: 1998
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Figure B.11 Environmental Protection Expenditure, Capital Expenditure and Operating Expenses of EU15: Manufacture of food products; beverages and tobacco products

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B.4 Manufacturing of chemical, pharmaceutical, rubber and plastic products

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Note: Adjusted country-years: Austria 2001
Links between production, the environment and environmental policy

Food, drink and tobacco – sector study
This report is an Annex to the Final Report *Links between production, the environment and environmental policy*, ordered and paid for by the European Commission, Directorate-General for Environment, Contract ENV.F.1/FRA/2014/0063. The information and views set out in this study are those of the authors and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

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1 Introduction

The food and drinks industry is the largest manufacturing sector in Europe, in terms of both output and employment – it has an annual turnover of almost €1,100 billion (FoodDrinkEurope, 2017) and employs over 4 million people. It sells 90% of its produce within the single market (FoodDrinkEurope, 2017). It is a heterogeneous sector consisting of a large number of small family-based companies operating alongside global food conglomerates (EEA, 2017b). Although the sector’s contribution to the EU gross value added is relatively small (1.7%) (FoodDrinkEurope, 2017), it has a fundamental social and cultural importance in many European regions, and is the main source of income for some local communities (EEA, 2014, 2017b).

Tobacco product manufacturing is by contrast a small sector in Europe. In 2010, 261 tobacco product manufacturing enterprises were operating in the EU, employing just over 42,000 people (0.2% of the manufacturing workforce) and generating almost €7 billion of value added (0.4% of the EU manufacturing total)80. Over 527 million cigarettes were produced in the EU in 2016, with Germany being the top producer. Production has gone down over time, with about a third less than ten years ago81.

The environmental impacts of individual agricultural products vary substantially, depending upon the specific requirements of production/product growth. As such, it is not possible to explicitly quantify all links between the sector and the environment in a detailed fashion. In this analysis, we have focussed on two areas which are viewed as having amongst the highest potential for reducing environmental impacts through policy; the production of meat, and sugar products (note that the study excludes seafood).

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2 Environmental footprint of the sector

2.1 Research questions

- How does this sector benefit from the environment?
- How does it affect the environment?
- What is its resource use?
- How does environmental policy affect these links?
- What are the links between these sectors and the Sustainable Development Goals and the different targets?

2.2 Overview

While the European food, drink and tobacco sector is in various ways dependent on the quality of the environment – on the whole – it is a significant source of environmental degradation and sustainability challenges generated throughout the economy. One important example is the sector’s energy use. According to the EEA, the amount of energy necessary to cultivate, process, pack and bring food to our tables accounted for 17% of the EU’s gross energy consumption in 2013, equivalent to about 26% of the EU’s final energy consumption that same year. Another example is the sector’s significant leakage of nutrients, particularly during agricultural production. Of the total input of nitrogen and phosphorus fertilisers, only 20-30% is actually embedded in the food that reaches consumers plates (EEA, 2017b).

Each step in the value chain of food, drinks and tobacco (FDT) has some impact on the environment. The links to the environment can be broadly broken down into four distinct phases;

- The manufacture of the bio-based content of the goods (i.e. the food, drink or tobacco product itself), including production and processing
- The manufacture and use of packaging
- The trade and distribution of products
- The waste (including the bio-based content)

2.3 Agricultural production

The agricultural production on land of food, drink and tobacco is reliant on an adequate supply of uncontaminated fertile soil, productive pasture land and timely and not excessive rainfall or irrigated water supply (see Figure 2.2).

About 45% of EU territory is covered by agriculture (see Figure 2.1). The great majority of EU agricultural land is used for food production either directly for human consumption, or as animal feed (European Commission, 2017a). Environmental impacts from the production of food through agriculture generally include soil erosion with consequent loss of carbon sinks; the removal of hedgerows, trees and other landscape features to increase room for agriculture with impacts both on carbon sinks and biodiversity; loss of farmland and soil biodiversity due to the harmful effects of pesticides, and emissions of greenhouse gases and pollutants to air and water from livestock and arable farming. In addition, large amounts of animal feed are imported from outside the EU where adverse environmental consequences such as deforestation are sometimes associated with their production (European Commission, 2013).
Tobacco growing is particularly liable to lead to eroded soils – so much so that some Member States require tobacco to be rotated with other crops so that the soil can recover. Less than one thousandth of a % of EU farmland was used for the production of tobacco in 2014, with most tobacco for EU consumption being imported.

Key impacts of the drinks industry include production of raw materials, packaging and transport, as liquids are heavy. A 2016 LCA of the impacts of production and consumption of British beer showed, for instance, that one litre of beer requires 10.3–17.5 MJ of primary energy, 41.2–41.8 litre of water and emits 510–842 g of CO$_2$ eq. Production of raw materials was the main impact (47–63%), followed by packaging (19–46%) (Amienyo and Azapagic, 2016).

A full account of all the environmental impacts and dependencies involved in the agricultural production and manufacture of the heterogeneous range of foods is beyond the scope of this case study. In order to illustrate the range we concentrate on two types of product – meat and sugar.
Figure 2.2 Links between the food, drink & tobacco sector and the environment

<table>
<thead>
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<th>Key transformations/ processes</th>
<th>Key outputs/ emissions to the environment</th>
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</thead>
<tbody>
<tr>
<td>Production</td>
<td></td>
<td></td>
<td>• GHG emissions from EU agriculture: 10% of EU's total emissions (Eurostat, 2015 data)</td>
</tr>
<tr>
<td>Clean water</td>
<td>Soil preparation/ tillage</td>
<td>Land use change</td>
<td>• Production of a 0.5 litre bottle of carbonated soft drink estimated to require 150-300 litre of water (Erzin et al, 2011)</td>
</tr>
<tr>
<td>Adequate, timely and not excessive rainfall</td>
<td>Removal of hedgerows, trees and other landscape features</td>
<td>GHG emissions</td>
<td>• EU food processing accounts for 28% of total energy used in food production (2012 data)</td>
</tr>
<tr>
<td>Uncontaminated fertile soils</td>
<td>Sowing, adding of nutrients, irrigation, crop protection, weeding</td>
<td>Air pollution (NH₃, etc.)</td>
<td>• Food processing is the 2nd largest contributor to food waste in EU-28 – 17 million tonnes (2015 data)</td>
</tr>
<tr>
<td>Productive pasture land</td>
<td>Harvest and storage</td>
<td>Loss/degradation of habitat</td>
<td>• BOD in food processing wastewater can be 10-100 times higher than domestic (2006 data)</td>
</tr>
<tr>
<td>Energy</td>
<td>Animal housing and health</td>
<td>Loss of biodiversity</td>
<td>• --</td>
</tr>
<tr>
<td>Chemicals and pharmaceuticals</td>
<td>Feeding</td>
<td>Pollution of water courses/ groundwater</td>
<td>• Europeans throw away over 360kg of plastic packaging per person per year.</td>
</tr>
<tr>
<td>Animal feed</td>
<td></td>
<td>Soil pollution, degradation and erosion</td>
<td>• Food waste, Europe: on average 173 kg per person (2014 data).</td>
</tr>
</tbody>
</table>

Processing

<table>
<thead>
<tr>
<th>Key inputs/ resources from the environment</th>
<th>Key transformations/ processes</th>
<th>Key outputs/ emissions to the environment</th>
<th>Examples of available data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clean water</td>
<td>Materials reception and preparation (handling, storage, sorting, peeling, etc.)</td>
<td>GHG emissions</td>
<td>• EU food processing accounts for 28% of total energy used in food production (2012 data)</td>
</tr>
<tr>
<td>Energy</td>
<td>Size reduction, mixing and forming (cutting, pressing, milling etc.)</td>
<td>Air pollution</td>
<td>• Food processing is the 2nd largest contributor to food waste in EU-28 – 17 million tonnes (2015 data)</td>
</tr>
<tr>
<td>Chemicals</td>
<td>Separation (extraction, centrifugation, filtration etc.)</td>
<td>BOD in waste water</td>
<td>• BOD in food processing wastewater can be 10-100 times higher than domestic (2006 data)</td>
</tr>
<tr>
<td></td>
<td>Product processing (soaking, dissolving, fermentation, smoking etc.)</td>
<td>Organic waste (scraps, etc.)</td>
<td>• --</td>
</tr>
<tr>
<td></td>
<td>Heat processing (melting, boiling, baking, frying, pasteurisation etc.)</td>
<td></td>
<td>• Europeans throw away over 360kg of plastic packaging per person per year.</td>
</tr>
<tr>
<td></td>
<td>Concentration by heat (evaporation, drying and dehydration etc.)</td>
<td></td>
<td>• Food waste, Europe: on average 173 kg per person (2014 data).</td>
</tr>
<tr>
<td></td>
<td>Chilling processes (cooling, freezing, freeze drying etc.)</td>
<td></td>
<td>• --</td>
</tr>
<tr>
<td></td>
<td>Post processing operations (packing, filling, storage under gas etc.)</td>
<td></td>
<td>• --</td>
</tr>
</tbody>
</table>

Trade and distribution

<table>
<thead>
<tr>
<th>Key inputs/ resources from the environment</th>
<th>Key transformations/ processes</th>
<th>Key outputs/ emissions to the environment</th>
<th>Examples of available data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Energy</td>
<td>Packaging</td>
<td>GHG emissions</td>
<td>• EU food processing accounts for 28% of total energy used in food production (2012 data)</td>
</tr>
<tr>
<td>Fossil fuels</td>
<td>Transportation</td>
<td>Air pollution</td>
<td>• Food processing is the 2nd largest contributor to food waste in EU-28 – 17 million tonnes (2015 data)</td>
</tr>
<tr>
<td>Packaging materials</td>
<td>Cooling</td>
<td>Noise</td>
<td>• BOD in food processing wastewater can be 10-100 times higher than domestic (2006 data)</td>
</tr>
</tbody>
</table>

Waste management

<table>
<thead>
<tr>
<th>Key inputs/ resources from the environment</th>
<th>Key transformations/ processes</th>
<th>Key outputs/ emissions to the environment</th>
<th>Examples of available data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Energy</td>
<td>Transportation</td>
<td>GHG emissions</td>
<td>• EU food processing accounts for 28% of total energy used in food production (2012 data)</td>
</tr>
<tr>
<td>Fossil fuels</td>
<td>Sorting/separation</td>
<td>Air pollution</td>
<td>• Food processing is the 2nd largest contributor to food waste in EU-28 – 17 million tonnes (2015 data)</td>
</tr>
<tr>
<td></td>
<td>Landfilling</td>
<td>Pollution of air and water</td>
<td>• BOD in food processing wastewater can be 10-100 times higher than domestic (2006 data)</td>
</tr>
<tr>
<td></td>
<td>Incineration</td>
<td>Noise</td>
<td>• --</td>
</tr>
<tr>
<td></td>
<td>Biogas generation</td>
<td>Littering</td>
<td>• Europeans throw away over 360kg of plastic packaging per person per year.</td>
</tr>
<tr>
<td></td>
<td>Anaerobic fermentation</td>
<td>Landfilling</td>
<td>• Food waste, Europe: on average 173 kg per person (2014 data).</td>
</tr>
<tr>
<td></td>
<td>Waste water treatment</td>
<td>Landfill leaching</td>
<td>• --</td>
</tr>
<tr>
<td></td>
<td>Discharge to water bodies</td>
<td></td>
<td>• --</td>
</tr>
</tbody>
</table>
2.1.1 Meat

The most important EU meat products are pig meat, beef and veal, poultry meat and meat from sheep and goats. In 2017, the EU produced 23.5 million tonnes of pig meat, 14.7 million tonnes of poultry meat, 8.1 million tonnes of beef and veal and just over 900,000 tonnes of sheep and goats’ meat. Animals are reared in very different circumstances but as a general rule the monogastric animals are reared indoors in intensive, enclosed facilities whilst ruminants such as cattle, sheep and goats are reared outdoors with varying degrees of intensity. In particular cattle may be reared “extensively” on semi-natural pasture land where the farmers’ activity is limited to occasionally hay cutting and scrub clearance, or “intensively” on so-called improved grassland where mineral fertiliser has been used to increase the production of grass in order to support more animals.

EU pig meat is very competitive on world markets and over 10% was exported in 2017. Although average EU prices for other meat products including beef and veal are close to world levels the market for beef is protected by very high tariffs and trade liberalisation discussions invariably arouse concerns about the future of the EU’s beef sector.

The environmental impacts of meat are vastly different depending on the type of meat. In terms of land use, poultry and pig meat have a mean demand of 12 and 17 m$^2$/kg of meat, compared to 326 and 369 m$^2$/kg for beef and lamb respectively. Similar are the differences in terms of GHG emissions, where a kg of meat delivered to the customer is associated with 10 and 12 kg CO$_2$ eq for poultry and pig meat and 40 and 100 kg CO$_2$ eq for lamb and beef respectively. Slightly different is the demand for freshwater, which starts from 660 l per kg of poultry and reaches 1 451 l for beef, 1 796 l for pig and 1 803 l for lamb.

Table 2.1 Environmental impacts per kg of meat

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Beef</th>
<th>Sheep and goat</th>
<th>Pig</th>
<th>Poultry</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>m$^2$</td>
<td>326.2</td>
<td>369.8</td>
<td>17.4</td>
<td>12.2</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kg CO$_2$ eq</td>
<td>99.5</td>
<td>39.7</td>
<td>12.3</td>
<td>9.9</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>g SO$_2$ eq</td>
<td>318.8</td>
<td>139.0</td>
<td>142.7</td>
<td>102.4</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>g PO$_4$ eq</td>
<td>301.4</td>
<td>97.1</td>
<td>76.4</td>
<td>48.7</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>l</td>
<td>1451</td>
<td>1803</td>
<td>1796</td>
<td>660</td>
</tr>
</tbody>
</table>

Extrapolated on a European scale, this would translate as the following total environmental impacts during the lifecycle of meat production in Europe$^2$:

Table 2.2 Total environmental impacts of meat production in Europe

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>km$^2$</td>
<td>348 792</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kt CO$_2$ eq</td>
<td>1 254 469</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>t SO$_2$ eq</td>
<td>7 495 703</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>t PO$_4$ eq</td>
<td>4 976 347</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>billion m$^3$</td>
<td>65 034</td>
</tr>
</tbody>
</table>

Meat products (beef, pork and poultry) along with dairy are among the foodstuffs with the greatest environmental impacts via livestock production (Notarnicola et al, 2017). The main environmental impacts of meat are related to livestock production. In the EU context, these impacts are:

- Greenhouse gas emissions, especially CH$_4$, from enteric emissions from ruminants (the largest single source of CH$_4$ in the EU-28 at 187 Mt CO$_2$ eq/year) (Eurostat, 2017)) and from manures (64 Mt), loss

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$^2$ Based upon EU production figures from FAOSTAT for 2016
of soil carbon from overgrazing, and CO₂ emissions from transport\(^{83}\) taking animals to and from markets and to places of slaughter;

- Loss of biodiversity when natural pastures are “improved” to allow more intensive livestock rearing. Conversely, extensive livestock rearing can help to conserve and increase biodiversity compared to a counterfactual of land abandonment although whether this happens is case-specific;
- Air pollution from NH₃ associated with manure management, and
- Water pollution with nitrate (NO₃) also associated with manure management.

Although animals do not add nitrogen or phosphorus to the environment, concentrated deposition of surplus nitrate and phosphate from fodder and feed can cause environmental problems such as eutrophication and poor water quality (Buckwell and Nadeu, 2016). According to Buckwell and Nadeu:

Livestock systems account for a very large share of losses of nutrients in the EU. They are responsible for 81% of the nitrogen input to the aquatic system from agriculture (Westhoek et al., 2015). In addition, the livestock sector is responsible for 23-47% of all N river load to coastal waters and 17-26% of P loads to rivers (Leip et al., 2015).

An important indirect environmental impact of EU livestock – and pig meat in particular – arises because a high proportion of protein used in animal feed is imported. Such imports have the potential to cause degradation in exporting countries, for instance through land use change.

### 2.1.2 Sugar

The main environmental impacts associated with the production of sugar are:

- Water consumption, particularly in non rain-fed systems such as beet production in Southern Europe and cane production;
- Soil erosion;
- Pollution and loss of biodiversity from the use of pesticides. However, beet is usually grown as part of a rotation with wheat or other cereals, acting as a break crop and reducing the inputs needed for those crops;
- Transport emissions and noise, and
- Factory emissions (restricted to BAT).

A study which examined 18 different systems for growing and processing sugar beet in the UK estimated average environmental impacts per hectare of sugar beet as 21.4 Gj energy consumption; 1.4 tonnes CO₂eq; 3.3 kg N leakage; and 15.2 kg N lost through nitrification (Tzilivakis et al, 2005). A water footprint study has estimated that 150-300 litre of water is required to produce a 0.5 litre PET-bottle of soft drink, concluding that agricultural ingredients including sugar of the soft drink have the biggest share of the total water footprint (Ercin, Aldaya and Hoekstra, 2011).

### 2.4 Food processing

Whilst some food is sold directly to consumers at the farm gate, over 90% of food consumed is processed (ESF and COST, 2009). Food processing takes place in a wide variety of ways, but most processes require large inputs of energy and clean water. Direct environmental impacts related to food processing include organic wastes which are liable to generate greenhouse gases if improperly managed and organic contamination of wastewater which, if discharged into watercourses, can result in excesses of both biological oxygen demand and nutrients.

Processed food is commonly packaged as it moves along the value chain towards the final consumer. In 2016, 40% of Europe’s total demand for plastics was used for packaging (Plastics Europe, 2016), of which a majority is only used once (Ellen MacArthur Foundation, 2017a). Less than 30% of plastic waste in Europe is collected for recycling, and landfilling and incineration are the dominant approaches to manage post-

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\(^{83}\) These emissions are not recorded separately in the EU’s GHG emissions inventory
Energy use in food processing is a major part of the environmental impact. In International Energy Agency (IEA) member countries, manufacturing industry is the second largest consumer of energy (27%), of which food and tobacco manufacturing is the fourth largest consumer (12%) (IEA, 2017). In the EU, food processing accounts for 28% of total energy used in food production (2013 figures) (Monforti-Ferrario et al, 2015). Of a total final energy consumption of about 28,300 kTOE in 2013, the food and beverage industry used an estimated 62% for process heating, 34% for electrical use and 10% for process cooling. While energy efficiency in the EU food, drink and tobacco sector has improved over time (Chan and Kantamaneni, 2015), efficiency gains in the sector are often offset by increasing production (IEA, 2010). In 2013, renewables accounted for only 7% of the energy used in food production and consumption in the EU, compared to 15% in the overall energy mix (Monforti-Ferrario et al, 2015).

Water use also contributes to the environmental footprint of the sector. Much of the water used in food processing is required to be of drinking water quality to avoid contamination, according to the EU rules on hygiene of foodstuffs (Chan and Kantamaneni, 2015). It therefore competes with the water needs of cities and local communities (Meneses, Stratton and Flores, 2017).

As food processing uses significant amounts of water, it also generates high volumes of wastewater. Food processing wastewater is often high in biochemical oxygen demand (BOD) – levels can be 10 to 100 times higher than in domestic wastewater (European Commission, 2006). BOD is the amount of dissolved oxygen required by microorganisms to decompose the organic matter present in water. Increased BOD reduces the amount of oxygen available to other organisms which can cause hypoxia which can be harmful to aquatic ecosystems. Organic matter can also pollute drinking and bathing water (EEA, 2017c).

The processing sector is meanwhile the second largest contributor to food waste in the EU-28 (FUSIONS, 2015), accounting for 19% or 17 million tonnes (Secondi, Principato and Laureti, 2015).

While the health impacts of tobacco have been widely studied, the environmental impacts of its manufacturing are less understood. The WHO recently produced a comprehensive report on the topic, suggesting that manufacturing and processing may in fact be one of the greatest sources of tobacco’s environmental damage. Impacts related to manufacturing cigarettes include, for instance, chemicals used for processing and coating tobacco and use of wood resources and related generation of effluents from paper manufacture and producing the packaging (WHO, 2017).

Since 2008, the European market for different types of Electronic Nicotine Delivery Systems (ENDS, or ‘e-cigarettes’) has increased drastically, reaching an estimated value of €2.16 billion in 2014 (EC, 2016). Although production of various components of ENDS devices are associated with a number of environmental impacts and dependencies (see Box 1 below), the overall environmental impacts of electronic cigarettes and their manufacturing are unknown (Chang, 2014; WHO, 2017).

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**ENDS and e-cigarettes**

ENDS heat and vapourise a liquid matrix (containing nicotine and flavours or only flavour), delivering an aerosol to the user. They typically contain a metal case, wires, light emitting diode (LED) lights and plastic components, and are powered by lithium ion batteries (which contain heavy metals with known toxic

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84 All OECD countries apart from Chile, Iceland, Israel, Mexico and Slovenia
[https://www.iea.org/countries/membercountries/](https://www.iea.org/countries/membercountries/)

85 Note that figures do not sum to 100% - these figures are drawn directly from Chan and Kantamaneni (2015), where the same is true.

86 In some circumstances non-drinking water (i.e. non-potable) is used by the food industry (e.g. for fire control, steam production). In these instances the water should be clearly identified as non-drinking water and not connect or mix with the drinking water supply used directly in food production [http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2004:139:0001:0054:en:PDF](http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2004:139:0001:0054:en:PDF).
effects (Kang, Chen and Ogunseitan, 2013) (Lerner et al, 2015). Most devices use either cartridges ('cartomisers') prefilled with liquid or are so called vapours where the liquid is poured into a larger metal dispenser.

2.2.1 Sugar

Sugar is made by processing either sugar beet (an arable crop usually grown in rotation with other crops, such as wheat in cooler areas such as Northern Europe, but also under irrigation in drier parts including Spain) or sugar cane (a permanent crop grown in tropical regions including a number of France’s Overseas Territories which are part of the EU). It has a variety of uses. Around 12% of EU sugar beet production in 2017 was processed into ethanol – a first generation biofuel.

To make sugar from beet the raw beet is first cleaned and cut into strips which are placed into a diffuser in which hot (50 - 80°C) water is diffused through them to yield a sugary juice. This is then carbonated. Milk of lime (Ca(OH)₂) is added to the juice and attracts impurities which are precipitated by passing bubbles of CO₂ through it. The liquor is then passed through a series of evaporation stages to concentrate it, pan-boiled to precipitate crystals which are finally milled in drums to produce pure white sugar crystals. A range of by-products are extracted including pulp from the diffusion process which can be fed to animals.

Cane sugar is usually refined. Phosphoric acid and calcium hydroxide are added to a strong syrup of the sugar to precipitate calcium phosphate which attracts and removes some of the impurities and can be skimmed off. The remaining liquor is repeatedly concentrated to super-saturation and then dried under vacuum to yield pure crystalline sugar. This is centrifuged to separate it from the remaining liquor and then dried in silos and bagged for distribution to industrial users and retailers.

Both cane and sugar refineries are regulated under the Industrial Emissions Directive if – as is usual – they have a thermal capacity of at least 50 MW. Their direct impacts are as a result required to comply with Best Available Techniques and are not further considered in this note.

Biofuels are produced from sugar beet or cane (among other feedstocks) by distillation in which case the crystallisation process is omitted.

In 2017, EU farmers devoted 1.7 million hectares to sugar beet production (European Commission, 2017b), producing 105 million tonnes of sugar beet. 13 million tonnes was sold for the production of ethanol with 92 million tonnes destined for the food market (ibid). 20.5 million tonnes of sugar was produced. Approximately 0.9 million tonnes went to industrial users for pill coatings etc., with a small quantity sold as granulated sugar directly to consumers. The remainder was bought by food and drink producers.

In terms of environmental impacts, a recent meta-analysis (Poore and Nemecek, 2018) calculated the following global mean environmental impacts per kg of beet and cane sugar delivered at the customer, with cane sugar having higher values compared to beet sugar (respectively below):

Table 2.3 Environmental impacts per kg of sugar production

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Beet sugar</th>
<th>Cane sugar</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>m²</td>
<td>1.8</td>
<td>2.0</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kg CO₂eq</td>
<td>1.8</td>
<td>3.2</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>g SO₂eq</td>
<td>12.6</td>
<td>18.0</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>g PO₄³⁻eq</td>
<td>5.4</td>
<td>16.9</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>l</td>
<td>218</td>
<td>620</td>
</tr>
</tbody>
</table>
Extrapolated on a European scale, based on FAOSTAT data on beet sugar production for 2014, this would translate as the following total impacts during the lifecycle of sugar production in Europe:

Table 2.4 Total environmental impacts of sugar production in Europe

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Europe</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>km$^2$</td>
<td>3,549</td>
</tr>
<tr>
<td>GHG emissions</td>
<td>kt CO$_2$eq</td>
<td>35,103</td>
</tr>
<tr>
<td>Acidifying emissions</td>
<td>t SO$_2$eq</td>
<td>245</td>
</tr>
<tr>
<td>Eutrophying emissions</td>
<td>t PO$_4$eq</td>
<td>105</td>
</tr>
<tr>
<td>Freshwater withdrawals</td>
<td>billion m$^3$</td>
<td>4,222</td>
</tr>
</tbody>
</table>

These impacts are associated with the full lifecycle of sugar production, including from activities occurring outside Europe - beginning with the extraction of resources needed to produce inputs for agricultural production, the initial impact of choice by farmers, and ending at the retail store, the point of choice for consumers.

A number of alternatives to sugar exist on the market. These include sweeteners which tend to provide a taste comparable to sugar but a lower calorific value. Sweeteners are considered to be food additives and are regulated by EFSA. The most common artificial sweetener is sucralose (E955). As sucralose is not broken down by the body, it has no calorific value. For this reason it is often advertised as a healthier alternative to sugar. However, this quality also makes it environmentally persistent. Evidence suggests that sweeteners including sucralose, saccharin and acesulfame are also resistant to waste water treatment, potentially allowing them to accumulate in aquatic ecosystems (Sang et al, 2014).

**Isoglucose**

Isoglucose is a sweetener high in fructose made from corn or wheat flour that competes with sugar. Its use in food production was held back before 2017 by the EU’s quota system (see below), but some forecasters expect it to take a substantial share of the sugar market within the next few years.

Stevia (E960) is a sweetener processed from the plant Stevia rebaudiana. Similarly to artificial sweeteners, Stevia is not metabolised by the body so is considered to be zero calorie. Stevia has been approved for consumption in the EU by EFSA since 2011 (EFSA, 2011). Some claims suggest that Stevia unlike other sweeteners is more readily biodegradable. In general, the environmental impacts of sweeteners are not well researched. There is conflicting evidence from producers of sugars and sweeteners about the merits and limitations of different products.

**2.5 Life Cycle Assessment of the food, drink & tobacco sector**

Poore and Nemecek (2018) show that environmental impacts can vary significantly among producers of the same product, which creates substantial mitigation opportunities. This variability, even among producers in similar geographic regions, implies substantial potential to reduce environmental impacts and enhance productivity in the food system. However, mitigation is complicated by trade-offs, multiple ways for producers to achieve low impacts, and interactions throughout the supply chain. Thus, reducing the impacts implies focusing on different areas for different producers and adopting different practices.

One of the most important conclusions from Poore and Nemcek (2018) is that impacts of the lowest-impact animal products typically exceed those of vegetable substitutes, which highlights the importance of dietary change. Dietary change can deliver environmental benefits on a scale not achievable by producers. Moving from current diets to a diet that excludes animal products has transformative potential, reducing land use.

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87 Based upon EU production figures for beet sugar production from FAOSTAT for 2014
for food by 76%, the GHG emissions associated with food by 49%, acidification by 50%, eutrophication by 49% and scarcity-weighted freshwater withdrawals by 19%.

Some of the alternatives to conventional animal products could also present an opportunity to contribute to a change in global agricultural land requirements, particularly for imitation meat and insects, which have the highest land use efficiency (Alexander et al., 2017). The energy and protein production per unit of agricultural area vary by more than 100-fold across conventional animal products and the alternatives, with soybean curd having the highest energy and protein yields (2.2 MJ/m² and 57 g/m²) and beef the lowest (0.02 MJ/m² and 0.4 g/m²). After soybean curd, insect species give the next highest yields, while the yields for cultured meat are similar to eggs, and also relatively close to those for poultry. For cultured meat particularly, the benefits claimed may not justify the substantially higher direct energy requirements for its production.

On the other hand, conducting an LCA of the environmental impacts of large-scale cultured meat production, Tuomisto et al 2011 show that in comparison to conventionally produced European meat, cultured meat involves approximately 7-45% lower energy use (only poultry has lower energy use), 78-96% lower GHG emissions, 99% lower land use, and 82-96% lower water use depending on the product compared (see Figure 2.3). Cultured meat is produced by cultivating muscle cells in a growth media. Tuomisto et al assumed Cyanobacteria hydrolysate was used as the nutrient and energy source for muscle cell growth in the cultivation of the meat.

Note(s): Measure is per 1000 kg edible meat as a percent of the impacts of the product with the highest impact in each impact category
Source: Tuomisto et al 2011

Another important aspect is the issue of food waste. According to a recent study (Usubiaga et al, 2018), measures to improve the efficiency of the value chain, including the reduction of waste, offer a significant potential to reduce environmental pressures from food production in Europe – around 20% of the environmental footprint could be reduced if food waste is completely eliminated. More than 90% of the potential environmental benefits are attributable to eliminating the fraction of household food waste, especially from wheat-based products, such as bread, pastries, or cakes, but also fruits and vegetables. Additionally, according to another study (Alexander et al., 2017a), system losses from overconsumption of

Figure 2.3 Comparison of primary energy input, greenhouse gas (GHG) emissions, land use, and water use of cultured meat production with conventionally produced European beef, sheep, pork and poultry

![Figure 2.3 Comparison of primary energy input, greenhouse gas (GHG) emissions, land use, and water use of cultured meat production with conventionally produced European beef, sheep, pork and poultry](image-url)
food are at least as substantial as the losses from food discarded by consumers, and therefore have comparable food security and sustainability implications.

Related to food waste, numerous studies have researched the lifecycle perspective of plastic food packaging. While it is generally true, that single-use packaging may lead to GHG life-cycle emission reductions due to reductions in food waste, there are additional aspects which need to be addressed, including marine litter and the health impacts of chemical migrations. Furthermore, packaging does not address the systemic drivers of food waste, including the oversupply of food, or the societal normalization of wasting edible products in rich countries (Schweitzer et al., 2018). Policies should be designed in such a way that food and packaging waste are addressed simultaneously.

2.6 Meat substitutes – impacts and acceptability

As touched upon in the previous section, some meat alternatives offer potential benefits in terms of reduced energy consumption and land use requirements, though this is not uniform for all meat substitutes (Alexander et al., 2017a). A comprehensive life-cycle study compares the most well-known meat alternatives, namely such based on dairy, gluten, insects, mycoproteins (such as Quorn™) and soymeal, as well as lab-grown meat, with chicken used as a meat comparison (Smetana et al., 2015). The scope is “cradle-to-plate”, i.e. the entire production life-cycle plus food preparation activities. The study showed highest impacts for lab-grown and mycoprotein meat analogues, medium impacts for chicken (with locally-produced feed), dairy and gluten-based analogues, as well as lowest impacts for insect and soy-based substitutes. This is in general agreement with the study of (Alexander et al., 2017), though including a much larger life-cycle scope. This was associated primarily due to higher impacts at the processing stage (esp. energy consumption). The study considers only available technologies and thus does not take into account potential improvements due to technological innovation - room for such exists for lab-grown meat (Tuomisto et al., 2011), which is not yet produced industrially, as well as to a minor extent for mycoprotein, gluten and dairy variants. For mycoprotein meat alternatives, which are currently commercially available, it must be noted that the study compares only versus chicken, while mycoprotein has been shown to outperform beef in terms of GHG impacts (Finnigan et al., 2010).

While meat alternatives have the potential to lower the environmental impacts of global meat consumption, they have a market share of 3-5% in Europe (MINTEL, 2013), though growth is expected, with one estimate purporting the global protein analogue market to be worth approx. 37 billion EUR by 2020 (Business Wire, 2018). A key determinant of meat substitute acceptance is familiarity, which is two-faceted – on one hand, consumers (esp. non-users) prefer products that are more similar to meat in terms of sensory properties (taste, texture, smell etc.); on the other hand is “neophobia” (extreme dislike for new & unfamiliar things), which has been found to not differ between non-users and heavy users of meat substitutes (Hoek et al., 2011). In general, plant and mycoprotein-based meat alternatives have an advantage over lab-grown and insect meats due to the issue of neophobia, but still suffer from the stigma of not being an alternative to “real” meat in the eyes on non-users (ibid.).

Apart from mycoproteins & plant-based alternatives, insect protein is a promising alternative which faces larger consumer resistance – multiple studies in the EU have shown a low level of willingness to eat insects, with the only socio-demographic factor so far identified as an influence being gender, with men reacting somewhat more positively to insects as food than women (Hartmann and Siegrist, 2017). Positive taste experiences (and thus increased familiarity) have been shown to increase acceptability, e.g. via bug tastings or “bug banquets”, though consumers with particularly negative attitudes may not be swayed (Hartmann and Siegrist, 2016). Some studies have found that incorporating insects into familiar dishes such as salads or spaghetti can be a good strategy for increasing acceptance of unfamiliar foods (Schosler et al., 2012), though even more effective is processing that removes negative visual stimuli (Hartmann et al., 2015). In any case, first impressions matter, and increased future aversion is shown when insect products are not sensorially satisfying the first time they are tried (Schouteten et al., 2016).
There are less studies on the acceptability of cultured meat but in general, the emergent pattern is that determinants of acceptability include 1) attitude toward the morality of the technology; 2) expectations of the product itself – its likeness to meat, as well as health considerations, i.e. whether consumers believe that the product is “safe” (Verbeke et al., 2015). While there has been much ado about the promise of cultured meat as a replacement for conventional meat, Stephens et al. (2018) argue that at this stage this may be premature due to a lack of general outlook on what a mass-market cultured meat sector would look like - in terms of resource, environmental and ethical requirements. What is more, the authors argue that it remains to be seen what the EU regulatory landscape would look like in terms of meeting health standards, as well more generally in terms of what sort of product lab-grown meat would be considered as (an animal product or otherwise, which would necessitate different sets of regulation). Finally, the assumption that cultured meat would displace traditional meat is yet to be adequately substantiated (ibid.).

2.7 Socioeconomic footprint of the sector

The European food, drinks and tobacco industry is the largest manufacturing sector in the EU with an annual turnover of almost €1,100 billion (FoodDrinkEurope, 2017). The industry is characterised by great heterogeneity, with a large number of small family-based companies operating alongside global food conglomerates (EEA, 2017b). The industry is also the leading employer in the EU, with over 4 million employees, selling 90% of its produce within the single market (FoodDrinkEurope, 2017). Although the sector’s contribution to the EU gross value added is relatively small (1.7%) (FoodDrinkEurope, 2017), it has a fundamental social and cultural importance in many European regions, and is the main source of income for some local communities (EEA, 2014, 2017b).

2.8 The impacts of environmental policy on resources used for the sector

Food, drink and tobacco production depends on the availability of a range of environmental resources and ecosystem services. Clean water, productive soils and pollinators are especially important. EU legislation and policy underpinning these public goods is therefore an important component of the food production system.

The most relevant EU environment policies are the Water Framework, Groundwater and Nitrates Directives which underpin the availability of clean water; the Common Agricultural Policy’s agri-environment climate measure, cross-compliance and greening measures, which help to preserve healthy soils, and the Pollinators Initiative under which the Commission and Member States have committed to a range of actions to support wild pollinators. More broadly, the Natura 2000 Directives, covering birds and habitats, protect a range of species and habitats providing other ecosystem services to food production, including pest control.

With regard to packaging – which applies to both the sugar and meat industry – the EU Packaging and Packaging Waste Directive (1994/62/EC) sets mandatory packaging recycling targets for Member States, aimed to prevent or reduce the impacts of waste on the environment. Member States have adopted different schemes to achieve and maintain these targets, creating various incentives for the food and beverage sector to reduce the environmental impacts of packaging. By optimising its use of packaging, the sector may achieve economic savings as well as lower the overall carbon impact of the food produced. Looking ahead, the 2018 EU Circular Economy Package, the recently-published EU Strategy on Plastics in a Circular Economy and the forthcoming 2018 food package may come to impact the sector’s use of certain types of plastics. The latter is expected to include a legislative proposal on the EU food supply chain.

The ongoing attention to the negative impacts of single-use plastics, including the recently proposed EU-wide rules limiting their use (2018/0172 (COD)), is likely to alter the resources used in, for instance, sugar-rich soft drink production. Single-use packaging is common for soft drinks, with five of the six leading global soft drinks firms selling an equivalent of two million tonnes of single-use plastic bottles every year (Greenpeace, 2017).

88 2000/60/EC (Water Framework); 2006/118/EC (Groundwater); 91/676/EEC (Nitrates)
89 Contained in regulations 2013/1305/EC, 2013/1306/EC and 2013/1307/EC
Waste legislation which limits the range of available disposal options (for example, by restricting the extent to which landfill may be used) raises the cost of disposal and can thus be expected to have impacts higher up the value chain. Similar impacts could be created artificially, for example by charging householders according to how much food (and potentially other) waste they throw. Hogg et al (2006) found such systems to be an effective means of reducing household waste.

Although not the primary focus of this study, it is interesting to note that existing EU policy related to electronic cigarettes focuses primarily on their potential health risks. For instance, article 20 of the Tobacco Products Directive (2014/40/EU) lays down rules for electronic cigarettes sold as consumer products in the EU. Other policies have relevance to the various components of electronic cigarettes, such as batteries. Like other producers of products that use batteries, for instance, producers of ENDS are responsible for the waste management of the batteries they place on the market, according to the EU Batteries Directive (2006/66/EC). The growing use of these devices creates new streams of waste – such as the disposable cartomisers – which might require additional/different intervention to avoid littering or other negative impacts on the environment.

2.4.1 Meat
The environmental resources used in livestock production include land (particularly grassland for ruminant livestock), nutrients such as nitrogen and phosphorus, animal feed (usually cereal-based) and energy to provide heating, lighting and transport (e.g. to market). The role of EU environmental policy in underpinning these is as follows:

- Restrictions on ploughing permanent grassland under the CAP help to ensure that the supply of land suitable for ruminants is maintained;
- Restrictions under the Nitrates Directive on the allowable concentrations of manure in Nitrate Vulnerable Zones incentivise alternative uses for manure, such as energy production, which can improve the economic viability and sustainability of livestock production.
- The Industrial Emissions Directive, which applies to the largest livestock units, helps to spread the use of best available technology which can have a positive impact on business viability, although as its primary purpose is the reduction of pollution it may also entail extra cost. Policies encouraging more plant-based diets in Europe could contribute to significantly lower environmental impacts of food. Westhoek et al (2014) suggest that replacing 50% of animal-derived foods with plant-based foods would achieve a 40% reduction in nitrogen emissions, 25-40% reduction in greenhouse gas emissions and 23% per capita less use of cropland for food production.

2.4.2 Sugar
Like other food and drink production, sugar production benefits from clean water, healthy soil and ecosystem services including pollination. No environmental resource needs are unique to sugar production but the industry benefits from imports of cane sugar which are governed by trade agreements.

As transport is another important source of environmental impact for the drinks industry, environmental policies impacting heavy road transports and incentivising more effective/environmentally benign transport options might have an impact on resource inputs of the drinks industry.

Finally, policies to incentivise healthier soft drinks may contribute to shifts from natural sweeteners to artificial sweeteners. Research suggest, however, that also artificial alternatives can have negative impacts on the environment, including unforeseen consequences for aquatic life (Borges et al, 2017).

2.9 The role of investment
As mentioned, food and drink businesses are exceptionally heterogeneous. Small and medium-sized enterprises make up more than half of the industry’s turnover (FoodDrinkEurope, 2017), operating alongside global businesses such as Unilever worth tens of billions of Euros. A company such as Unilever has a capital structure diversified between many different equity investors and lenders which provides a degree of protection from any sudden shortage of investment. In addition, such companies manufacture hundreds if not thousands of different products, which reduces the likelihood that an environmental issue affecting
any particular product line would inflict major damage on the business as a whole. By contrast, an SME producing a handful of products and reliant on a combination of family and banking capital would be significantly more vulnerable.

With regard to the sugar industry, the EU production quotas and import tariffs were lifted in September 2017, causing production to increase and sugar prices to drop to match world market levels. With this market liberalisation and facing the price volatility of the global market, it has been suggested that less productive areas in the EU will see a shrinkage of the market, and production will concentrate further to the most cost-effective regions. Depending on the intensity of production and how well these production units mitigate their environmental impacts, this rather dramatic change in the EU sugar industry is likely to also influence its impacts on the environment. Overall, European production is projected to stabilise at a higher level compared to when quotas were in place, and the EU is likely to become a net exporter (Rossi, 2018).

A number of large investors have divested their holdings of tobacco industry stock. However, they have done so out of concern for ethical and health factors rather than environmental ones.

2.10 Environmental pollutant releases

Many sectors of the economy use and/or create pollutants during production processes, with the potential to damage human health and the wider natural environment. Data on pollutant releases is published in the European Pollutant Release and Transfer Register (E-PRTR). Data is classified by NACE code of the emitter, pollutant, location (including country), year, release medium and volume, and as such it is possible to track its evolution over time.

In the analysis below, we present data in volume terms, but also environmental impacts. These are calculated by applying coefficients reflecting the toxicity of different pollutants, taken from ReCiPe2016 LCIA, according to whether they were released via air, water or land. This allows the summation of different pollutant based upon the impact that they have on human health (measured in disability-adjusted life years, DALYs) and ecosystem health (measured as disappeared species per year, species.years).

The number of pollutant releases associated with the food, drink and tobacco sector in the E-PRTR database steadily declines over the period 2007-16; however, the number of datapoints with complete information (and which can therefore be included in the environmental impact analysis below) is relatively steady, so coverage improves over time (see Figure 2.4).

*Figure 2.4* Sectoral observations available in E-PRTR

*Figure 2.5* The environmental impacts of pollutant releases from the food, drink & tobacco sector
The environmental impacts of releases peaks in the late 2000s, and then shows a relatively steady decline thereafter (see Figure 2.5). This is led by CO2 emissions, which dominate the environmental effects. In terms of non-CO2 pollutants, carbon monoxide releases occur throughout the period, while methane releases peak in 2012, but neither shows substantial correlation with trends in CO2 releases (see Figure 2.6).

At a Member State level, releases were consistently highest in France, although the peak in methane

Figure 2.6 Non-CO2 releases from the food, drink & tobacco sector

Figure 2.7 Pollutant releases from the food, drink and tobacco sector by Member State
emissions in 2012 can clearly be attributed to a specific release in Spain (see Figure 2.7).
3 Current trajectory – direction of travel

3.1 Research questions

- How does environmental policy affect the links between the sector and the environment?
- How does the environment and environmental policy affect the link between these sectors and growth, jobs and investment?
- The evolution in consumers and investors demand for increased transparency on environmental performance?
- Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?
- What has driven changes over time (post-2000), and what changes are expected to occur in the future?
- How do SMEs differ from other firms in their answers to these questions?

3.2 Overview

There is evidence of growing consumer awareness and demand for greater transparency of environmental performance of different food and drinks on the market. A 2010 Eurobarometer survey indicated that up to 40% of Europeans were willing to pay more for products whose production preserves the environment, respects social conditions or helps developing countries (Eurobarometer, 2010). Diets and markets for different food and beverages can reflect how consumers and industry are responding to growing environmental awareness.

3.3 How consumer diets are changing

In the EU, demand for products labelled as organic is arguably a reflection of this reality. In 2016, the market for organic food was estimated to be worth 30.7 billion in the EU. Sales of organic food and dedicated organic farmland have increased by 47.7% and 18.7% respectively between 2012 and 2016 (European Parliament, 2016). Support for fair trade products is also increasing. The 2014 annual report from Fair Trade International showed that global Fair Trade certified sales reached EUR 5.9 billion, also experiencing year on year growth (WFTO, 2014).

Diets and food preferences also reflect consumer demand and are also one of the pathways to reducing the environmental impacts from the food and beverage sector. As outlined, vegetarian and vegan diets could significantly reduce the environmental impact of the food sector, primarily due to the land and carbon intensity of livestock and animal agriculture (Vettese, 2018).

There is a lack of reliable statistics on dietary choices in the EU (EVU, 2018). However, some estimates suggest vegetarianism accounts for around 4% of EU citizens, and rates in individual MS vary between 1 and 10%. Recent data from a survey in the UK suggested that the number of vegans had reached 3.5 million in 2018. In 2016, a separate survey suggested the number of vegans in the UK was 540,000 suggesting rapid growth or variability in survey data (Hancox, 2018).

Changing diets are also reflected in investments and market shares. Products such as soy milk, as vegan and lactose free alternatives to milk, have experienced rapid growth in sales and market share in recent years. Previously marginal dietary choices are also open to new channels of investment. One widely publicised UK crowd funded kick starter, which aimed to develop a vegan burger company “Vurger” hit its target of GBP 150,000 in 24 hours (Hancox, 2018).

Overall, Commission data on per capita consumption for meat, fish and dairy suggests gradually declining meat consumption between 2015 and 2025. This is anticipated due to growing social concerns over animal welfare, human health and the environment. These trends are disaggregated for different products – for example fish, poultry and cheese are expected to experience increasing demand over that period. EU trends should however, be put in the context of the global trend which suggest a growing demand for meat products. This will likely result in a greater export of EU meat products (European Commission, 2015).
The EU funded project Live Well for LIFE explored how diets could change in order to reduce GHG emissions in the food supply chain. The projects outputs were three pilot studies in France, Sweden and Spain, with recommendations for sustainable diets which would reduce emissions by 25% compared to the average diet without costing more than the average diet (WWD, 2015).

Investments in research and innovation are also generating demands for novel foods, particularly alternative proteins to conventional products. Examples of these include insects, lab produced meat tissue, soy-based proteins, and mycoprotein - which are based on fungi, as well as insects. Soy based products, such as tofu and bean curd, are already well established on EU markets, whilst other products are less established or in the case of lab proteins, still in their pilot stages. A study comparing conventional and unconventional protein sources showed that alternative meats and insects had the highest land use efficiency. The best performing product was soy-based bean curd, and beef was the worst performing. The study did not consider other high protein vegetables, grains or pulses (Alexander et al, 2017).

3.4 The impact of changing consumer preferences

The scope of this review does not allow a comprehensive assessment of the impacts of consumer preferences on the largely heterogeneous range of products associated with the European food and drink sectors. However, rising pressure on healthcare systems along with a recognition that many consumers eat meat and dairy products well in excess of dietary guidelines suggests that national governments may eventually seek greater influence over consumer diets. This is likely to bear particularly heavily on production of sugar, dairy and meat products. (Buckwell and Nadeu, 2018)

Consumer preferences for food are influenced by a range of factors, including psychological and marketing factors (see Figure 3.1). Many of these factors have been widely studied (Font-i-Furnols and Guerrero, 2014).

Source: Font-i-Furnols and Guerrero (2014)
In general, European consumers consider meat to be a healthy and important component of the diet and have a negative view of excessive manipulation and lack of naturalness of beef production (Verbeke et al, 2010). In fact, health appears to be the primary reason for shifting diets and reducing meat in the diet or avoiding meat altogether (EEA, 2017a; Latvala et al, 2012). Further, a British study has found UK country of origin food labels to be the highest valued food label attribute for the fresh/chilled/frozen (i.e. not processed) meats (excluding chicken) (Fraser et al, 2015), and Verbeke and Ward (2006) show that meat quality and origin become more important for consumers after having been exposed to information campaigns.

Globally, consumption of meat is projected to continue to increase, driven mainly by poultry (Henchion et al, 2014). Consumer trends in meat demand are associated with higher incomes (OECD, 2018), however some research indicates that the influence of price and income factors on meat consumption is likely to decline over time while other factors, such as quality, becomes more important (Henchion et al, 2014). Further, the millennial generation has been associated with trends of reducing meat consumption for ethical and lifestyle reasons (Bord Bia Insight Centre, 2016).

A number of recent reports have identified that even current levels of EU livestock production exceed the carrying capacity of the environment in respect of a number of pollutants. Buckwell and Nadeu, for example, (Buckwell and Nadeu, 2018) attempt to estimate the “safe operating space” for livestock production and conclude that quantitative reductions in meat consumption will be required in addition to achievable improvements in production itself. Policy means to deliver such reductions in consumption – along with a shift towards diets higher in plant-based foods – does not appear to be developed to a significant degree anywhere in the EU, although the issue is beginning to be taken up by NGOs.

Health is similarly a key driver for consumption of sugar. Globally, consumer behaviour has shifted in line with an ambition to eat healthier foods and live healthier lifestyles, including a rising demand for products free from sugar (Kelly et al, 2018). As one example, according to data based on consumption and shopper panels, purchase levels of sugar-rich carbonated drinks dropped 8.6% among all British consumer groups between 2015 and 2016. The authors associate this drop with health preferences. The same research concluded that consumer concerns about sugar has impacted sales of biscuits and chocolate (Quick, 2016). Multi-nationals have adopted various commitments to reduce the amount of sugar in their products or continue the shift to non-sugar sweeteners, including Nestle and CocaCola.

The EU is the second largest consumer of sugar, preceded only by India (USDA, 2018). Consumption of sugars has increased over time but changing consumer preferences and increasing health concerns are expected to cause a 5% reduction in sugar consumption by 2030 (in favour of isoglucose and other sweeteners). Meanwhile, the use of sugar beet and molasses for biofuel production is projected to increase slightly, mentioned as a possible gateway for directing sugar oversupply following the 2017 end of quotas (European Commission, 2017a).

Interestingly, a report by DSM Food Specialities (2015) suggests that the sizeable increase in consumption of sugared dairy products around the world could be a result of dairy products generally being perceived as nutritious options and preferable to other snacks. This would add to the understanding that general health concerns about sugar have complex interactions with consumption patterns. Popkin and Hawkes (2016) argue that, in the absence of intervention, sugar content in various foods worldwide will follow the same increasing pattern as has been seen in the US, where 74% of products in the food supply chain today contain caloric or low-calorie sweeteners, or both.

### 3.5 Sustainability in the food, drink & tobacco sector

The FDT sector is highly heterogenous, and the sustainability challenges varied and substantial. Any move towards sustainability must consider both the supply side (how production processes can be made less environmentally damaging), but also the demand side (for example, how policy can be utilised to shift

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90 See for example the work of the UK-based Eating Better Alliance [https://www.eating-better.org/](https://www.eating-better.org/)
consumer preferences away from, or at least to less frequent consumption of, the most environmentally damaging types of product).

Current policy, both at a European and Member State level, has very much focussed on the former; on sustainable farming techniques on the agricultural side, but also in terms of processing (in terms of energy demand), packaging (through the recent European Strategy for Plastics) and transportation. However, the sector is extremely competitive, and shifting the market rapidly based only on supply-side measures is likely to mean extensive policy, likely to be fiercely resisted by the industry.
4 Future policy priorities

4.1 Research questions

 What are the opportunities for these sectors (including jobs, growth and investment opportunities) provided by the environment and by environmental policy?

 What are the potential business evolutions (business model, product specificities, etc.) driven by environmental policies or voluntary initiatives influenced by the environment?

 Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?

 How does environmental policy affect the links between the sector and the environment?

 How does the environment and environmental policy affect the link between these sectors and growth, jobs and investment?

4.2 Links between the sector and the Sustainable Development Goals

The FDT sector as a source of environmental challenges and dependencies is relevant to all of the UN SDGs (for example, the quality of education may help to determine choices as to what food is grown and eaten, how it is grown and how any residue is disposed of). SDG 2 and 3 have the strongest links, as they summarise the needs which the production of food and drink seeks to satisfy.

4.2.1 SDG 2 – End hunger, achieve food security and improved nutrition and promote sustainable agriculture

Targets 2.1 and 2.2 on ensuring access to food and ending all forms of malnutrition can be linked to the use of agricultural inputs (such as fertilisers and pesticides) to ensure sufficient and reliable harvests in parts of the world with limited production. However, in Europe, the primary driver for lack of access to food and of malnutrition is poverty (FAO, 2017).

Target 2.3 on doubling the agricultural productivity and incomes of small-scale food producers has links to supporting Europe’s large number of small family-based companies. In terms of the pursuit for more sustainable food and drink systems in Europe, the most appropriate size of food production is a much-debated matter91, and likely depends on a range of factors, including the type of produce in question.

Target 2.4 on sustainable food production provides the clearest link between the sector and its environmental performance. Although it does not explicitly mention links between sustainable food production and biodiversity, nor the role of agriculture in mitigating climate change, it does provide an incentive for adopting policies in support of low-impact agricultural production.

Target 2.5 on maintaining plant and animal genetic diversity has indirect implications for food and drinks production in that it has relevance for the ways in which crops are produced and agricultural land is managed.

The main EU policy instrument that helps to ensure a contribution from FDT sectors to this SDG is the Common Agricultural Policy.

4.2.2 SDG 3 – Ensure healthy lives and promote well-being for all at all ages

The links between food and drink and SDG 3 are more indirect in that healthy diets can promote well-being.

Target 3.9 on pollution to air, water and soil has perhaps the most direct link to food and drink production, as all stages of the value chain tend to generate some degree of pollution and contamination. Conventional agriculture has perhaps the clearest links to pollution of soils with potentially hazardous chemicals, whereas

91 See, e.g. discussion piece by Alan Matthews on capreform.eu, from October 2015: http://capreform.eu/does-farm-size-matter/.
BOD contamination of waste water in food processing is a potential health hazard if not managed appropriately.

The main EU policy instruments which help to ensure a contribution from FDT sectors to this SDG are the Directive 2010/75/EU on industrial emissions (IED), the Urban Waste Water Treatment Directive (UWWTD), the Tobacco Products Directive and the Directive 2010/12/EU on excise duty on tobacco.

The following SDGs all require, *inter alia*, that food and drink are produced, consumed and disposed of in ways that care for the Earth’s resources and environment:

- SDG 6 – Clean water and sanitation
- SDG 7 – Affordable and clean energy
- SDG 12 – Responsible consumption and production
- SDG 13 – Climate Action
- SDG 14 – Life below water
- SDG 15 – Life on land

Tobacco is a threat to SDG 3 (the goal of good health) and also, indirectly as a result of death and illness, to SDG 1 (Zero poverty).
Figure 4.1 Mapping the SDGs against the general EU FDT value chain

Note(s): All data sourced from Eurostat, 2017b
4.3 Policy recommendations

Based on the preceding analysis, the future EU-level policy scenarios we consider worth investigating are as follows:

- **Excluding the production of sugar from CAP support.** Reduced EU production of sugar could reduce pressures to plough permanent grassland for arable use, avoiding significant GHG emissions associated with the release of carbon sequestered in the disturbed soil. Yet, sugar producers benefit from CAP support – even though Governments are increasingly trying to deter their consumption through taxation. Although support has been largely decoupled – meaning it cannot generally be targeted to the producers of particular products – there is precedent for the exclusion of particular crops from CAP support. Production of hemp with high concentrations of cannabinoids is only eligible in special circumstances, for instance. Excluding the production of sugar from CAP support could be expected to reduce EU growers’ willingness to grow these crops, with knock-on impacts including higher imports of sugar and tobacco.

- **An EU-level requirement for variable charges for waste.** There is strong evidence for the deterrent effect of variable charges for household waste (of which a high proportion is food and food packaging) on householders. Charges for residual household waste have the potential to boost participation in recycling including through deposit refund schemes, but could also have negative consequences including distributional ones.

- **Minimum harmonised public procurement rules** to encourage healthy and sustainable food in public institutions, including limiting the amount of red meat and sugary drinks purchased with public funds and provided in public contexts. Certain Member States have developed different criteria for different food categories, including meat served in schools (Caldeira et al, 2017).

- **Consumer-facing carbon tax on products (national level).** Such a tax would have particularly high impact on red meat.

- **Raising (food) industry charges for water, based on compulsory metering.** Although the largest food processing plants are permitted under the Industrial Emissions Directive, and so must to an extent benchmark their use of water and other environmental performance against standards of Best Available Techniques, evidence shows a high potential for reduced water consumption in the industry. High and variable charges would be the most direct means of incentivising better water use. In the short term (before industry responded to the price signal) such higher charges might also provide finance for necessary improvements in the water sector itself.

- **Extended producer responsibility for food and drink producers.** The costs of excessive food (and other) packaging are to an extent internalised through the existing system of packaging waste legislation but the same is not true of the externalities associated with the supply of food itself. One option might be simply to charge food processors and/or retailers an amount which reflected the environmental cost of all food waste sent for disposal. The charge would be assessed based on each year’s waste arisings and levied on industry/retail according to an indicator such as weight of sale or market share. Exemptions would be needed for smaller businesses (and might be desirable for other policy reasons) and the scenario work would examine the impact of different choices.

- **Measures to internalise the environmental cost of Electronic Nicotine Delivery Systems (ENDS).** These costs are not well-known pending life cycle analysis but could potentially be more significant than those of conventional cigarettes. If so, a tax or similar disincentive would be warranted for environmental reasons, although it may have health-related side effects. Currently, the ENDS market is mostly unregulated or inadequately regulated (WHO, 2017).

A crucial consideration in the evaluation of relevant policy options is the extent to which the policies directly impact upon prices faced by consumers, and therefore the potential for the policy to have regressive distributional impacts (i.e. to impact more negatively upon those on lower incomes than on higher). This is a particular issue for policy related to essential products (such as food and drink), where a minimum level of consumption is necessary for all households; therefore, particularly at the lower of the
income distribution, the price sensitivity of consumers is minimal (i.e. they cannot reduce consumption in volume terms in response to price rises) and are forced to pay higher prices (reducing the amount they are able to spend on other goods and services, and leading to worse socio-economic outcomes for these households).

4.4 Impacts of future policy

4.4.1 Eliminating edible household food waste

An initial scenario was developed in E3ME to examine the impact of eliminating (edible) household food waste\(^{92}\). The scenario does not detail the policies required to achieve this, but rather examines the potential benefits of achieving this aim. FUSIONS (2016) estimates the value of household food waste of the EU28 to be around €98bn. We used detailed data of per capita waste across Member States to form model inputs. Consumer expenditure on food and drink was reduced accordingly (see Appendix A).

The key impact in the scenario is a reallocation of consumer expenditure from food to other consumer goods and services\(^{93}\). There is a decline in production by sectors associated with the food supply chain: agriculture (-3.0%), food, drink & tobacco (-3.3%), and retail (-1.1%) in 2030. This decline is offset by reallocation of expenditure; the net effect is therefore limited. The scenario results show a minor increase in GDP and employment in 2030, of 0.08% and 0.06% respectively, suggesting that there may be limited or no net economic cost to reducing food waste. These limited economic benefits should be interpreted as an upper boundary of potential impacts: this finding is contingent upon food waste change being achieved through no additional measures (which could incur costs). The economic logic is applicable across all Member States.

Note that this policy does not have detrimental distributional effects; although no explicit judgement is made in the modelling as to where the waste reductions are made, it can be assumed that it is a fixed percentage of consumption that is achieved – so that all households reduce their expenditure on food by the same percentage. Since low-income households spend a higher proportion of their income on food, the savings (expressed as a percentage of household income) would be expected to be greater in low income households than in more affluent households.

Examining consumer welfare purely through total consumer expenditure underestimates the wider benefits of eliminating waste. Whilst net change in consumer expenditure is limited, it is redirected from excess food, which it can be assumed has no utility, to beneficial goods and services.

The environmental impacts of this scenario are only partially captured in E3ME, given that there is no treatment of land use in the model. Levels of energy related carbon emissions change very little under elimination of food waste: by 2030 across the EU28 they increase by 0.85 million tonnes (0.03% compared to baseline). The reason for the increase in emissions is that the redistribution of consumer expenditure from food to other sectors pushes consumption to higher carbon products. Similar results were found by Martin van de Lindt et al. (2017) within the Carbon-CAP project, aimed at exploring what policy options might be available to reduce emissions from consumption.

Whilst land use change emissions are not measured in E3ME, a rough calculation can give an estimate of the unaccounted environmental impacts. The reduction in land use emissions arising from elimination of the household food waste modelled is approximately 22.1 million tonnes CO\(_2\) equivalent, by far outweighing the increased emissions in the E3ME model. This value corresponds to embedded emissions in EU consumption, not to production located in the EU.

\(^{92}\) ‘Food waste’ refers to waste of both food and drink throughout this analysis. The edible share of household food waste is estimated at 60% (FUSIONS, 2016).

\(^{93}\) In this case, reallocation of expenditure is spread across all other consumption categories, approximately proportional to relative share of total consumption expenditure, see Appendix A.
4.4.2 Extended producer responsibility

The first scenario examines the impacts of an assumed reduction of edible household food waste; a key question is how policy can be used to achieve such a reduction. One option is to extend producer responsibility in the FDT sector. This scenario models the impact of a waste tax on the FDT sector, for waste produced by households. Such a policy would encourage FDT manufacturers to adjust behaviour to minimise wastage in consumption.

Actions taken by the FDT sector to reduce food waste could include:

- Measures to discourage overpurchasing by consumers – e.g. removing multipack promotions and misleading advertising
- Shortening supply chains – e.g. sourcing from local agricultural suppliers to reduce transit losses and emissions, providing fresh produce, and support local the local agri-economy
- Providing clearer messages to consumers on the storage, preparation and consumption of products – e.g. greater clarity on date labelling
- Optimising packaging to extend the shelf life on high value (and high environmental impact) products – e.g. dairy and meat (avoiding unnecessary application of single use plastics)
- Utilising edible products close to their “use by” dates to generate value added for example in restaurants, or through donations to food banks

This scenario uses data from FUSIONS (2016) to estimate the FDT’s tax liability under a waste tax of €50 (2016 prices) per tonne of household food waste. The scenario examines the ‘worst-case scenario’ of no change in household waste.

The magnitude of the impacts is relatively small, suggesting that this policy would not impose significant economic costs. Unit costs in FDT only increase by an average of 0.25% across Member States in 2020. There is a very small reduction in output of the agriculture and FDT sectors (-0.01%). GDP in the EU is slightly higher than in the baseline. The positive impact on GDP is a result of tax revenues being used to reduce income taxation, and tax incidence falling on manufacturers as well as consumers.

Implementation of this policy would be difficult. It would likely not be possible to trace household waste by FDT manufacturer source. The tax would likely have to be industry-wide, calculated as a function of total household food waste collected/reported: this formulation, however, would blunt incentives for individual FDT manufacturers to take action.

In addition, the impacts are likely to be regressive. As outlined above, unit costs of food is increasing in the scenario; and since poorer households spend more of the income on food, the unit cost increase will be a larger proportion of household income for poorer households than for those with higher incomes.

4.4.3 Emissions pricing of food

Whilst the largest energy consumers in the food, drink & tobacco sector itself fall under the EU ETS, inputs from agriculture do not. This scenario examines the potential to levy a tax on FDT for purchases from agriculture, based on the GHG emissions embedded in the production of these inputs. The scenario uses two modelling approaches. The first examines changes in demand across food groups, and the second the macroeconomic impacts of the tax. The tax rate is set equal to the EU ETS price (see Appendix A for details).

The first part of the modelling in this scenario uses the food demand equations in E3ME; these food demand equations model wholesale demand for agricultural produce, not consumer demand. Changes in the total quantity of food demanded, and of individual food groups, were modelled in physical units. The average wholesale price of one tonne of food across the EU28 is 4.7% higher in 2020 if the tax is levied. Given the inelastic nature of food demand, total EU demand (in tonnes) of food is only 0.4% lower in 2020. Demand for dairy and beef decrease by 0.9% and 1.9% respectively, and demand for cereals and vegetables increase by 0.25% and 0.29% respectively. The estimated reduction in land use change emissions is 2.4
million tonnes of CO₂ equivalent\textsuperscript{94}. The inelastic nature of food demand is a key factor limiting the efficacy of emissions pricing.

The main modelling for this scenario examined the macroeconomic impact of applying the emissions tax to agricultural produce purchased by the FDT sector.\textsuperscript{95} Applying the tax to purchases by FDT, rather than to sales from agriculture, means that EU FDT manufacturers face the tax regardless of the origin of agricultural produce, thus reducing the issue of production leakage at this stage of the supply chain. In the scenario formulation, policy revenue neutrality is achieved through a reduction in direct taxation.

The first order effect of the tax is an increase in costs in the FDT sector leading to an increase in the price of industry output, though at less than full pass-through. Output of the FDT sector is 0.22\% lower by 2030. Net economic impacts are positive however: GDP increases by 0.15\% and employment by 0.04\%. As in the previous scenario, positive impacts are a result of revenue recycling and (in the short run at least) less than full pass-through of taxation. A key effect in this carbon pricing scenario is an increase in consumer food prices. Consumer food and drink prices increase by 1.0\% in this scenario by 2030. This estimate is to some extent a ceiling because it is assumed there is no change in diet; if consumers choose to adjust dietary choices, to products with lower embedded emissions, food and drink price increases would be lower. An increase in food prices is regressive, and the tax could therefore be difficult to implement. The targeted recycling of carbon tax revenues has potential to mitigate negative distributional impacts, however.

The environmental impacts are limited:

\begin{itemize}
  \item Rebound effects. Whilst consumers reduce expenditure on food and drink expenditure is substituted to other consumer categories, increasing emissions throughout other parts of the economy.
  \item Fixed diets and no substitution to less emissions-intensive food. Total production of the FDT and agricultural sectors fall, by 0.22\% and 0.21\% respectively by 2030. However, given model limitations, there is no change in production across food types. Carbon emissions in the FDT sector fall by 0.5\% by 2050.
\end{itemize}

\textbf{4.4.4 Excluding the production of sugar from CAP support}

This policy is more focused on the agricultural sector. We do not have access to a model of the supply side of agricultural produce, and therefore would not be able to assess supply decisions in response to this policy. Financial support available to sugar crops demonstrates difficulties in balancing policy priorities. A recent study, ‘Impact of coupled EU support for sugar beet growing: More production, lower prices’ (Wageningen Economic Research, 2017) examines in detail the effect of voluntary coupled support payments on EU sugar beet production. This demonstrated that such support increased production by around 1.3\%, reducing prices by 4.5\%; so the removal of such support could be expected to have a similar effect in reverse. The removal of such support would lead to a combination lower domestic production (which, assuming no change in EU demand, would lead to increased imports from countries where the environmental impact of production may be higher) and lower profits for producers (assuming that the price support is not solely used to support consumer prices).

A key consideration of excluding sugar from CAP support is production leakage, leading to higher environmental impact globally. Data from Poore & Nemecek (2018) suggest cane sugar production results in, on average, 75\% more GHG emissions than sugar beet (this difference is attributable to land use change emissions).

This policy will lead to increases in food prices, and therefore prove regressive; increases in prices take up a larger proportion of household income for poorer households, and therefore the negative impact is more pronounced amongst this group.

\textsuperscript{94} This is an estimation of land use change emissions in supplying EU demand, including imports.

\textsuperscript{95} For data reasons, no tax was applied for purchases from the fishing sector.
4.4.5 Minimum harmonised public procurement rates
A 2017 report from the Maltese Presidency and the European Commission notes that 77% of EU Member States (and Norway and Switzerland)\footnote{Survey carried out by the Maltese Presidency and DG SANTE, sample only 22 countries.} have public procurement rules for food; many of the rules detailed have a ‘focus on environment and sustainability aspects’, rather than health issues. The total value of the social food service market in the EU has been estimated at €82bn\footnote{€82bn includes both public and private (Maltese Presidency and the European Commission, 2017c)}; additional minimum harmonised public procurements laws could directly affect a significant market. The total magnitude of effect is likely to depend on behavioural effects: encouraging a wider dietary shift by consumers and shifting provision in the catering industry. Upstream effects of this policy on agricultural production are likely to be intuitive: a shift of agricultural production in the EU from meat and sugar to more sustainable produce. The effect on the FDT sector would be a shift in agricultural products processed. The distributional effects (i.e. whether this has more or less impact on low income households) is unclear, and would depend upon the substitutability of specific products and their relative weight in the typical purchases of low-income households.

4.4.6 Raising (food) industry charges for water, based on compulsory metering
This policy would cause an increase in costs and, again, likely result in a regressive increase in food prices. As noted earlier in the chapter, there is no environmental or economic reason to only apply compulsory metering to the FDT sector. Increases in costs would be a function of water-intensity of different industries, and how readily water consumption can be made more efficient. Any revenues raised through metering could be used to finance water investment, where requirements are substantial (as demonstrated in the Water Treatment and Supply Annex report).
Appendix A E3ME macroeconomic methodology

A.1 Scenario One: Eliminating edible household food waste

Scenario One models the potential impacts of the elimination of household food waste. The scenario exogenously reduces consumer expenditure on food and drink using data from FUSIONS (2016). The scenario does not introduce additional policy to reduce food waste; it models the potential effects of eliminating food waste in the absence of the costs of policies.

Estimates of per capita household food waste by category by MS are given in Table A.1; where available in FUSIONS (2016) the individual MS data is used, otherwise the average is applied. The edible share of food waste is estimated at 60%. To give a monetary value of waste, physical weight of edible food waste is multiplied by an estimate of per kg retail value of edible food waste (€3596 per tonne). Population data are then used to yield a total value of household food waste by Member State. We assumed a constant value (in constant prices) of food wastage per capita over time in the baseline.

Food and drink are individual categories of consumer expenditure in E3ME. In apportioning waste value to E3ME consumer expenditure categories, values for ‘collected by local authorities/municipalities’ and ‘home composting’ are allocated to food, and ‘down the sewer’ is allocated to drink.

Table A.1 Household food waste by category (kg per person per year)

<table>
<thead>
<tr>
<th>Member State</th>
<th>Collected by local authorities / municipalities</th>
<th>Down the sewer</th>
<th>Home composting</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td>43.7</td>
<td>8.9</td>
<td>10.7</td>
</tr>
<tr>
<td>Belgium</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Bulgaria</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Croatia</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Cyprus</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Denmark</td>
<td>82.6</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Estonia</td>
<td>52.8</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Finland</td>
<td>63.6</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>France</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Germany</td>
<td>62.7</td>
<td>11.7</td>
<td>9.1</td>
</tr>
<tr>
<td>Greece</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Hungary</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Ireland</td>
<td>54.7</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Italy</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Latvia</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Lithuania</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Luxembourg</td>
<td>78.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Malta</td>
<td>129.6</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Netherlands</td>
<td>66.7</td>
<td>6.2</td>
<td>7.0</td>
</tr>
<tr>
<td>Poland</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Portugal</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Romania</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Slovakia</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
<tr>
<td>Slovenia</td>
<td>70.9</td>
<td>14.9</td>
<td>7.0</td>
</tr>
</tbody>
</table>
Consumer expenditure is estimated in a two-step process: 1) aggregate consumer expenditure; 2) expenditure by category, scaled to match aggregate. The exogenous reduction in expenditure affects only the second stage of this estimation because there is no exogenously imposed change in total consumer expenditure. The effect is a reallocation of expenditure from food, to all other categories, proportional to their relative shares of consumption. If the reduction in consumer expenditure was constructed to reduce total consumption, effectively increasing the propensity to save, the rebound effects in the scenario would be lessened.

E3ME does not have a model of land use, and therefore does not capture the important land use change emissions which would arise from a reduction in food production. In order to provide an indication of the potential magnitude of these land use change emissions, we used available data to provide an indicative estimate. E3ME data are used to estimate the reduction in food demand by category, given that FUSIONS (2016) does not provide data on the composition of food waste. The assumption is that food waste by category is proportional to demand; that is, the percentage of meat purchased which is wasted, is the same as for grain etc. For the purposes of this estimate, given the E3ME food equations do not include drink, all waste is considered food, not drink. Land use change emissions are taken from Poore and Nemecek (2018)\textsuperscript{98}. As with the calculations of monetary value, an estimate of 60% edible share is applied. Combining these data sources, it is estimated that the reduction in food production arising from eliminating food waste in FUSIONS (2016) implies an annual reduction of 22.1 million tonnes CO\textsubscript{2} equivalent from land use change.

### A.2 Scenario Two: Extended producer responsibility

Scenario Two models the introduction of an extended producer responsibility policy applied to the FDT industry. A waste tax is applied to the FDT industry, based on waste arising in households. The tax rate is based on the landfill tax discussed in the EC’s ‘Study on assessing the environmental fiscal reform potential for the EU28’. This value is comparable to the packaging waste taxes recommended in the report, which range from €21 to €318 depending on material. Although beyond the scope of this study, an appropriate tax would be equal to the externalities of the food waste. The scenario models a ‘worst case scenario’ where there is no change in household food waste. That is, this scenario estimates the highest cost to FDT that could be expected under such a tax policy, with the lowest environmental impact.

### A.3 Scenario Three: Emissions pricing of food

Scenario Three models the introduction of an emissions tax applied to agricultural produce purchased by the FDT sector. The emissions tax is applied to all GHG emissions, not only CO\textsubscript{2}, with methane being a key GHG in agriculture. The tax rate is set equal to the EU ETS price; the E3ME baseline uses the International Energy Agency, World Energy Outlook 2016, Current Policy Scenario values.

Data published in Poore and Nemecek (2018)\textsuperscript{99} are used to calculate how the emissions tax is applied to production across different food groups (see Table 1). The dataset provides GHG emissions accrued throughout the food supply chain. For this scenario, only emissions arising in agricultural production itself are included (feed and farm), excluding land use change. Emissions arising in transport, retail, packaging etc are covered under different policy.

\textsuperscript{98} To convert food classifications to be consistent with E3ME, some aggregation was required. Where required, a weighted average was calculated, using production levels reported in Poore & Nemecek (2018) as weights.

\textsuperscript{99} One caveat is that the dataset is global. However, given the quality and detail of the data, it was judged to be suitable. See \url{http://science.sciencemag.org/content/360/6392/987/tab-figures-data} for supplementary information and data.
The modelling for Scenario Three comprises two distinct parts. The first uses the food demand equations in E3ME, to examine demand response across different categories of food. The second applies an increase in intermediate demand costs to the FDT sector, to examine wider socio-economic costs.

The first approach uses the food demand equations which are a recent development to E3ME and are not yet fully integrated. They model demand for agricultural produce in physical units. Demand is modelled at an aggregate and food group level; food group level results are scaled as required. Behavioural relationships in the equations are estimated econometrically; the most important factor for this scenario is the price elasticity of demand.

The emissions tax is applied to each food group as an exogenous increase in price. Table A.2 details the value of the emissions tax by 2030, using a carbon price of €34.25 per tonne of CO2 equivalent. Reported land use change emissions are estimated using data from Poore and Nemecek (2018) and the food demand results from E3ME.

Table A.2 Emissions tax and demand response in 2030 by E3ME food group

<table>
<thead>
<tr>
<th>Food Group Classification</th>
<th>Emissions tax (€/tonne)</th>
<th>Demand change from baseline (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cereals</td>
<td>22.7</td>
<td>0.42%</td>
</tr>
<tr>
<td>Rice</td>
<td>121.7</td>
<td>-0.65%</td>
</tr>
<tr>
<td>Potatoes</td>
<td>6.6</td>
<td>0.22%</td>
</tr>
<tr>
<td>Other roots</td>
<td>7.5</td>
<td>1.01%</td>
</tr>
<tr>
<td>Maize</td>
<td>16.3</td>
<td>-0.56%</td>
</tr>
<tr>
<td>Soy</td>
<td>22.6</td>
<td>0.21%</td>
</tr>
<tr>
<td>Sunflower</td>
<td>73.6</td>
<td>-1.12%</td>
</tr>
<tr>
<td>Rapeseed</td>
<td>80.3</td>
<td>-0.33%</td>
</tr>
<tr>
<td>Other oil crops</td>
<td>77.8</td>
<td>-0.40%</td>
</tr>
<tr>
<td>Sugar crops</td>
<td>17.0</td>
<td>-0.98%</td>
</tr>
<tr>
<td>Poultry</td>
<td>83.8</td>
<td>-0.26%</td>
</tr>
<tr>
<td>Pork</td>
<td>158.7</td>
<td>-0.32%</td>
</tr>
<tr>
<td>Beef</td>
<td>1068.6</td>
<td>-0.75%</td>
</tr>
<tr>
<td>Other meat &amp; animal products</td>
<td>605.7</td>
<td>-1.15%</td>
</tr>
<tr>
<td>Fish</td>
<td>115.1</td>
<td>0.24%</td>
</tr>
<tr>
<td>Dairy</td>
<td>83.1</td>
<td>-0.94%</td>
</tr>
<tr>
<td>Fruit</td>
<td>12.2</td>
<td>-0.38%</td>
</tr>
<tr>
<td>Vegetables</td>
<td>10.3</td>
<td>0.31%</td>
</tr>
<tr>
<td>Other</td>
<td>78.1</td>
<td>-0.16%</td>
</tr>
</tbody>
</table>

The second modelling approach applies an emissions tax to purchases of the FDT sector from agriculture. Given that output of the agricultural sector in E3ME is homogenous, this second approach calculates an emissions tax based on baseline wholesale food demand. The requisite assumption is that there is no response in diet in the emissions tax scenario. This approach, therefore, represents the costliest situation, where diets do not respond to substitute to foods with lower emissions tax rates.
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Links between production, the environment and environmental policy

Rubber and plastics – sector study
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1 Introduction

The rubber & plastics sector is a manufacturing sector which generates environmental impacts through use of materials and energy, as well as the generation of waste and emissions. The sector presents a number of opportunities for environmental policy to alter its environmental impact, including circular economy and recyclability/reusability of product issues, as well as impacts of marine litter and the specific challenge of single use plastics.

For the main part, rubber & plastics is important because of product life-cycle impacts. It should be noted that this sector is linked to packaging in the food, drink, & tobacco sector. Plastics is one of the five priority areas addressed in the ‘EU action plan for the Circular Economy’.

The sector is strongly relevant to the SDGs, particularly to: Goal 12: responsible consumption & production; Goal 13: climate action; and Goal 14: life below water. The sector is of direct relevance to the 7th EAP regarding resource efficiency and sustainability.

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101 See points 39, 40, 43(d), & 43(viii) of the Annex of the 7th EAP.
2  Current state of play

2.1  Research questions

- How do these sectors benefit from the environment?
- How do they affect the environment?
- What is their resource use?

2.2  Overview

The plastics sector is a large and economically important sector that is closely coupled to fossil fuel production which provides the sector’s primary feedstock. Rapid growth in the plastics sector has taken place over the course of the last 50 years, notably in the packaging sector and for single use plastics which account for a significant proportion of plastic applications. Mismanaged plastic waste and leakage to the terrestrial and marine environment have become a global sustainability crisis.

2.2.1  Resource use

90% of the plastics produced globally are derived from fossil fuels (Bourguignon, 2017). In total, plastics consume 8% of the global oil production, 4% of which is used as raw material (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016). Plastics production has increased significantly since the mid-twentieth century, reaching 322 and 58 million tonnes in 2015 globally and in Europe respectively. The global plastics production is expected to reach 1.2 billion tonnes annually by 2050 (Plastics Europe, 2016).

2.2.2  Manufacturing

Plastics manufacturing requires large amounts of energy. The main processes for production are polymerisation and polycondensation, which require specific catalysts (Bourguignon, 2017). The production process is responsible for approximately 4% of fossil fuels (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016) and in 2012, CO2 emissions generated from plastics production amounted to 390 million tonnes (WEF, EMF and McKinsey&Company, 2016).

2.2.3  Consumption

Total plastics demand in Europe amounted to 49 million tonnes in 2016. Packaging represents the biggest share, with 39.9%, followed by building and construction (19.7%), automotive (8.9%), electric and electronic (5.8%) and agriculture (3.3%). The rest of the market is dedicated to other materials (22.4%) (Plastics Europe, 2016). Plastics consumption can lead to a loss of material value due to single-use and low recycling rates (Bourguignon, 2017). This is particularly the case of plastic packaging.

2.2.4  Single use plastics and packaging

Single use plastics can include any disposable plastic item which is designed to be used only once. Single use items are often used in packaging, consumer products, cosmetics and healthcare. Examples include: lightweight plastic bags, disposable utensils, beverage containers, coffee capsules, wet wipes, and razor blades. 95% of plastic packaging is single-use, and results in the loss of €70-100 billion in material value from the economy per year (WEF, EMF and McKinsey&Company, 2016). Packaging accounts for 42% of the plastics produced globally since 1950 (Geyer, Jambeck and Law, 2017). Plastic packaging has witnessed a significant increase in production volumes and is expected to double by 2030 and to more than quadruple by 2050 (WEF, EMF and McKinsey&Company, 2016). In particular, plastics is widely used to package food, accounting for 37% of the European market share (Muncke, 2016). Within all plastics application, packaging has the highest recycling rate, 40.8% in 2016 (Plastics Europe, 2018). However, most of the plastic packaging is lost within the same year of its production, contrarily to all other plastic applications – see Figure 2.1 (Geyer, Jambeck and Law, 2017). Recycled plastics tend to be diverted to lower value applications, which could impact the demand for plastic recyclates for specific applications (WEF, EMF and
McKinsey&Company, 2016). The revised waste legislative framework\textsuperscript{102}, adopted in 2018, includes new EU-level recycling targets. In particular, a common EU target for recycling is set at 70% for packaging waste, and 55% for plastic packaging, both by 2030. In addition, the proposed EU directive on single use plastics\textsuperscript{103} will require separate collection for recycling of 77% of single-use plastic beverage bottles placed on the market by 2025, and 90% by 2029.

Figure 2.1 Product lifetime distributions (in years) for plastics from different sectors (log normal probability distribution function)

Source: Geyer, Jambeck and Law, 2017

\textbf{2.2.5 Bioplastics}

Defined as bio-based, biodegradable or both (European Bioplastics, 2017), bioplastics currently have a global capacity of 4 million tonnes (Geyer, Jambeck and Law, 2017). The use of bioplastics comes with advantages as well as drawbacks, therefore representing an ambiguous and often debated alternative to conventional plastics. The main concern of using biological feedstock, such as corn, soya, wheat or sugarcane, as plastics raw materials is found in the associated land use change, previously dedicated to crop cultivations. This could create conflicts with food production and an increase in the use of fertilisers and pesticides which would threaten sensitive habitats (EPA Network, 2017; European Commission, 2011) (see Future Policy Priorities for more detail).

\textbf{2.2.6 Waste management}

The durability of plastics gives this material utility and makes it environmentally damaging. It is estimated that in 2015 plastic waste generation amounted to 6300 million tonnes. Of this amount, 9% was recycled, 12% incinerated and 79% accumulated in landfills or in the natural environment. Leakage into terrestrial and marine environment is a global issue. It is estimated that approximately 8 million tonnes of plastics are released into the world’s ocean each year. Of the amount of plastics already present in the marine environments, packaging represents the largest share. Today, there are 150 million tonnes of plastics in the ocean (WEF, EMF and McKinsey&Company, 2016). The accumulation of litter in marine environments originates principally from land-based sources, accounting for more than 80%, while less than 20% comes from sea-based sources (Ocean Conservancy and McKinsey&Company, 2015).

Identifying the origin and sources of marine litter represents a complex task with inherent levels of uncertainty in the results. A large fraction of marine litter is composed of unidentifiable items (such as small pieces of a larger item), but also identifiable items which could have multiple sources (such as a whole pieces of polystyrene packaging). Nevertheless, a large portion of marine litter can be attributed to specific products, and particularly to single use plastics. Table 2.1 summarises the top ten marine litter items in Europe as laid out in the European Commission Marine litter Impact Assessment, Annex 3\(^{104}\) (SWD(2018) 254 final).

**Table 2.1 Top 10 marine litter items by count during the year 2016**

<table>
<thead>
<tr>
<th>Product Type</th>
<th>Total number</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Drinks bottles, caps and lids</td>
<td>24,541</td>
</tr>
<tr>
<td>2. Cigarette butts</td>
<td>21,854</td>
</tr>
<tr>
<td>3. Cotton bud sticks</td>
<td>13,616</td>
</tr>
<tr>
<td>4. Crisps packets / sweets wrappers</td>
<td>10,952</td>
</tr>
<tr>
<td>5. Sanitary applications</td>
<td>9,493</td>
</tr>
<tr>
<td>6. Bags</td>
<td>6,410</td>
</tr>
<tr>
<td>7. Cutlery, straws and stirrers</td>
<td>4,769</td>
</tr>
<tr>
<td>8. Cup and cup lids</td>
<td>3,232</td>
</tr>
<tr>
<td>9. Balloons and balloon sticks</td>
<td>2,706</td>
</tr>
<tr>
<td>10. Food containers including fast food</td>
<td>2,602</td>
</tr>
</tbody>
</table>


Microplastics (small pieces of plastic litter) which are smaller than 5mm in diameter represent a specific environmental challenge. These can have either primary or secondary sources, the latter being those which have fragmented from a larger item. Important sources include fragmentation of larger items in the environment, release of abrasives from cosmetics and other products, tyre wear and tear, fibres released from the washing of textiles and the spillage of pre-production pellets or powders that are in transit or process prior to being made into everyday plastic items (GESAMP\(^{105}\), 2017).

**Microplastic emissions from tyres and service models**

Motor vehicle tyres, typically made from a combination of synthetic and natural rubber as well as other materials, are an essential technology for the delivery of mobility services. The life-cycle of tyres are also associated with a number of environmental impacts, relating to the sourcing of feedstocks from rubber trees or fossil based synthetic sources, and correct disposal of tyres at their end-of-life. Well-designed or correctly inflated tyres can also improve the performance and reduce the emissions of vehicles. One environmental impact which has been less well researched is the contribution of tyre wear to the emission of microplastics - small plastic particles generated from everyday use of tyres which enter the terrestrial and marine environment (via run-off). A report by IUCN estimated that of all plastic pollution an estimated 15% to 31% originated from primary microplastics – rather than those resulting from the break-up of bigger pieces of plastic in the environment. 28.3% of releases of primary microplastics were the result of the erosion of tyres, second only to releases of synthetic fibres from laundry which


accounted for 34.8% (Boucher and Friot, 2017).

The abrasion of motor vehicle tyres represents a challenge for policy makers, particularly as there are currently no tyres commercially available which result in no or significantly less emissions of microplastics. In 2000 tyre producer Michelin set up its “Fleet Solutions” service, which offered high performance tyres for haulage companies. Tyres are leased to companies on a per mile basis rather than being sold (Michelin, 2017). If tyre technology were developed which would reduce the release of microplastics, the capital costs for implementing this technology could be made available to tyre users through access over ownership models.

2.2.7 Recycling

In 2014, 29.7% of the 25.8 million tonnes of post-consumer plastic waste generated were recycled (Plastics Europe, 2016). Plastic packaging has the highest rate of recycling, 40.8% in 2016 (Plastics Europe, 2018) The recycling potential of plastics exceeds this, but most plastic items are not collected for recycling, currently only 14% (WEF, EMF and McKinsey&Company, 2016). The recycling rate of plastics is also lower than for other materials, such as paper (58%) and iron and steel (70-90%). Plastics are often recycled into products of lower value which degrade in quality with successive cycles (WEF, EMF and McKinsey&Company, 2016).

2.3 The environmental impact of single use plastics

Twelve products and their potential alternatives were originally considered for modelling (see Table 2.2). Their selection was aligned with the Commission Eunomia study for the Strategy on Plastics Impact Assessment “Reducing Marine Litter: action on single use plastics and fishing gear” which was conducted under the overall Strategy on Plastics Impact Assessment. The criteria for selection of plastics alternatives (single-use non-plastic items – SUNP and multi-use items – MU products) were that:

4. The materials of which SUNP items are composed avoid the generation of microplastics. This thus excluded biodegradable plastics from the study scope as such biodegradability can only be insured in specific conditions which are seldom met in the marine environment (Thompson, 2006; Kershaw, 2015).

5. Alternative products meet the same function as the plastic products that they substitute in terms of properties that the materials ensure. Such products were not identified for product groups Crisps packets and Sweet wrappers (transmission of O2 and water vapour, opacity), as well as for SUNP Drinks cups and lids (permeability and resistance of insulating layer to heat) and sanitary towels (permeability and absorbency).

6. Multi-use items need to ensure that use of single-use plastics is avoided. This ruled out reusable cigarette filters, as such are used in addition of a traditional cigarette (as an additional filter) and would thus not displace the use of a cellulose acetate filter.

7. Alternatives need to satisfy broadly the same market. This ruled out items such as e-cigarettes, which are tobacco substitutes and thus not necessarily targeting an analogous market segment.

<table>
<thead>
<tr>
<th>Product category</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cigarette butts</td>
<td>Cellulose acetate filter</td>
<td>Natural fibre filter (hemp/cotton)</td>
<td>-</td>
</tr>
<tr>
<td>Drinks bottles</td>
<td>Average volume PET bottle</td>
<td>Average non-plastic container (Aluminium/glass)</td>
<td>Average multi-use container: Consumer-led: PET/Aluminium Industry-led: PET/Glass</td>
</tr>
<tr>
<td>Products</td>
<td>PP bud</td>
<td>Paper bud</td>
<td>Reusable MDPE bud</td>
</tr>
<tr>
<td>---------------------------</td>
<td>----------------------</td>
<td>------------------------------------</td>
<td>-------------------------------------</td>
</tr>
<tr>
<td>Crisps packets</td>
<td>Excluded from scope</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sweet wrappers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sanitary towels</td>
<td>Ultrathin pad (PE, PP, PET, SAP)</td>
<td>-</td>
<td>Washable cotton pad</td>
</tr>
<tr>
<td>Wet wipes</td>
<td>Wet wipe (w/ lotion)</td>
<td>Cotton ball + lotion</td>
<td>Cotton handkerchief + lotion</td>
</tr>
<tr>
<td>Cutlery</td>
<td>Average PP utensil</td>
<td>Average wooden utensil</td>
<td>Average steel utensil</td>
</tr>
<tr>
<td>Straws</td>
<td>PP straw</td>
<td>Paper straw</td>
<td>Average reusable straw (steel/silicone)</td>
</tr>
<tr>
<td>Stirrers</td>
<td>PP stirrer</td>
<td>Wooden stirrer</td>
<td>Steel stirrer</td>
</tr>
<tr>
<td>Food containers</td>
<td>PS clamshell container</td>
<td>Paperboard + wax container</td>
<td>PE tupperware box</td>
</tr>
<tr>
<td>Drinks cups and lids</td>
<td>Paper cup w/ PE coating and LDPE lid</td>
<td>-</td>
<td>Reusable PP cup (w/ LDPE, rubber, silicone components)</td>
</tr>
</tbody>
</table>

Note(s): Products with materials separated by forward slash are market averages of separate products made of the materials given.

In choosing the reference products for each product category in Table 2, generally the most widely used products have been selected. Where multiple such products exist (such as different volumes of drinks bottles), averaged products have been modelled, either in terms of mass (in the case of different sizes of the same product) or in terms of composition (in the case of alternatives from different materials existing for SUNP and MU items). Where possible, market reports have been used in order to derive average reference products.

The specification of each reference product is detailed in Appendix A.

The major strength of life-cycle assessment is that it allows for comparison of different environmental aspects for the same product. The figures below show the impacts considered in this study, namely:

- Climate change, expressed as kilograms of CO2 equivalent for a 100-year time-horizon (GWP100)
- Water use – volume of water in m³
- Water pollution impacts on ecosystems due to eutrophication in kilograms of Phosphorus-input equivalent
- Water pollution impacts on human health – exposure to pollution in Chemical Toxicity Units (standardised units of toxicity of different substances, the same principle as e.g. CO₂-equivalence)
- Air pollution as kilograms of PM2.5-equivalent emissions
- Land use, in terms of kilograms of soil organic carbon displaced, serving as a proxy for land-use intensity
- Resource use (fossil fuels, minerals, renewable materials) in kilograms of Sb-equivalent.

All impacts used follow the ILCD 1.0.8 2016 recommended methodology, except for water use, for which a simple summation of water inputs from products’ life cycle inventories is used due to lack of an appropriate indicator.

Thus, the chosen indicators allow us to assess both the environmental impacts of the products under scrutiny, as well as their burdens in terms of resource use.
2.3.1 Life-cycle impacts on a per-functional unit basis

Figure 2.2 summarises the life-cycle impact assessment results for one use of each product considered (inclusive of washing).

At first glance, several products exhibit negative (i.e. avoided) impacts. This is due to the nature of the Ecoinvent consequential model’s treatment of co-products. Upon inspection of life-cycle inventories of offending products, it was found that this negative result is dominated by avoided burdens due to incineration of waste, where the energy generated from waste would be treated as displacing energy production from primary fuel sources. However, two issues must be kept in mind when interpreting such results:

- Impacts with a negative sign in the life-cycle inventories do not mean that the use of the reference product has caused a positive impact on the environment for a particular impact category. Rather, through inclusion of the use of co-products (such as waste heat) in the system boundary of the product, this use has displaced production of an input somewhere else in the economy (e.g. energy production from primary sources).

- The strong influence of incineration is sensitive to the assumptions made on incineration shares in the end-of-life of the products considered.

Figure 2.2 Impacts of products considered on a per-use basis (inclusive of washing)

<table>
<thead>
<tr>
<th>Product group</th>
<th>Product</th>
<th>Climate change</th>
<th>Water use</th>
<th>Water pollution - ecosystems</th>
<th>Water pollution - human health</th>
<th>Air pollution</th>
<th>Land use</th>
<th>Resource use</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cigarette butts</td>
<td>SUP</td>
<td>100%</td>
<td>3%</td>
<td>100%</td>
<td>81%</td>
<td>27%</td>
<td>13%</td>
<td>-53%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>54%</td>
<td>100%</td>
<td>74%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td>Drinks bottles</td>
<td>SUP</td>
<td>49%</td>
<td>7%</td>
<td>100%</td>
<td>2%</td>
<td>11%</td>
<td>6%</td>
<td>6%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>100%</td>
<td>100%</td>
<td>37%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>1%</td>
<td>5%</td>
<td>1%</td>
<td>1%</td>
<td>1%</td>
<td>2%</td>
<td>0%</td>
</tr>
<tr>
<td>Cotton buds</td>
<td>SUP</td>
<td>100%</td>
<td>95%</td>
<td>100%</td>
<td>100%</td>
<td>16%</td>
<td>2%</td>
<td>38%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>57%</td>
<td>100%</td>
<td>65%</td>
<td>67%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>2%</td>
<td>0%</td>
<td>0%</td>
<td>1%</td>
<td>0%</td>
<td>0%</td>
<td></td>
</tr>
<tr>
<td>Sanitary towels</td>
<td>SUP</td>
<td>100%</td>
<td>95%</td>
<td>62%</td>
<td>55%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>42%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>28%</td>
<td>33%</td>
<td>25%</td>
</tr>
<tr>
<td>Wet wipes</td>
<td>SUP</td>
<td>100%</td>
<td>6%</td>
<td>63%</td>
<td>39%</td>
<td>100%</td>
<td>9%</td>
<td>118%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>8%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>31%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>13%</td>
<td>0%</td>
<td>5%</td>
<td>27%</td>
<td>14%</td>
<td>2%</td>
<td>9%</td>
</tr>
<tr>
<td>Cutlery</td>
<td>SUP</td>
<td>29%</td>
<td>12%</td>
<td>4%</td>
<td>1%</td>
<td>10%</td>
<td>3%</td>
<td>53%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>6%</td>
<td>8%</td>
<td>-12%</td>
<td>19%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>23%</td>
<td>22%</td>
<td>94%</td>
</tr>
<tr>
<td>Straws</td>
<td>SUP</td>
<td>71%</td>
<td>0%</td>
<td>100%</td>
<td>-3%</td>
<td>8%</td>
<td>-2%</td>
<td>-3%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>100%</td>
<td>100%</td>
<td>19%</td>
<td>14%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>8%</td>
<td>7%</td>
<td>22%</td>
<td>100%</td>
<td>49%</td>
<td>29%</td>
<td>12%</td>
</tr>
<tr>
<td>Stirrers</td>
<td>SUP</td>
<td>100%</td>
<td>-1%</td>
<td>100%</td>
<td>-5%</td>
<td>6%</td>
<td>-2%</td>
<td>-20%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>20%</td>
<td>9%</td>
<td>-5%</td>
<td>23%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>81%</td>
<td>100%</td>
<td>15%</td>
<td>100%</td>
<td>24%</td>
<td>22%</td>
<td>30%</td>
</tr>
<tr>
<td>Food containers</td>
<td>SUP</td>
<td>100%</td>
<td>8%</td>
<td>71%</td>
<td>50%</td>
<td>43%</td>
<td>9%</td>
<td>20%</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>53%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>17%</td>
<td>11%</td>
<td>29%</td>
<td>57%</td>
<td>11%</td>
<td>7%</td>
<td>11%</td>
</tr>
<tr>
<td>Drinks cups and lids</td>
<td>SUP</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
</tr>
<tr>
<td></td>
<td>MU</td>
<td>13%</td>
<td>21%</td>
<td>10%</td>
<td>47%</td>
<td>15%</td>
<td>8%</td>
<td>60%</td>
</tr>
</tbody>
</table>

Note(s): Results are relative magnitudes compared to worst-performing product (100%). Red data bars represent avoided impacts.

The results indicate that single-use plastics are generally worse-performing in terms of the climate change impact category, with the noted exception of SUNP Drinks bottles (i.e. glass & aluminium). Their SUNP reference product is an average of both aluminium cans and glass bottles, and closer inspection shows that glass strongly dominates the result, chiefly due to the impacts from raw materials acquisition and
manufacturing, which is in-line with other studies comparing drinks bottles packaging, though such results depend strongly on the detail and specific conditions reflected by modelling of end-of-life (Simon, Ben Amor & Foldenyi, 2015). SUNP straws (made of paper) are also worse-performing within their product group due to the raw materials and manufacturing stages, where use of heavy fuel and diesel oils are common such as for boiler operations, as well as pulp bleaching and wood production (Mourad et al., 2014).

For the water use impact category, SUNP items generally have larger impacts. As SUNP products are bio-based, we can expect larger water burdens at the raw materials stage. The influence of bio-based materials is strongly evident for both sanitary items where burdens from cotton production dominate the results and lead to a near-tie in impacts between alternatives, showing that the impacts of their plastic components are negligible compared to their bio-based inputs.

SUP items are the worst performers in the air quality impact category, except for SUNP items with paper or wood as inputs, where manufacturing and raw materials burdens have the most significant contribution. The generally larger air quality impacts (represented by particulate matter) of bio-based plastics has recently been established in a JRC review screening alternative feedstocks for plastics production, also evidencing higher water pollution impacts from bio-based materials with feedstocks entailing proportionally larger chemical emissions to water such as due to fertiliser and pesticide inputs (Nessi et al., 2018), as also seen in results herein.

The land-use impact category is mostly dominated by SUNP items. We can expect products with bio-based inputs to be worse-performing in this impact category due to bio-based products’ higher requirements for land.

Impacts with a negative sign aside (explained at beginning of this section), the resource use impact category has SUPs being the worst performers. This can be explained at the raw materials stage due to the strong fossil fuel intensity of the plastics sector.

### 2.3.2 Upscaling to the whole-market level

The prior comparison of life-cycle impacts at the product level allows for identifying the differences in impacts between products within the same product group, so as to establish whether non-plastic alternatives are indeed better-performing from an environmental and resource-use perspective.

Following this, we also look into the effects of the total environmental and resource use burdens at the level of the entire EU markets for different products. This allows us to identify priority products for possible policy interventions.
Figure 2.3 Life-cycle impacts of all products considered, upscaled to the whole-market level where estimates of total market volumes are available

<table>
<thead>
<tr>
<th></th>
<th>Climate change</th>
<th>Water use</th>
<th>Water pollution - ecosystems</th>
<th>Water pollution - human health</th>
<th>Air pollution</th>
<th>Land use</th>
<th>Resource use</th>
<th># uses projected (million)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>kg CO2-Eq</td>
<td>m3</td>
<td>kg P-Eq</td>
<td>CTUh.m3.yr</td>
<td>kg PM2.5-Eq</td>
<td>kg Soil O.</td>
<td>kg Sb-Eq</td>
<td></td>
</tr>
<tr>
<td>Cigarette butts</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>627561</td>
</tr>
<tr>
<td>SUP</td>
<td>261 690 863</td>
<td>2 241 636</td>
<td>154 488</td>
<td>2 403 610 734</td>
<td>196 498</td>
<td>399 379 619</td>
<td>4 074</td>
<td></td>
</tr>
<tr>
<td>SUNP</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Drinks bottles</td>
<td></td>
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<td></td>
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<td></td>
<td></td>
<td>73550</td>
</tr>
<tr>
<td>SUP</td>
<td>16 481 764 767</td>
<td>36 034 305</td>
<td>27 566 330</td>
<td>91 972 819 961</td>
<td>7 541 290</td>
<td>10 602 301 600</td>
<td>621 492</td>
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<td>SUNP</td>
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<td>Cotton buds</td>
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<td>41 440</td>
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<td>28 907 645</td>
<td>8 403 238</td>
<td>15 876</td>
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<td>19 637</td>
<td>228 733 699</td>
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<td>SUNP</td>
<td>34 466 804</td>
<td>17 833 766</td>
<td>22 959</td>
<td>442 290 094</td>
<td>50 162</td>
<td>605 696 414</td>
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<td>32</td>
<td>2</td>
<td>27 408</td>
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<td>Crisp packets</td>
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<td>97 929</td>
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<td>838 823</td>
<td>45 711</td>
<td>353 279 817</td>
<td>140 345</td>
<td>239 106 160</td>
<td>2 995</td>
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<td>Sanitary towels</td>
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<td>19 790</td>
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<tr>
<td>SUP</td>
<td>229 478 641</td>
<td>6 705 150</td>
<td>91 485</td>
<td>15 432 300 099</td>
<td>205 196</td>
<td>12 424 876 795</td>
<td>454</td>
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<tr>
<td>SUNP</td>
<td>12 744 689</td>
<td>776 455</td>
<td>9 798</td>
<td>353 016 936</td>
<td>12 226</td>
<td>48 355 675</td>
<td>313</td>
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<td>Wet wipes</td>
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<td>43 382</td>
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<td>SUP</td>
<td>454 938 118</td>
<td>15 432 246</td>
<td>4 542 424</td>
<td>2 548 685 155</td>
<td>3 836</td>
<td>897 750 449</td>
<td>9 096</td>
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<td>Cutlery</td>
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<td>79 463</td>
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<td>SUP</td>
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<td>1 075 991</td>
<td>95 826</td>
<td>586 818 547</td>
<td>155 514</td>
<td>282 207 917</td>
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<td>144 763</td>
<td>11 819</td>
<td>404 942 674</td>
<td>252 733</td>
<td>349 778 008</td>
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<td>MU</td>
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<td>79 551</td>
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<td>81 049 801</td>
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<td>8 685 411</td>
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<td>22 112</td>
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<tr>
<td>SUP</td>
<td>261 635 652</td>
<td>7 479</td>
<td>1 016 49</td>
<td>13 059 546</td>
<td>60 795</td>
<td>11 110 197</td>
<td>34</td>
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<td>Stirrers</td>
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<td>1 600 153</td>
<td>20 560 303</td>
<td>95 712</td>
<td>17 491 344</td>
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<td>Food containers</td>
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<td>31 781</td>
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<tr>
<td>SUP</td>
<td>757 668 813</td>
<td>2 429 489</td>
<td>196 631</td>
<td>2 521 508 865</td>
<td>412 631</td>
<td>412 775 373</td>
<td>3 660</td>
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<td>SUNP</td>
<td>597 908 650</td>
<td>31 766 630</td>
<td>322 887</td>
<td>5 891 797 905</td>
<td>1 116 445</td>
<td>5 145 185 025</td>
<td>21 334</td>
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<td>MU</td>
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<tr>
<td>Drinks cups and lids</td>
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<td>21 601</td>
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<tr>
<td>SUP</td>
<td>752 629 090</td>
<td>103 11 640</td>
<td>726 116</td>
<td>5 452 485 910</td>
<td>580 904</td>
<td>3 190 583</td>
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<tr>
<td>SUNP</td>
<td>841 473</td>
<td>20 358</td>
<td>578</td>
<td>20 399 151</td>
<td>757</td>
<td>2 238 305</td>
<td>28</td>
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</tbody>
</table>

Note(s): Projected number of uses for each product considered as per Strategy on Plastics Impact Assessment assumptions (red data bars).

Impacts of products upscaled to their total number of projected uses (in million units of impact for each category; blue data bars).

Data bars represent relative ranking between products for each impact category (i.e. single ranking per column).

Drinks bottles (given in bold) exceed all other products’ impacts by at least one order of magnitude and are excluded from data bars for visual clarity.

Blank lines represent missing market data which does not allow upscaling.

Figure 2.3 gives the impacts of all products considered based on projected number of uses from the Strategy on Plastics Impact Assessment (where such are available). The lack of data for upsaling all products limits the assessment. However, some conclusions can be drawn from Figure 2.3, as well as reflecting on the product-level impacts from Figure 2.2.

It is immediately evident that number of uses does not necessarily correlate with magnitude of impact. This is most strongly true for cigarette butts - while exceeding all other products in number of uses, they have

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106 We are grateful for the contributions of Eunomia and ICF, who conducted the Strategy on Plastics Impact Assessment.
relatively modest market-level environmental impacts. However, LCA cannot address the health impacts of cigarette use, nor is it currently able to capture marine litter impacts of discarded cigarette butts. The present analysis shows that LCA environmental impacts are not a primary concern for cigarette butts and that efforts should rather focus on their health and possibly marine litter impacts.

Drinks bottles surpass all other products in impacts by approximately an order in magnitude, even though their number of uses is exceeded by other products. Inspection of LCA results shows that this is primarily due to impacts of raw material manufacturing and their fossil fuel intensity. However, these results should be interpreted with care. Results on a per-functional unit basis illustrate that SUNP items greatly surpass SUP drinks bottles in impacts due to the intensity of raw materials production. Though these results are very sensitive to assumptions on recycling and reverse logistics and should be interpreted in this light, the data herein shows that there is no clear-cut case for replacing SUP drinks bottles with non-plastic alternatives. However, the lesser per-functional unit impacts of multi-use drinks bottles suggests that promotion of reuse and refill at least for water consumption (such as through municipal refill schemes) is an avenue for lessening the impacts of drinks bottles use, as has been found in other studies comparing tap and bottled water systems (e.g. Garfi et al., 2016).

Finally, food containers illustrate that the general trend of bio-based products having higher impacts on a per-functional unit basis holds at the market level. Though SUP and SUNP alternatives have roughly equal market shares, SUNP impacts exceed those of SUP for all categories apart from Climate change (greenhouse gas emissions). Inspection of life-cycle inventories shows that this is mostly attributable to the large impacts of paper production (incl. raw material production).

2.4 Socio-economic footprint of the sector

The plastics industry operates across approximately 60,000 companies in Europe (Plastics Europe, 2016), representing a major sector of Europe’s economy. In Europe, 1.5 million jobs are directly linked to the plastics industry (Plastics Europe, 2016). These include 30,000 jobs within the plastics recycling industry (Plastics recyclers Europe, 2016) and 167,000 in the production phase (Plastics Europe, 2012). The plastic industry’s turnover reached €340 billion in 2015 and today is the 7th European largest industry in terms of value added (Plastics Europe, 2016). In addition to its economic relevance in Europe, the plastics industry largely contributes societal benefits through its various applications and properties. However, plastics are also highly damaging to the environment and there are costs of inaction. According to UNEP (2014), environmental damage to marine ecosystems from plastics amounts to $13 billion per year, including financial losses incurred by fisheries and tourism as well as time spent cleaning up beaches (UNEP, 2014). Figure 2.4 illustrates global natural capital costs associated with plastics from a number of consumer good sectors.
Recycling represents an important sector with associated economic and social benefits. As waste is turned into raw materials, recycling can contribute to job creation, economic growth, enhance competitiveness, foster innovation and secure access to critical resources (European Environment Agency, 2011). In addition, the direct employment opportunities associated with recycling include low-skilled work (European Environment Agency, 2011; Plastics recyclers Europe, 2016), therefore offering potential improvements in social inclusion and poverty alleviation. The main direct jobs associated with recycling are found within sorting, separation and collection, while indirect jobs are found within construction of recycling facilities, manufacturing of equipment, research and innovation, as well as management related jobs (Plastics recyclers Europe, 2016).

The European bioplastics industry presents opportunities for employment generation, with potential improvements in the development of rural areas. In 2013, there were 23,000 jobs associated with this industry in Europe. Within the industrial biotechnology sector, bioplastics has the second largest share, after bio-based chemicals, for direct employment generation in Europe, comprising 23% of total employment within the sector (Debergh, Bilsen and Van de Vekve, 2016).

2.5 Trends in environmental pollutant releases

Many sectors of the economy use and/or create pollutants during production processes, with the potential to damage human health and the wider natural environment. Data on pollutant releases is published in the European Pollutant Release and Transfer Register (E-PRTR\(^{107}\)). Data is classified by NACE code of the emitter, pollutant, location (including country), year, release medium and volume, and as such it is possible to track its evolution over time.

In the analysis below, we present data in volume terms, but also environmental impacts. These are calculated by applying coefficients reflecting the toxicity of different pollutants, taken from ReCiPe2016 LCIA, according to whether they were released via air, water or land. This allows the summation of different

pollutant based upon the impact that they have on human health (measured in disability-adjusted life years, DALYs) and ecosystem health (measured as disappeared species per year, species.years).

A higher proportion of data on releases is properly classified (in terms of pollutant) in the E-PRTR data on rubber & plastics than is the case for most other sectors in earlier years (around 25%), although this falls steadily over time (see Figure 2.5).

Figure 2.5 Sectoral observations on rubber & plastics available in E-PRTR

Figure 2.6 The health impacts of pollutant releases from the rubber & plastics sector
The impacts upon human and ecosystem health from rubber & plastic sector pollutant releases mirror largely each other, although in some years ecosystem health impacts are relatively worse, reflecting higher releases of non-CO2 pollutants. There was a spike in pollutant releases in 2010, reaching 290 DALYs. This was driven in part by an increase in CO2 releases, but also 300 tonnes of trichloroethylene (see Figure 2.7). Over the period from 2011 to 2016, health impacts rose steadily, reaching 278 DALYs in 2016 across the EU as a whole (see Figure 2.6).

Poland contributes almost all of the recorded pollutant releases associated with the sector; cumulative human health impacts were 2,538 DALYs over 2007-16, while Belgium had the next highest human health impact, at only 43 QALYs cumulatively over the same period. This suggests that Poland is classifying or recording data on pollutant releases in this sector in a different way to all other Member States; the releases are associated with energy generation, suggesting that there may be more on-site energy generation in Poland (or, at least, Poland is recording this release while other Member States are not).

Figure 2.7 Non-CO2 release volumes from the rubber & plastics sector
3 Current trajectory – direction of travel

3.1 Research questions

- The evolution in consumers and investors demand for increased transparency on environmental performance?
- Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?
- What has driven changes over time (post-2000), and what changes are expected to occur in the future?
- How have the answers to all of these questions changed over time, and how are they forecast (modelled) to change in the future?
- How do SMEs differ from other firms in their answers to all of these questions?
- Are there examples of striking differences to any of these questions between Member States (and if so, why)?

3.2 Overview of trends in the plastics sector

Trends in plastics production and consumption imply that an increasing amount of oil will be dedicated to the plastics industry in the future. However, the scarcity of oil and the volatility of its price represent an obstacle to the development of petroleum-based plastics and call for the development of alternative feedstocks (WEF, EMF and McKinsey & Company, 2016). Alternatives to conventional plastics, such as bioplastics, recycling and CO₂ capture, are options which are increasingly being explored.

The Ellen MacArthur Foundation predicts that the plastics sector will account for 20% of total oil consumption by 2050. The environmental impacts associated with plastics production are expected to increase as well and, if current trends continue, the plastics industry is expected to account for 15% of global annual carbon budget by 2050, from 1% today (WEF, EMF and McKinsey & Company, 2016).

While the sector continues to be closely coupled to fossil fuel production, this also links it to climate change objectives. Closing the loop on plastic production, including through prevention, reduction, reuse and recycling activities has the potential to dramatically reduce the emissions of the sector. A study by Zero Waste Scotland estimated that circular economy activities in the Scottish economy could reduce territorial emissions by 11 million tonnes CO₂e per annum by 2050 compared to business as usual (Pratt and Lenaghan, 2015).

The bioplastics market is predicted to grow in the near future, with predictions of increased production capacity of approximately 6.11 million tonnes by 2021.

If current waste generation and management trends for all plastic continue, 12,000 million tonnes of plastics will be accumulated in landfills or leaked into the natural environment by 2050 (Geyer, Jambeck and Law, 2017). Furthermore, if no action is taken, it is expected that there will be almost 250 million metric tonnes of plastic in the oceans by 2025 (Ocean Conservancy and McKinsey & Company, 2015). The Ellen MacArthur Foundation estimates that by 2025, plastic litter in the oceans is expected to reach 1 tonne for every 3 tonnes of fish in a business-as-usual scenario, and to exceed the amount of fish (by weight) by 2050 (WEF, EMF and McKinsey & Company, 2016).

A growing accumulation has consequences on human activities which rely on these environments. For instance, the tourism sector can be severely impacted by plastic leakage on coastal and marine environments due to the consequent loss of aesthetic value. Moreover, marine litter causes harm to human and wildlife health (Werner et al, 2016). Waste generation and mismanagement are the main land-based sources of plastic litter.

Bioplastics are increasingly being used for packaging. However, replacing conventional plastics with bioplastics does not guarantee a solution to the marine litter issue, nor to low levels of reuse and recycling. Biodegradable plastics require specific temperatures and time to properly degrade, these are conditions that cannot be guaranteed in the environment (Surfrider Foundation et al, 2017). In addition, bio-based and biodegradable plastics need appropriate infrastructure for recycling which does not always match the
process used for conventional plastics. Nevertheless, bio-plastics are often included in the existing recycling process, generating concerns for the plastic converters industry as the quality of recyclates is not ensured. When landfilled or incinerated, bioplastics are associated with greenhouse gas emissions. Bioplastics might also act as a perverse incentive and lead to increase littering trends as consumers tend to consider them as easily biodegradable products (EPA Network, 2017; Surfrider Foundation et al, 2017).

Socio-economic benefits associated with the plastics sector are expected to be influenced by future trends in recycling, eco-design and bioplastics in the EU. Labour demand is expected to increase as a response to changes in chemical and manufacturing processes of plastics. For instance, redesigning plastics requires new skills and technologies (European Commission, 2011).

Growth in sub-sectors including waste management and plastic conversion have the potential to create new SMEs and employment opportunities, contributing to socio-economic objectives in the SDGs. Recycling is expected to contribute to employment by generating 50,000 new direct jobs by 2020. These are then expected to impact the wider economy and generate an additional 75,000 indirect jobs in Europe. The jobs derived from recycling have the potential to reach 80,000 direct jobs and 120,000 indirect jobs by 2025. A Club of Rome study showed that employment activities linked to the circular economy, including activities favouring reuse and recycling, could create 75,000 new jobs in Finland, 100,000 in Sweden, 400,000 in Spain and 500,000 in France (Wijkman, Skånberg and Berglund, 2016).

Increasing recycling of plastic packaging can bring significant benefits in terms of reduced environmental costs. Trucost investigated how environmental costs would change if plastic packaging recycling reached a target of 55% and landfilling was reduced to 10% in both Europe and North America. The results of the study show that the environmental costs of plastic use would drop by US $4.8 billion per year. This includes an environmental benefit of material and energy recovery of US $3.9 billion (Trucost, 2016). Jobs associated with the bioplastics industry are expected to reach 200,000 by 2030 (European Bioplastics, 2017).

Indirectly, the plastics sector also could contribute to targets relevant to other sectors in the economy. For instance, innovation in polymers could facilitate lightweighting in transport (9.1). Plastic packaging is often cited as contributing positively towards food waste (12.3) but trends towards over-packaging threaten to undermine these benefits, with packaging often linked to wastefulness and over-purchasing (include ref).

Developing comprehensively sustainable solutions for packaging food will be a key challenge for the sector in the future.

3.3 The impact of changing consumer preferences

Plastics properties can also influence consumption levels by attracting or discouraging consumers. In the first case, plastic properties such as lightweight or food preservation increasingly encourage the application of the material due to the associated benefits. Plastics can be used to lightweight motor vehicle chassis in order to reduce fuel use and emissions, likewise plastics are often lighter than alternative packaging materials such as glass or metals which can reduce emissions in logistics (Thompson et al, 2009). Despite such benefits, plastics are associated with a throw-away society and high levels of waste, especially when it comes to single-use plastics and packaging. Growth in plastics applications in packaging has come as a result of a global shift to single-use containers (Geyer, Jambeck and Law, 2017). This is particularly the case of food packaging where plastics are widely used to protect, market, and deliver food products. The characteristics of food packaging, such as size and shape, have an impact on consumers’ purchasing choices (Aschemann-Witzel et al, 2015). However, food products are often considered over-packaged by consumers (INCPEN, 2008) and such feature has been reported as a reason to avoid purchasing (INCPEN, 2008; Which?, 2011). By 2020, Europe is estimated to consume more than 900 billion items of packaged food and drink annually (Smithers Pira, 2015). Plastic packaging has been found to be associated with the migration of harmful chemicals such as endocrine disruptors (Karamfilova and Sacher, 2016). These chemicals are largely used in food contact materials and plastic packaging in particular. Evidence of health impacts associated to chemicals migration into food and beverages can have an impact on consumers’ purchasing choices.

Plastics products manufactured with recycled content are often perceived of lower value (Rajendran et al, 2012; WEF, EMF and McKinsey&Company, 2016). This translates into lower demand for and consumption of recycled plastics (Rajendran et al, 2012). Polymer mix and contamination of other materials and additives lower the technical and economic value of secondary plastics (Geyer, Jambeck and Law, 2017). Packaging is
generally recycled through mechanical open-loop recycling and reprocessed into lower value products, typically non-packaging (WEF, EMF and McKinsey&Company, 2016). The quality gap between virgin and recycled resin can be closed through technological advances which support closed-loop or primary recycling. Closed-loop recycling reprocesses plastics into products with the same properties, while secondary recycling lowers the properties of the reprocessed products (Hopewell, Dvorak and Kosior, 2009). Technological advances aimed at bridging the quality gap can have important positive impacts on consumption levels and therefore on recycling rates.

Issues related to plastics, such as marine and terrestrial litter, as well as toxic additives are contributing to making consumers more and more aware of the environmental consequences of plastic products (Worldwatch Institute, 2015). As a consequence, consumers are becoming increasingly aware of their responsibility of making more environmentally-friendly choices at the purchasing stage (Karlaite, 2016). In the case of packaging, purchasing choices have generally been based on factors such as aesthetics, convenience of use, and design. However, a preference for ethical or green products has lately been observed among consumers (Smithers Pira, 2017). As a response to such growing consumer environmental preferences, coupled with the demand for increased transparency, alternatives to conventional plastics are being explored (Green Dot Bioplastics, 2017) as well as zero-packaging or reusable options (Schweitzer and Janssens, 2018), with inevitable consequences on demand.

3.3.1 Bioplastics
Consumers growing environmental preferences are contributing to an increasing demand for bioplastics due to the green-sounding credentials of the bioplastics industry (Surfrider Foundation et al, 2017).

3.3.2 Product substitution
Consumers’ preference for alternative more sustainable materials can impact demand for plastics products (e.g. packaging, bags). In particular, reusable packaging is attracting consumers’ interest which can lead to lower demand for single-use packaging.

3.4 Sustainability in the plastics sector
It is clear that the current trends in the production and use of plastics are not sustainable from an environmental perspective. Substantial analysis and policy work has been carried out in recent times, by the European Commission and others, to illustrate the past and ongoing impact of plastics on the natural environment and human health.

However, the plastics sector is heterogenous, as our previous analysis has shown. The implications of this are that, while a sustainable plastics sector might share many characteristics across products, the policy landscape required to realise this is varied. The European Commission’s European Strategy for Plastics (2018) demonstrates this clearly; a range of product-specific policies are required, rather than a single cross-industry measure.

One example of this can be found in the success in reducing the use of carrier bags across Europe. This trend is widespread, yet it has been achieved through different policies in different Member States, including voluntary agreements with the retail sector (Germany, Austria), levies on bags (UK, Netherlands) or outright bans on non-sustainable bags (France, Italy).

For this reason, the European Strategy for Plastics concentrates primarily on broad aims, and tools to achieve those. It focuses on;

- Facilitating higher rates of re-use and recycling;
- Reducing waste and preventing the leakage of plastic waste into the environment;
- Driving innovation in sustainable solutions.

It is clear that a sustainable plastics industry must meet these goals; less extraction of raw materials (including, but not limited to, oil) requires changes in the way that plastics are used, as well as produced, and waste from the sector must be drastically reduced but also much better contained.
3.4 Dependence on investment and future risks

3.4.1 Crude oil
The price of plastics feedstock, such as crude oil, determines production costs and therefore the price of plastics products (Weinhagen, 2006; Wrap, 2008). The price of recovered plastics is determined in turn by the price of virgin plastics. Environmental issues such as climate change represent a challenge to plastics produced from fossil fuels as clean-energy and climate-change call for emission reductions, eroding demand for oil and therefore for the products whose production is based on oil (OECD, 2011) (Council on Foreign Relations, 2015).

3.4.2 Packaging
Increasing demand for green products and eco-efficiency requirements are linked to climate change and resource scarcity issues. Increasing awareness of environmental issues represents one of the factors influencing packaging demand. Sustainable packaging is increasingly being demanded, not only by consumers, but by retailers and manufacturers as well, increasing the pressures to introduce sustainable alternatives to plastic, or to increase its recyclability (Smithers Pira, 2017). Packaging is also often considered as a way to prevent food waste. This can lead to an increase in the use of plastic packaging as food waste becomes an increasingly important issue. Nevertheless, evidence of health impacts of food contact materials, such as chemical migration, may have the reverse effect. Permanent materials such as glass and steel represent better and less risky food contact materials and are often preferred to plastic packaging (Muncke, 2017). The elimination of specific chemicals is also being explored, as well as the possibility to invest in green chemistry (Smithers Pira, 2017).

3.4.3 Automotive
EU legislation sets CO₂ emission reduction requirements for cars to improve fuel economy. These targets are leading to a growing trend among auto manufacturers to use plastics in order to achieve better fuel efficiency levels (Germany Trade & Invest, 2016). The use of lightweight material components, such as plastics, is considered one of the key measures to reduce vehicles fuel consumption (Trucost, 2016).

3.4.4 Construction
Plastics is used in building construction to increase energy efficiency (through plastics-derived thermal insulation materials) (Germany Trade & Invest, 2016).

3.4.5 Waste management
Plastic debris in the ocean and marine litter issues are pushing investments towards extending municipal waste collection services and improving waste management practices (Trucost, 2016). Current challenges associated with the recyclability of plastics, as well as reusability and the use of recycled content, warrant investments to be directed towards proper waste management infrastructure.

3.5.6 R&D
There has been an increase in research in plastics innovation, the recycling subsector, and sustainable packaging, and such activity is likely to increase given the recent publicity around single use plastics and the likelihood of policy to reduce consumption of such materials.

3.5 The impact of policy
Environmental policies directly and indirectly augment the interactions of the plastics sector and its value chain with the environment. At the European level, there are a number of policies which influence the production of plastics, but particularly in the context of the standardisation of products placed on the common market. Having said this, as much of the European legislation is governed via the principle of subsidiarity, there is a significant level of diversity in how policies are interpreted at national, regional and municipal levels. This is evidenced in the heterogeneity of waste management systems and supporting instruments which exist for the collection of packaging waste. In the context of the Circular Economy Action
Plan, the European Plastics Strategy (COM(2018)28final) provides the first European level strategy on a specific material, and provides a road map for policy guidelines on the sector for the future.

Table 3.1 Overview of key environmental policy in the EU relevant to the plastics sector

<table>
<thead>
<tr>
<th>Value chain</th>
<th>European Policies</th>
</tr>
</thead>
</table>
| Raw materials | Resource efficiency strategy  
Circular Economy Package  
Strategy for Plastics in a Circular Economy |
| Manufacturing | Regulation on the registration, evaluation, authorisation and restriction of chemicals (REACH)  
Eco-design Directive  
Climate change policies (ETS)  
Circular Economy Package  
Strategy for Plastics in a Circular Economy  
Extended Producer Responsibility schemes  
Microbeads bans |
| Consumption   | Directive on Packaging and Packaging waste  
Regulation on classification, labelling and packaging  
Sustainable packaging guidelines  
Regulation on food contact materials  
Circular Economy Package  
Strategy for Plastics in a Circular Economy  
Market based instruments targeting consumption: deposit-refund schemes, charges on plastic bags, disposable cutlery and other one-use items |
| Disposal      | Waste Framework Directive  
Water Framework Directive  
Packaging and Packaging Waste Directive  
End of Life Vehicles (ELV) Directive  
Port Reception Facilities Directive  
Circular Economy Package  
Strategy for Plastics in a Circular Economy  
Market-based instruments: landfill taxes/bans |

In January 2018 the European Commission adopted the Strategy for Plastics in the Circular Economy. To an extent, this Communication outlines the EU’s future policy priorities for the plastics sector. Though in many areas it includes guidelines for future initiatives, assessments and policy developments, without defining
proposals in concrete terms. The most notable proposal in the strategy is to “ensure that by 2030 all plastics packaging placed on the EU market can be reused or recycled in a cost-effective manner”.

In addition to Strategy for Plastics in the Circular Economy, and as part of a wider package, several further legislative proposals were presented.

- A Communication on the Interface between Chemicals, Products and Waste
- A Monitoring Framework on the Circular Economy
- A new Directive on Port Reception Facilities

To a greater or lesser extent these proposals all have relevance to the plastics sector. The Interface Communication for instance identifies the challenges in improving the traceability of chemicals through product life-cycles and addressing the issue of legacy substances in recycled streams. This issue is particularly relevant to plastics where substances of concern and issues of legacy substances are more common. The monitoring framework provides a single instrument, supporting macro level metrics to assess the circularity of the EU economy, including specific indicators on plastic for both waste generation and the contribution of recycled materials to raw material demand.

As stated, many of the measures outlined in the Strategy for Plastics, but also other legislative proposals (e.g. Interface) remain undefined or only broadly defined. The full list of future EU measures to implement within the Strategy for Plastics, including an indicative timeline for implementation are provided in the Annexes to the strategy (COM(2018)28 final Annexes 1 to 3).

108 The example of DEHP in PVC is included in the interface Communication Staff Working Document (SWD(2018)20 final)
4  Future Policy Priorities

4.1  Research questions

- What are the links between these sectors and the Sustainable Development Goals and the different targets?
- What is stopping the development of ‘water tech’ sector in Europe? Is it a lack of relevant R&D, or a lack of investment? If the latter, how much investment are we talking about?
- Are there some differences and good practices to be learnt from non-EU countries?
- What are the opportunities for these sectors (including jobs, growth and investment opportunities) provided by the environment and by environmental policy?
- What are the potential business evolutions (business model, product specificities, etc.) driven by environmental policies or voluntary initiatives influenced by the environment in a given sector?
- How does environmental policy affect the links between the sector and the environment?
- How does the environment and environmental policy affect the link between these sectors and growth, jobs and investment?

4.2  Links between the sector and the Sustainable Development Goals

Activities throughout the plastic sector’s value chain can contribute both positively and negatively towards the Sustainable Development Goals and targets at the global and European level (see Figure 4.1). The future development of the sector will determine how it contributes towards the goals. A number of the SDGs targets are closely linked to the plastics sector, notably those which relate to waste management (12.4, 12.5), and marine pollution (6.3, 14.1). This reflects the reality that poorly managed plastic waste has become a sustainability challenge with global relevance.
Figure 4.1 Illustration of Sustainable Development Goals and Targets relevant to the plastic value chain

Mapping the SDGs against the plastics sector value chain

Increasing the use recycled material can provide an alternative feedstock, replacing virgin material inputs. As well as reducing sectoral emissions.

9.4.1 CO2 emission per unit of value added; 12.2 Sustainable management and efficient use of natural resources

+ve contribution

12.5 Reduce waste generation through prevention, reduction, recycling and reuse

12.6 Rationalise inefficient fossil-fuel subsidies

12.7 Promote public procurement practices that are sustainable. In accordance with national policies and priorities

12.8 Promote in waste management practices that contribute to employment by generating 50,000 new direct jobs (sorting, separation and collection) and 75,000 new indirect jobs in Europe by 2020

New activities in waste management and plastic conversion will create new businesses and jobs. Recycling is expected to contribute to employment by generating 50,000 new direct jobs (sorting, separation and collection) and 75,000 new indirect jobs in Europe by 2020

6.3 Improve water quality by reducing pollution and minimizing release of hazardous materials

8.4.1 Material footprint, material footprint per capita, and material footprint per GDP

8.5.1 Tourism direct GDP as a proportion of total GDP and in growth rate

14.1 Prevent and reduce marine pollution of all kinds, including marine debris

Plastics remain a major source of marine pollution. It is estimated, if no action is taken, there will be almost 250 million metric tonnes of plastic in the oceans by 2050.

A growth in the demand for bio-based plastics could place further pressure on agricultural land. An emerging issue concerns the contamination of soils with plastic, e.g., from films.

The sector remains closely coupled to fossil feedstocks and emissions are forecasted to increase. 90% of the plastics produced globally are derived from fossil fuels. If current trends continue, the plastics industry is expected to account for 15% of global annual carbon budget by 2050.

Deliberately added microplastics and toxic additives should be designed out of plastic products to reduce their release into the environment.

Measures to improve waste management and reduce the consumption of single use plastics will reduce the per capita consumption of plastics. Globally, consumption of plastics is forecasted to continue to grow.

Plastic leakage on coastal and marine environments can severely impact the tourism sector due to the loss of aesthetic value.
4.3 Increased recycled content in products

Increasing the use of recycled material can provide an alternative feedstock for some plastic products, replacing virgin material inputs. The advantages of this are efficiencies in material and energy use, as well as potential for improvements in waste management and markets for secondary materials. Possible disadvantages exist in a real, or perceived, reduction in the material quality of recyclates compared to virgin material. An increase in the rate of recycling may not translate into an increase in the recycled content of the collected product, for example where recyclates are diverted to different lower value applications (e.g. packaging to synthetic fibres) and/or where there is a net growth in the material output of the sector (i.e. Jevons paradox).

Changing sector priorities away from virgin inputs towards the management and conversion of plastic waste has the potential to generate employment as these activities are foreseen to be more labour intensive. Increasing recycled content in the manufacturing of plastics products can reduce emissions as less energy is needed per unit of output (Hopewell, Dvorak and Kosior, 2009; WEF, EMF and McKinsey&Company, 2016). Emissions can be reduced by 1.1-1.3 tonnes of CO₂ by recycling one additional tonne of plastics rather than producing it from virgin fossil feedstock (WEF, EMF and McKinsey&Company, 2016). According to a life cycle analysis, producing plastic bottles by using 100% recycled PET rather than virgin PET could reduce CO₂ emissions by 27% (WRAP 2008; Hopewell et al, 2009)(Thompson et al., 2009).

Plastics and plastic packaging in particular are made of different polymers and often contain additives and other chemicals which hamper their recyclability or present risks in terms of toxicity for specific applications (Hopewell, Dvorak and Kosior, 2009). For example food contact materials, and other applications such as toys, present a specific challenge in the context of plastic recycling because of a risk of chemical transfer - permanent materials such as glass or steel offer advantages for food contact materials.

A range of policy measures could be useful to increase the uptake of recycled plastic in plastic products. Increasing the uptake, quality and economics of recycling requires measures to be adopted on upstream design (e.g. eco-design) and downstream collection, sorting and reprocessing.

Extended Producer Responsibility (EPR) schemes present opportunities to incentivise the uptake of recycled plastics in plastic products through eco-modulation of fees. Fee modulation implies a diversified cost applied to producers according to the criteria of the products placed on the market. In addition to criteria based on weight and materials, fees can be modulated based on aspects of eco-design related to the level of recyclability and separability of products and their components as well as on the amount of recycled plastics.

Deposit-refund schemes on one-way plastic containers have been proven to significantly increase the rate of recycling (e.g. PULPA) as the containers (e.g. PET bottles) are returned to the producers and sent to recycling companies where new products are generated and transported back to the consumers.

In addition, the proposed Single-Use Plastics Directive includes targets for recycled content in PET bottles, more precisely at least 25% of recycled plastics by 2025 and 30% by 2030.

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**Modulated EPR measures – CONAI, Italy**

The Italian extended producer responsibility organisation (PRO) CONAI (Consorzio Nazionale Imballaggi) has introduced modulated fees within its EPR scheme from January 2018. Under the previous system, plastic packaging producers were charged a fixed rate according to the amount (in tonnes) of plastic packaging they put on the Italian market. In the new system, three different rates are established for recyclable industrial packaging, recyclable domestic packaging and unrecyclable packaging. Unrecyclable packaging is charged a higher rate, 228 EUR/tonne, compared to recyclable waste: €179/tonne and €208/tonne for industrial and domestic packaging respectively (CONAI, 2017). The objective of this new system is to incentivise packaging producers to design packaging which is easily recyclable, and to support the market for secondary plastics. Between 1997 and 2014 plastic packaging recycling rates increased from 9.6% to 109%

Environmental taxes are one type of market-based instrument which may be used to implement environmental policy priorities. These measures can “raise fiscal revenues while furthering environmental goals” (OECD, 2005) (World Bank, 2005). Environmental taxes are already present in a number of sectors including energy, transport, carbon and natural resources. On a political level growing attention has been given to environmental taxes, recognising their potential advantages in comparison to traditional ‘command and control’ regulations. Furthermore, specific environmental challenges, such as climate change, can be important drivers of such reforms (Withana, 2015). If well designed environmental taxes can bring a range of benefits which can support policy objectives in a range of areas.

Table 4.1 Potential benefits of introducing plastic tax

<table>
<thead>
<tr>
<th>Benefit</th>
<th>Examples/explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Financial</td>
<td>Budget contributions and macroeconomic stability</td>
</tr>
<tr>
<td></td>
<td>Mobilisation of revenues for policy priorities</td>
</tr>
<tr>
<td></td>
<td>Reduced requirements for budget expenditure (due to savings, e.g. via ESS)</td>
</tr>
<tr>
<td>Economic</td>
<td>Revision of price signals (internalised externalities, application of ‘polluter/user pays’ principle)</td>
</tr>
<tr>
<td></td>
<td>Catalyse innovation</td>
</tr>
<tr>
<td></td>
<td>Reduce growth distorting taxes</td>
</tr>
<tr>
<td>Social</td>
<td>Health (reduce pollution)</td>
</tr>
<tr>
<td></td>
<td>Employment creation (double dividend hypothesis)</td>
</tr>
<tr>
<td></td>
<td>Availability of natural resource/public goods for future generations</td>
</tr>
<tr>
<td></td>
<td>Poverty reduction / Address environmental impacts unfairly impacting the poor / Positive distributional effects from shifting taxation away from labour</td>
</tr>
<tr>
<td></td>
<td>Mobilise funds for pro-poor investments</td>
</tr>
<tr>
<td></td>
<td>Improve access to ecosystem services ESS</td>
</tr>
<tr>
<td>Environmental</td>
<td>Incentivise efficient resource use</td>
</tr>
<tr>
<td></td>
<td>Reduce harmful emissions/pollutants</td>
</tr>
</tbody>
</table>

A European wide plastic tax has been suggested as an instrument which could help to increase the update of recycled material in the plastics sector, as well as addressing other policy priorities (including budgetary issues). In January 2018, Budget Commissioner Günther Oettinger suggested a plastics tax\textsuperscript{110} as one of the ‘own resources’ for the EU Budget. In the Strategy for Plastics, it is also noted that “national and regional authorities should make better use of taxation and other economic instruments to reward the uptake of recycled plastics and favour reuse and recycling over landfilling and incineration.”

While some examples of taxes relevant to plastics exist for specific products (see example from Finland below), there are no cases of an economy wide tax on non-energetic fossil materials. The proposed Own Resource contribution would be directly proportional to the quantity of non-recycled plastic packaging waste generated in each Member State and reported each year to Eurostat. Member States would pay EUR 0.80/kg, which it is estimated could raise around EUR 7 billion per year (European Commission, 2018).

Packaging represents the biggest plastics application (39.9%) and products within this category are typically characterised by short lifetimes. This arguably suggest the application of a tax on plastic packaging. Nevertheless, narrowing the tax to this category presents limitations as many single-use plastic items do not belong to the packaging sector and would therefore remain unaddressed.

**Deposit Refund System and packaging tax, Finland**

A beverage packaging tax exists since 1994; since 2008 packaging producers who participate in a registered deposit refund system receive a complete exemption from the tax regardless of whether they use reusable or one-way containers in the scheme. Finnish deposit refund system operator, Suomen Palautuspakkaus Oy (PALPA), is the largest in the country. In 2015, it achieved average return rates for one-way packaging of between 89% to 95%, and government targets for recycling and re-use of all returnable packaging materials are 90% (Ettlinger, 2016).

### 4.4.1 Assessing the socioeconomic impact of plastics taxes

Three scenarios of a plastics tax were developed to assess the potential impacts as outlined in ‘Future policy priorities’. These scenarios are constructed in a stylized manner, to assess first-order effects of a plastics tax. The scenarios model imposition of a tax on various plastic products: 1) tax on non-recyclable plastic waste generation; 2) tax at production level; 3) tax on external imports of plastic sheets into the European Union. The tax base in these three scenarios differs, providing evidence of potential impacts across different targeted taxes. Scenario One examines taxation levied on generation non-recyclable waste. Scenario Two examines taxation levied on production of all plastics. Scenario Three is significantly more limited in ambition/scope, only imposing tax on external imports of plastic sheets. Economic and environmental effects of a plastics tax are likely to be highly dependent upon substitutability of other materials/elasticity of demand, and the associated cost. For this reason, sensitivity analysis of substitutability was employed across scenarios.

Scenario One uses the rate 0.5€ per kg of non-recyclable plastic waste, selected as an indicative magnitude of such a tax. Scenario Two and Three use the rate of 0.102€ per kg of plastic, following the suggested plastic packaging tax in EC (2016) ‘Study on Assessing the Environmental Fiscal Reform Potential for the EU28’.

The key results to examine from this modelling exercise are: 1) changes in plastic use; 2) socioeconomic outcomes; 3) magnitude of revenue raised from the tax.

The key result of the modelling is that aggregate level economic consequences of a plastics tax are likely to be minimal. Economic activity in production of virgin plastics would likely contract, but demand would be displaced to waste management and recovery of materials: key sectors to the circular economy. This analysis suggests that there is no significant net cost of taxation designed to reduce use of plastics.

Scenarios of taxation of plastic waste and production result in very similar effects: contraction in the plastics sector, significant tax revenue, and a small net positive impact on GDP and employment. Magnitude of impact is driven by size of the tax: the per kg tax on waste is almost five times higher than that for production and imports. The impacts of the border tax scenario are significantly smaller than the waste and production tax. This is because the tax is only applied to imports, not any EU domestic production, and only to plastic sheets.

The positive net effects on GDP and employment to 2030 in these scenarios is a result of tax revenue and changing composition of economic activity. All the revenue from the plastics tax is spent by government,
providing a demand stimulus\textsuperscript{111}. The results illustrate the trade-off between reducing plastics production and generating revenue through environmental taxation. Change in intermediate demand from the plastics sector to the waste management and materials recovery sector reduces imports of plastics and directs demand to economic activity within the EU. This increase in domestic activity is partly an artefact of assumption: whilst waste collection is necessarily a domestic activity, materials recovery and recycling is not, the EU exports a significant volume of waste for processing in other regions.\textsuperscript{112}

Demand response to the change in price of plastic use is the most important variable in the modelling. Three stylized substitution elasticities for plastic were used to adjust intermediate demand. Under assumption of fixed production behaviour, the tax has very limited impact on output of the plastics sector. Estimation of behavioural response to changes in price of plastic use is beyond the scope of this project, but given the importance of this variable it is an important subject for further study. The price elasticities of 0 and 1.5 were chosen as illustrative boundary-like examples, to examine the impacts across potential behavioural responses. The price elasticity of 0.5 represents an intermediate scenario, which enables examination of the extent to which effects are likely to be linear between boundary scenarios.

An important effect in the production tax scenario is production leakage. External exports from the EU28 contract by €667 million by 2030 in the first production tax scenario. Production outside of the EU increases, with the largest absolute increase in production being in China. In the first border tax scenario, EU domestic output increases, replacing otherwise imported products.

\begin{table}[h!]
\centering
\begin{tabular}{|c|c|c|c|c|c|c|}
\hline
Scenario & Price demand response (%) & EU28 GDP (%) & EU28 emp. ('000s) & NACE 22 Rubber & Plastics output (%) & NACE 22 Rubber & Plastics emp. ('000s) & Tax revenue (bn 2018€) \\
\hline
Waste tax & 0 & 0.048 & 59.5 & -0.02 & -0.91 & 5.40 \\
& 50 & 0.051 & 52.4 & -0.82 & -7.60 & 5.26 \\
& 150 & 0.056 & 37.8 & -2.41 & -21.08 & 5.00 \\
Production tax & 0 & 0.046 & 60.4 & -0.29 & -6.61 & 5.40 \\
& 50 & 0.05 & 55.5 & -0.97 & -12.61 & 5.36 \\
& 150 & 0.057 & 46.7 & -2.34 & -24.63 & 5.29 \\
Border tax & 0 & 0.004 & 5.0 & 0.03 & 0.15 & 0.39 \\
& 50 & 0.004 & 4.5 & -0.01 & -0.30 & 0.39 \\
& 150 & 0.005 & 3.9 & -0.11 & -1.21 & 0.39 \\
\hline
\end{tabular}
\caption{Macroeconomic impacts for different plastic tax scenarios, EU28}
\end{table}

Environmental impacts from these policies are limited, primarily due to the small impact on GDP (and therefore minimal changes in production across the economy). Production of plastics decreases slightly, the tax revenue is used to increase spending across all categories of government expenditure, proportional to baseline expenditure composition.

Collection of waste and materials recovery are both categorised under NACE 38 ‘Waste collection, treatment and disposal activities; materials recovery’. Wholesale of waste and scrap is found under NACE 47.6 ‘Other specialised wholesale’.

\textsuperscript{111} The tax revenue is used to increase spending across all categories of government expenditure, proportional to baseline expenditure composition.

\textsuperscript{112} Collection of waste and materials recovery are both categorised under NACE 38 ‘Waste collection, treatment and disposal activities; materials recovery’. Wholesale of waste and scrap is found under NACE 47.6 ‘Other specialised wholesale’.
leading to less demand for relevant raw materials (including oil), although most of the environmental benefit of this is felt outside of Europe (due to Europe relying on imports of oil). The primary environmental benefits within the EU come from the reduction in waste from lower use of plastics; this can be expected to reduce landfill and waste incinerations, leading to reductions in GHG emissions and other pollutants. However, such emissions are not well captured in either E3ME or the EXIOBASE data that has been used to quantify environmental impacts in other parts of this study, so it has not been possible to quantify such impacts.

Scenario Two is defined as a sector-wide production tax, no distinction is made between production using virgin vs recycled feedstock. If a production tax was introduced which differentiated by content of recycled feedstock, incentives would be stronger for sourcing strategies which promoted the circular economy.

4.5 Environmental impacts of a plastics tax

In addition to the economic modelling of plastics tax scenarios, environmental impacts have further been estimated via coupling E3ME and the EXIOBASE environmentally extended input-output database. As for economic results, all scenarios display very similar environmental impacts, with small variation with price demand response.

Acidification, eutrophication and photochemical oxidation potentials see falls in impacts from combustion emissions in industry but in the cases of acidification and eutrophication, these are respectively balanced and outweighed by emissions of NOx and NH3 from fertiliser/manure application, stemming from the growth of the agricultural sector (which is linked to rising food use rather than the effects of the modelled policy instruments). In the case of summer smog potential, decreases in impact drivers from combustion emissions are balanced by an increase in non-combustion emissions, as the modelling of these via EXIOBASE is linked to the (growing) gross output of the industrial sector.

Human toxicity potential shows the largest increase out of non-resource use environmental impacts on account of increasing industrial emissions of heavy metals and polycyclic aromatic hydrocarbons (PAH). Global warming potentials show the largest decreases in non-resource use impacts driven chiefly by falling CO2 emissions.

In the case of all environmental impact drivers apart from CO2 and resource use, technological assumptions are held fixed at EXIOBASE 2011 levels given lack of concrete data on future technological development. Hence, the observed increases in industrial emissions are likely overestimated, as well as to a lesser extent the increases in emissions from the agricultural sector, where even given fixed technologies, a more direct link between demand, production and manure/fertiliser emissions can be expected.

Overall, the results for environmental impacts show a weak overall response to the policy instruments modelled but rather reflect overall economic trends such as rising food use and respective growth in agricultural production. Where increased industrial emissions are seen, this is partly confounded by the assumption of unchanging production technologies toward the future for non-CO2 emissions.

Table 4.3 Environmental impact modelling results for different plastics tax policy options and price demand response scenarios. BAU=Business as usual. All results are given as 2030 % differences from values in 2018.

<table>
<thead>
<tr>
<th></th>
<th>BAU</th>
<th>Waste tax</th>
<th>Production tax</th>
<th>Border tax</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2030</td>
<td>0%</td>
<td>50%</td>
<td>150%</td>
</tr>
<tr>
<td>Acidification potential</td>
<td>th tonnes SO2eq</td>
<td>-0.79</td>
<td>-0.78</td>
<td>-0.79</td>
</tr>
<tr>
<td>Eutrophication potential</td>
<td>th tonnes PO4--- eq</td>
<td>3.23</td>
<td>3.24</td>
<td>3.25</td>
</tr>
<tr>
<td>Photochemical oxidation potential</td>
<td>th tonnes ethylene eq</td>
<td>-0.06</td>
<td>-0.03</td>
<td>-0.03</td>
</tr>
<tr>
<td>Human toxicity potential</td>
<td>th tonnes 1,4-DCBeq</td>
<td>4.98</td>
<td>5.01</td>
<td>5.03</td>
</tr>
</tbody>
</table>
### Global warming potential

<table>
<thead>
<tr>
<th>Global warming potential</th>
<th>CO2eq th tonnes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land use</td>
<td>km²</td>
</tr>
<tr>
<td>6.05</td>
<td>6.05</td>
</tr>
<tr>
<td>6.04</td>
<td>6.06</td>
</tr>
<tr>
<td>6.06</td>
<td>6.06</td>
</tr>
<tr>
<td>6.05</td>
<td>6.05</td>
</tr>
<tr>
<td>Food use</td>
<td>th tonnes</td>
</tr>
<tr>
<td>59.46</td>
<td>59.48</td>
</tr>
<tr>
<td>59.46</td>
<td>59.49</td>
</tr>
<tr>
<td>59.49</td>
<td>59.46</td>
</tr>
<tr>
<td>59.46</td>
<td>59.46</td>
</tr>
<tr>
<td>Feed use</td>
<td>th tonnes</td>
</tr>
<tr>
<td>-0.10</td>
<td>-0.12</td>
</tr>
<tr>
<td>-0.12</td>
<td>-0.13</td>
</tr>
<tr>
<td>-0.11</td>
<td>-0.12</td>
</tr>
<tr>
<td>-0.12</td>
<td>-0.12</td>
</tr>
<tr>
<td>Forestry use</td>
<td>th tonnes</td>
</tr>
<tr>
<td>58.94</td>
<td>58.97</td>
</tr>
<tr>
<td>58.95</td>
<td>58.97</td>
</tr>
<tr>
<td>58.97</td>
<td>58.96</td>
</tr>
<tr>
<td>58.94</td>
<td>58.94</td>
</tr>
<tr>
<td>Construction minerals</td>
<td>th tonnes</td>
</tr>
<tr>
<td>26.03</td>
<td>26.05</td>
</tr>
<tr>
<td>26.06</td>
<td>26.06</td>
</tr>
<tr>
<td>26.06</td>
<td>26.06</td>
</tr>
<tr>
<td>26.03</td>
<td>26.03</td>
</tr>
<tr>
<td>Industrial minerals</td>
<td>th tonnes</td>
</tr>
<tr>
<td>18.38</td>
<td>18.39</td>
</tr>
<tr>
<td>18.38</td>
<td>18.36</td>
</tr>
<tr>
<td>18.38</td>
<td>18.36</td>
</tr>
<tr>
<td>18.39</td>
<td>18.39</td>
</tr>
<tr>
<td>Ferrous ores</td>
<td>th tonnes</td>
</tr>
<tr>
<td>15.61</td>
<td>15.65</td>
</tr>
<tr>
<td>15.64</td>
<td>15.64</td>
</tr>
<tr>
<td>15.64</td>
<td>15.64</td>
</tr>
<tr>
<td>15.62</td>
<td>15.62</td>
</tr>
<tr>
<td>Non-ferrous ores</td>
<td>th tonnes</td>
</tr>
<tr>
<td>20.68</td>
<td>20.70</td>
</tr>
<tr>
<td>20.70</td>
<td>20.71</td>
</tr>
<tr>
<td>20.70</td>
<td>20.70</td>
</tr>
<tr>
<td>20.69</td>
<td>20.69</td>
</tr>
</tbody>
</table>

**4.6 Increase the re-use of plastics, and avoiding single use plastics**

Reusing plastics can offset the input of virgin material in the value chain. With reference to the waste hierarchy, re-use is viewed as preferable to recycling. In practice, the re-use of materials might have context-specific environmental strengths and disadvantages, for example in relation to the product weight-based emissions, the number of iterations of re-use, or the distance and mode of reverse logistics (WRAP, 2010).

Plastic products contain chemical additives, such as plasticisers and flame retardants, which are used to ensure diverse properties in the products (Geyer, Jambeck and Law, 2017). As many of these substances may be hazardous, and therefore harmful to the environment and to human health, some plastics applications and their reuse may result problematic. Plastic packaging, especially those in contact with food, represent an example where there may be a high risk of chemical exposure which can hamper reuse and recycling. More transparent information on substances of concern in products across the supply chain can promote non-toxic material cycles and consequently contribute to improving the safety and risk management of reuse, repair and recovery processes (European Commission, 2018).

Re-usable plastics are relevant in B2B and B2C scenarios, for instance in reusable pallets for distribution or re-usable carrier bags respectively. Increasing the re-use of products can also help to reduce the use of single-use plastics. Differential tax rates for production of items from recycled products (as compared to those based on virgin materials) should provide an incentive for more resource-efficient production processes.

**Re-usable pallets in distribution networks, France**

Using reusable packaging in B2B distribution can reduce packaging waste in the food value chain. One French retailer implemented 1.8 million reusable plastic packaging crates for distributing fresh fruit and vegetables, replacing single use packaging and wooden pallets. Reverse logistics on crates is supported with RFID technology, and the cases are rented rather than owned by the retailer. Total savings in operations are 150 tonnes of waste and 30% of emissions compared to previous practices (Leblanc, 2011).

Specific policies might facilitate re-use of plastics, e.g. deposit refund schemes, or taxes/charges on single use products. Re-usable plastics might also necessitate a change in business models, for example introducing service over ownership models for dispensers and re-usable cutlery in the catering sector, in order that the higher upfront costs of durable materials can be made affordable to businesses.
Appendix A: LCA methodology

A.1 Functional unit
In order to align the modelling of all products under consideration, the functional unit used throughout this work is one use of a product in question or of its alternative(s). For single-use plastic and non-plastic items, this equates to the production of 1 item. For multi-use items, this is the production of 1 item divided by its number of reuses, plus the burdens of 1 wash cycle.

A.2 Data sources & system boundaries
The life-cycle inventories compiled for all products under consideration are fully based on Ecoinvent v3.4 for both foreground and background data (Wernet et al., 2016). This was necessitated due to the breadth of the study but also in order to ensure comparability between all modelled systems. All stages from raw materials extraction up to and including use phase and end-of-life are considered.

For dealing with co-product allocation, the system expansion method is used via the Ecoinvent “consequential” model, as is generally recommended for studies with decision support in mind (Ekvall and Weidema, 2004). Via system expansion, the consequences of changes in demand for products from unconstrained suppliers (such that can respond to changes in demand, i.e. those that are expected to change) are modelled. Under system expansion, the products modelled receive the full burdens of the impacts of their inputs and emissions but also receive benefits (“credits”, i.e. impacts that are subtracted) for any by-products produced that can substitute other products (such as waste heat used for energy generation). For a fuller discussion on consequential modelling, refer to Ekvall and Weidema (2004) and Wernet et al. (2016).

Where possible, Ecoinvent market datasets have been used. Market datasets represent the consumption mixes of products in different regions, including also transport burdens, as well as additional product inputs in order to compensate for losses at the transportation stage (e.g. transmission losses for electricity). Thus, market datasets offer geographical representation, as well as a fuller view of the supply chains of product systems. Where market datasets have not been available, such have been constructed with generic Ecoinvent transport data used (Borken-Kleefeld and Weidema, 2013).

With regards to geography, geographic differentiation in products’ life cycles was out of scope for this work. Thus, market datasets with global geographies (GLO) have been used, except for the use phase of products, where data representative for Europe has been utilised (RER, Europe without Switzerland or other appropriate Ecoinvent geography). The use of globally-representative data is for avoiding the need for accounting for the geographical origins of the products used, which is increasingly difficult further up their life cycles (e.g. with respect to sourcing of primary materials such as fuels or metals). In contrast, the use phase is known to occur within Europe and thus representative datasets are used.

Ecoinvent datasets typically also include infrastructure burdens where appropriate. This has thus been included in the analysis where bundled in existing datasets, but no special effort has been made to add infrastructure burdens where such are missing. The same treatment is applied for secondary and tertiary packaging. For products such as drinks bottles, only the packaging itself is considered, i.e. filling of bottles etc. has not been considered.

The use phase - consisting of washing of items and wastewater treatment, as well as the life cycles of the aforementioned – is modelled for multi-use items only as the use phase of non-MU items was not deemed significant for inclusion.

Figure A.1 illustrates the system boundary of all products considered, as well as the life-cycle impact categories included.
A.3  Product systems studied

Twelve products & their potential alternatives were originally considered for modelling (see Table A.1). Their selection was aligned with the Strategy on Plastics Impact Assessment. The criteria for selection of plastics alternatives (SUNP and MU products) were that:

8. The materials of which SUNP items are composed avoid the generation of microplastics. This thus excluded biodegradable plastics from the study scope as such biodegradability can only be insured in specific conditions which are seldom met in the marine environment (Thompson, 2006; Kershaw, 2015).

9. Alternative products meet the same function as the plastic products that they substitute in terms of properties that the materials ensure. Such products were not identified for product groups Crisps packets and Sweet wrappers (transmission of O₂ and water vapour, opacity), as well as for SUNP Drinks cups and lids (permeability and resistance of insulating layer to heat) and sanitary towels (permeability and absorbency).

10. Multi-use items need to ensure that use of single-use plastics is avoided. This ruled out reusable cigarette filters, as such are used in addition of a traditional cigarette (as an additional filter) and would thus not displace the use of a cellulose acetate filter.

11. Alternatives need to satisfy broadly the same market. This ruled out items such as e-cigarettes, which are tobacco substitutes and thus not necessarily targeting an analogous market segment.

Table A.1 Product systems considered - with materials specified - for single-use plastic items (SUPs), single-use non-plastic items (SUNPs) and multi-use items (MU)
In choosing the reference products for each product category in Table 2, generally most widely used products have been selected. Where multiple such products exist (such as different volumes of drinks bottles), averaged products have been modelled, either in terms of mass (in the case of different sizes of the same product) or in terms of composition (in the case of alternatives from different materials existing for SUNP and MU items). Where possible, market reports have been used in order to derive average reference products.

### A.4 Washing and reusability of multi-use items

For modelling the washing of multi-use items, representative datasets were compiled from Ecoinvent data. Three *markets* for washing were compiled, representing an aggregate dataset consisting of inputs of water, energy, detergent and wastewater treatment (see Figure A.1). Due to the reusability of multi-use items, their burdens up to and including the manufacturing stage would be small and the product system would thus be dominated by its use (washing) phase. Washing impacts can strongly differ based on the technology used, especially with respect to EU Ecodesign criteria and uptake of newer appliances over time. Thus, we model a *best-case* and *worst-case* washing scenario, representing new and old technologies. The modelling of washing distinguishes between washing of sanitary and non-sanitary items. Table A.2 depicts the applicable washing market for different product groups. Burdens of infrastructure/building of machinery are taken as-is from Ecoinvent for input datasets but no additional effort has been made for full inclusion.

<table>
<thead>
<tr>
<th>Product category</th>
<th>Market for dishwashing</th>
<th>Market for laundry</th>
<th>Number of reuses</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cotton buds</td>
<td>PP bud</td>
<td>Paper bud</td>
<td>Reusable MDPE bud</td>
</tr>
<tr>
<td>Crisps packets</td>
<td>Excluded from scope</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sweet wrappers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sanitary towels</td>
<td>Ultrathin pad (PE, PP, PET, SAP)</td>
<td>-</td>
<td>Washable cotton pad</td>
</tr>
<tr>
<td>Wet wipes</td>
<td>Wet wipe (w/ lotion)</td>
<td>Cotton ball + lotion</td>
<td>Cotton handkerchief + lotion</td>
</tr>
<tr>
<td>Cutlery</td>
<td>Average PP utensil</td>
<td>Average wooden utensil</td>
<td>Average steel utensil</td>
</tr>
<tr>
<td>Straws</td>
<td>PP straw</td>
<td>Paper straw</td>
<td>Average reusable straw (steel/silicone)</td>
</tr>
<tr>
<td>Stirrers</td>
<td>PP stirrer</td>
<td>Wooden stirrer</td>
<td>Steel stirrer</td>
</tr>
<tr>
<td>Food containers</td>
<td>PS clamshell container</td>
<td>Paperboard + wax container</td>
<td>PE tupperware box</td>
</tr>
<tr>
<td>Drinks cups and lids</td>
<td>Paper cup w/ PE coating and LDPE lid</td>
<td>-</td>
<td>Reusable PP cup (w/ LDPE, rubber, silicone components)</td>
</tr>
</tbody>
</table>

Note(s): Products with materials separated by forward slash are market averages of separate products made of the materials given.
Cigarette butts - - - 
Drinks bottles X 2808 
Cotton buds X 734 
Sanitary towels X 426 
Wet wipes X 6330 
Cutlery X 4416 
Straws X 5412 
Stirrers X 11274 
Food containers X 515 
Drinks cups and lids X 564 

Notes(s): Dash depicts product groups where no multi-use items are modelled. Further given are the number of reuses assumed for multi-use products.

Figure A.2 further gives the number of assumed reuses for multi-use items, which are taken directly from the assumptions of the Strategy on Plastics Impact Assessment.

Figure A.2 Structure of washing models for sanitary and non-sanitary items, including shares of technologies used

A.5 End-of-life
End-of-life of products is modelled via EcoInvent datasets representative for Europe and with assumptions for end-of-life pathways shares taken from those used in the Strategy on Plastics Impact Assessment. Where no data was available, assumptions have been made, detailed in Table A.3. For SUNP drinks bottles, industry data has been applied (FEVE, 2015; van der Harst et al., 2016, citing EU Aluminium Association).

Table A.3 Assumptions on end-of-life shares used

<table>
<thead>
<tr>
<th>Product group</th>
<th>Product</th>
<th>Recycling</th>
<th>Incineration</th>
<th>Landfill</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cigarette butts</td>
<td>SUP</td>
<td>52%</td>
<td>48%</td>
<td></td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>SUNP</td>
<td>52%</td>
<td>48%</td>
<td></td>
<td>Same as SUP</td>
</tr>
<tr>
<td>Drinks bottles</td>
<td>SUP</td>
<td>65%</td>
<td>13%</td>
<td>21%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td>Item</td>
<td>SUNP</td>
<td>MU</td>
<td></td>
<td>FEVE, EU Aluminium Association</td>
<td></td>
</tr>
<tr>
<td>-----------------------</td>
<td>------</td>
<td>------</td>
<td>-----</td>
<td>---------------------------------</td>
<td></td>
</tr>
<tr>
<td></td>
<td>68%</td>
<td>68%</td>
<td>32%</td>
<td>32%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Same as SUNP</td>
<td></td>
</tr>
</tbody>
</table>

**Cotton buds**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0%</td>
<td>61%</td>
<td>38%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>0%</td>
<td>62%</td>
<td>37%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>50%</td>
<td>50%</td>
<td></td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Same as SUNP</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Same as MU</td>
</tr>
</tbody>
</table>

**Sanitary towels**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>62%</td>
<td>63%</td>
<td>38%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>63%</td>
<td>63%</td>
<td>37%</td>
<td>Strategy on Plastics IA</td>
</tr>
</tbody>
</table>

**Wet wipes**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>59%</td>
<td>63%</td>
<td>37%</td>
<td>Same as SUP</td>
</tr>
<tr>
<td></td>
<td>63%</td>
<td>63%</td>
<td>37%</td>
<td>Same as MU Sanitary towels</td>
</tr>
</tbody>
</table>

**Cutlery**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0%</td>
<td>60%</td>
<td>39%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>1%</td>
<td>61%</td>
<td>38%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>0%</td>
<td>64%</td>
<td>36%</td>
<td>Strategy on Plastics IA</td>
</tr>
</tbody>
</table>

**Straws**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>61%</td>
<td>61%</td>
<td>39%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>60%</td>
<td>60%</td>
<td>39%</td>
<td>Same as SUP</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Same as MU Cutlery</td>
</tr>
</tbody>
</table>

**Stirrers**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>5%</td>
<td>62%</td>
<td>38%</td>
<td>As SUP but no recycling</td>
</tr>
<tr>
<td></td>
<td>0%</td>
<td>64%</td>
<td>36%</td>
<td>Same as MU Cutlery</td>
</tr>
</tbody>
</table>

**Food containers**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>8%</td>
<td>65%</td>
<td>13%</td>
<td>Same as SUP bottles</td>
</tr>
<tr>
<td></td>
<td>10%</td>
<td>51%</td>
<td>39%</td>
<td>Strategy on Plastics IA</td>
</tr>
</tbody>
</table>

**Drinks cups and lids**

<table>
<thead>
<tr>
<th>Item</th>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0%</td>
<td>61%</td>
<td>39%</td>
<td>Strategy on Plastics IA</td>
</tr>
<tr>
<td></td>
<td>19%</td>
<td>56%</td>
<td>25%</td>
<td>Strategy on Plastics IA</td>
</tr>
</tbody>
</table>

### A.6 Life-cycle impact assessment

The major strength of life-cycle assessment is that it allows for comparison of different environmental aspects for the same product. Figure 4 states the impacts considered in this study, namely:

- Climate change, expressed as kilograms of CO2 equivalent for a 100-year time-horizon (GWP100)
- Water use – volume of water in m³
- Water pollution impacts on ecosystems due to eutrophication in kilograms of Phosphorus-input equivalent
- Water pollution impacts on human health – exposure to pollution in Chemical Toxicity Units (standardised units of toxicity of different substances, the same principle as e.g. CO2-equivalence)
- Air pollution as kilograms of PM2.5-equivalent emissions
- Land use, in terms of kilograms of soil organic carbon displaced, serving as a proxy for land-use intensity
- Resource use (fossil fuels, minerals, renewable materials) in kilograms of Sb-equivalent.
All impacts used follow the ILCD 1.0.8 2016 recommended methodology, except for water use, for which a simple summation of water inputs from products’ life cycle inventories is used due to lack of an appropriate indicator.

Thus, the chosen indicators allow us to assess both the environmental impacts of the products under scrutiny, as well as their burdens in terms of resource use.

A.7 Specification of washing datasets

The following section details the modelling of washing of multi-use items and the construction of its life-cycle inventories:

Market for laundry

Washing of sanitary items is modelled as carried out via a household washing machine and dried via a weighted average process representative for shares of drying behaviours in Europe (Schmitz and Stammingner, 2014). Best and worst-case washing machines and tumble dryers are modelled based on Boyano et al. (2017a) and is representative of EU Ecodesign and Energy Label criteria for household washing machines and washer-dryers. Best and worst-case options are based on the average lifetime of the appliances, 10 and 8 years respectively (ibid.). Energy use for drying in a heated room is based on the residual moisture content on cotton (62%) and calculated via the latent heat of vaporisation of water of 2257 kJ/kg (Schmitz and Stammingner, 2014). Detergent use per cycle is based on a common-sense assumption of 0.035 kg/cycle as for a typical household washing machine. All water used is treated.

Table A.4 details the assumptions behind the washing model, represented by the market for laundry dataset.

### Table A.4 Assumptions behind the market for laundry process

<table>
<thead>
<tr>
<th></th>
<th>Best-case</th>
<th>Worst-case</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Machine wash</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/kg)</td>
<td>0.11</td>
<td>0.19</td>
</tr>
<tr>
<td>Water use (l/kg)</td>
<td>6.29</td>
<td>9.95</td>
</tr>
<tr>
<td>Capacity (kg/cycle)</td>
<td>7.22</td>
<td>5.16</td>
</tr>
<tr>
<td>Technology (year)</td>
<td>2014</td>
<td>2004</td>
</tr>
<tr>
<td><strong>Tumble dry</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/kg)</td>
<td>0.58</td>
<td>0.69</td>
</tr>
<tr>
<td>Technology (year)</td>
<td>2013</td>
<td>2006</td>
</tr>
<tr>
<td><strong>Air dry, heated room</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/kg)</td>
<td>0.39</td>
<td>0.39</td>
</tr>
<tr>
<td><strong>Air dry, unheated room or outside</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/kg)</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td><strong>Detergent</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Use (kg/kg)</td>
<td>0.005</td>
<td>0.007</td>
</tr>
</tbody>
</table>

Note(s): Appliance data is sourced via the European Committee of Domestic Equipment Manufacturers (CECED) in Boyano et al. (2017a).

Market for dishwashing

Washing of non-sanitary items is modelled as a mix between dishwasher use and handwashing, assuming a dishwasher penetration of 60% (Boyano et al., 2017b). The modelling of the dishwasher appliance is representative of EU Ecodesign and Energy label criteria for household dishwashers and assumes 12 items per place setting, 140 items per cycle and an average appliance age of 12 years (ibid.). Handwashing and detergent use best and worst cases are taken from Stammingner et al. (2007). For the former, these are values for Germany and Spain/Portugal respectively, while for the latter these are for Germany and Italy.

Table 10 details the non-sanitary item washing model, represented by the market for dishwashing dataset.
Table A.5 Assumptions behind the market for dishwashing process

<table>
<thead>
<tr>
<th></th>
<th>Best-case</th>
<th>Worst-case</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Machine wash</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/item)</td>
<td>0.006</td>
<td>0.008</td>
</tr>
<tr>
<td>Water use (l/item)</td>
<td>0.070</td>
<td>0.115</td>
</tr>
<tr>
<td>Technology (year)</td>
<td>2014</td>
<td>2002</td>
</tr>
<tr>
<td><strong>Handwash</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Energy use (kWh/item)</td>
<td>0.009</td>
<td>0.030</td>
</tr>
<tr>
<td>Water use</td>
<td>0.319</td>
<td>1.181</td>
</tr>
<tr>
<td>Detergent Use (kg/item)</td>
<td>0.0002</td>
<td>0.0005</td>
</tr>
</tbody>
</table>

Note(s): Appliance data is sourced from the European Committee of Domestic Equipment Manufacturers (CECED) in Boyano et al. (2017b).

A.8 Specification of reference products:

The following appendix details the specification of reference products for all product groups considered for life-cycle inventory compilation:

**Cigarette butts**

SUP cigarette butts are modelled as a typical cellulose acetate filter, the mass of which taken as per O’Connor et al. (2008) and with composition following Bin et al. (2017). As Ecoinvent 3.4 does not provide a cellulose acetate tow dataset, this is modelled following its chemical reaction with stoichiometry as per Campbell et al. (1973) and with magnitudes of energy inputs from Ecoinvent 3.4 dataset viscose production, GLO (Althaus et al., 2017) but with European market datasets used. Process electricity is taken from dataset market for spinning, bast fibre, GLO (Ecoinvent, 2017a) and transport burdens to end-users are added from category group 1200 Tobacco products from Borken-Kleefeld and Weidema (2013).

SUNP cigarette butts are modelled in the same way as SUP cigarette butts but with typical composition of filter tow taken from Lisauskas, Van Osten and Greenbutts Llc (2012). In order to ensure that both modelled filters achieve the same filterability, the mass of alternative materials used has been adjusted based on the difference in densities between cellulose acetate and the cotton/hemp mix serving as alternative.

The full composition of reference products for the Cigarette butt product group is given in Table A.6.

Table A.6 Composition of reference products for modelling the Cigarette butt product group

<table>
<thead>
<tr>
<th></th>
<th>SUP</th>
<th></th>
<th>SUNP</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
<td>Weight</td>
<td></td>
</tr>
<tr>
<td>Cellulose acetate filter</td>
<td>0.12</td>
<td>Natural fibre filter</td>
<td>0.13</td>
<td></td>
</tr>
<tr>
<td>Acetate tow</td>
<td>0.10</td>
<td>Natural fibre tow</td>
<td>0.11</td>
<td></td>
</tr>
<tr>
<td>Plug wrap paper</td>
<td>0.01</td>
<td>Hemp</td>
<td>0.03</td>
<td></td>
</tr>
<tr>
<td>Tipping paper</td>
<td>0.01</td>
<td>Cotton</td>
<td>0.06</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Plug wrap paper</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tipping paper</td>
<td>0.01</td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.
**Drinks bottles, caps and lids**

Drinks packaging is a diverse market in terms of volume capacity of packages, as well as in terms of materials used. Market report data from GlobalData (2017) is used for determining averaged reference products.

SUP drinks bottles are taken to be made of PET and modelled in volume as the weighted average of what is most common on the European market (80+% market share), with weights taken from industry data. Bottle-grade PET data is taken directly from Ecoinvent and bottle manufacturing is assumed to be comprised of injection and stretch-blow moulding. Transport data to end-users for product group 2220 Plastics products is used (Kleefeld and Weidema, 2013).

The reference SUNP drinks container is modelled as an average mix between an aluminium can and a glass bottle, with market shares from GlobalData used assuming full substitution of plastics by aluminium/glass. Transport data is for product group 2310 Glass and glass products only due to absence of a suitable category for aluminium packaging (a conservative assumption given larger burdens due to higher weight of glass).

The reference MU item is modelled representing a market-averaged refillable flask of PET/aluminium. Market shares are determined assuming a 50:50 split between materials. Transport data is for 2220 Plastics products. MU bottles receive washing burdens from the market for dishwashing dataset. The weight and composition of an aluminium flask is taken from Simon et al. (2016).

Table A.7 details the composition of reference products for the Drinks bottles, caps and lids product group.

*Table A.7 Composition of reference products for modelling the Drinks bottles product group*

<table>
<thead>
<tr>
<th></th>
<th>SUP</th>
<th>SUNP</th>
<th>MU, consumer-led</th>
<th>MU, industry-led</th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
</tr>
<tr>
<td>PP bottle (w/ cap)</td>
<td>36</td>
<td>Container, average of:</td>
<td></td>
<td>Container, average of:</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Glass bottle (72%)</td>
<td></td>
<td>PET flask (w/ cap) (50%)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Aluminium can (17%)</td>
<td></td>
<td>Aluminium flask (50%)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Flask</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>PET cap</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Cotton buds**

SUP and SUNP cotton buds are modelled as having the same weights but with PP and paper sticks respectively. The MU reference product is a washable MDPE bud. All weights have been estimated. The MU item receives washing burdens from the market for dishwashing dataset. Transport to end-user data for product group 2023 Soap and detergents, polishes, perfumes, toilet preparations is used.

The SUP product is modelled as the extrusion of a plastic pipe via the market for earth tube heat exchanger, polyethylene, DN 200, GLO (Ecoinvent, 2017b) dataset but with polypropylene substituting PE. Process burdens are included via the market for spinning, bast fibre, GLO (Ecoinvent, 2017a) dataset, the assumption being that the process burdens for working cotton are similar to those of modelled textile products. The modelling of SUNP buds is identical save for the extrusion process, where kraft paper production is used instead (50:50 mix of bleached vs non-bleached paper assumed). The MU product is assumed to be injection moulded, with a 50:50 mix of HDPE and LDPE used to represent MDPE.
Table A.8 represents the composition of the Cotton bud reference products.

Table A.8 Composition of reference products for modelling the Cotton bud product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>Material</th>
<th>Weight</th>
<th>SUNP</th>
<th>Material</th>
<th>Weight</th>
<th>MU</th>
<th>Material</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Cotton bud</td>
<td>0.23</td>
<td></td>
<td>Paper bud</td>
<td>0.23</td>
<td></td>
<td>MDPE washable bud</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>PP stick</td>
<td>0.17</td>
<td></td>
<td>Paper stick</td>
<td>0.17</td>
<td></td>
<td>Cotton</td>
<td>0.06</td>
</tr>
<tr>
<td></td>
<td>Cotton</td>
<td>0.06</td>
<td></td>
<td>Cotton</td>
<td>0.06</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.

Sanitary towels

The SUP reference product is taken to be an ultrathin sanitary pad with composition taken from the EDANA nonwovens association Sustainability Report for 2008. This is stated to be the most common pad on the market and is also the lightest of all variants listed therein. The superabsorbent polymer (SAP) of the ultrathin pad is modelled as polyacrylamide. Transport to end-user data for product group 2023 Soap and detergents, polishes, perfumes, toilet preparations is used.

The MU reference product is a washable cotton towel. Transport is modelled as for product group 1300 Textiles, weight is estimated. The MU product receives washing burdens from the market for laundry dataset.

Table A.9 details the composition of the Sanitary pads reference products.

Table A.9 Composition of reference products for modelling the Sanitary towels product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>Material</th>
<th>Weight</th>
<th>MU</th>
<th>Material</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Ultrathin pad</td>
<td>6.20</td>
<td></td>
<td>Washable cotton pad</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td>Paper</td>
<td>0.21</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Adhesive</td>
<td>0.43</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Superabsorbent polymer</td>
<td>0.35</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Pulp</td>
<td>2.99</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>PE</td>
<td>0.997</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>PP</td>
<td>0.997</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>PET</td>
<td>0.997</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.

Wet Wipes

SUP wet wipes are modelled via a market average mix of materials in use in Europe based on market report data from Smithers Pira (2016). The weight and lotion content of the reference wipe are taken from industry data, with the composition of the lotion itself from Faught et al. (2014). Transport to end-user data for product group 2023 Soap and detergents, polishes, perfumes, toilet preparations is used. Process burdens are modelled via the market for spinning, bast fibre, GLO dataset as for all other textile products.

SUNP wipes are modelled as cotton balls with the same proportion of lotion by mass assumed as for SUP wipes. Weights have been estimated. All other modelling is analogous to SUP wipes.
The MU reference product is a washable cotton handkerchief. The kerchief is modelled analogously to the above but with double the lotion usage as a conservative assumption due to more wasteful application by end-users. The MU product receives washing burdens from the market for laundry dataset.

Table A.10 Composition of reference products for modelling the Wet wipes product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>Weight</th>
<th>SUNP</th>
<th>Weight</th>
<th>MU</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Synthetic wipe</td>
<td>3.80</td>
<td>Cotton ball + lotion</td>
<td>4.30</td>
<td>Cotton handkerchief</td>
<td>12</td>
</tr>
<tr>
<td>Fibre</td>
<td>1.10</td>
<td>Cotton ball</td>
<td>2.50</td>
<td>Cotton</td>
<td>6.70</td>
</tr>
<tr>
<td>Viscose fibre</td>
<td>0.47</td>
<td>Lotion</td>
<td>1.80</td>
<td>Lotion</td>
<td>5.40</td>
</tr>
<tr>
<td>PET fibre</td>
<td>0.53</td>
<td>Water</td>
<td>1.18</td>
<td>Water</td>
<td>3.59</td>
</tr>
<tr>
<td>PP fibre</td>
<td>0.10</td>
<td>Glycerine</td>
<td>0.32</td>
<td>Glycerine</td>
<td>0.97</td>
</tr>
<tr>
<td>Lotion</td>
<td>2.70</td>
<td>Colloidal oatmeal</td>
<td>0.04</td>
<td>Colloidal oatmeal</td>
<td>0.11</td>
</tr>
<tr>
<td>Water</td>
<td>1.80</td>
<td>Benzyl alcohol</td>
<td>0.01</td>
<td>Benzyl alcohol</td>
<td>0.03</td>
</tr>
<tr>
<td>Glycerine</td>
<td>0.49</td>
<td>Sodium Chloride</td>
<td>0.00</td>
<td>Sodium Chloride</td>
<td>0.00</td>
</tr>
<tr>
<td>Colloidal oatmeal</td>
<td>0.05</td>
<td>Cetyl alcohol</td>
<td>0.05</td>
<td>Cetyl alcohol</td>
<td>0.16</td>
</tr>
<tr>
<td>Benzyl alcohol</td>
<td>0.01</td>
<td>Petrolatum</td>
<td>0.01</td>
<td>Petrolatum</td>
<td>0.03</td>
</tr>
<tr>
<td>Sodium Chloride</td>
<td>0.00</td>
<td>Isopropyl Palmitate</td>
<td>0.05</td>
<td>Isopropyl Palmitate</td>
<td>0.17</td>
</tr>
<tr>
<td>Cetyl alcohol</td>
<td>0.08</td>
<td>Distearyldimonium Chloride</td>
<td>0.09</td>
<td>Distearyldimonium Chloride</td>
<td>0.27</td>
</tr>
<tr>
<td>Petrolatum</td>
<td>0.02</td>
<td>Dimethicone</td>
<td>0.02</td>
<td>Dimethicone</td>
<td>0.07</td>
</tr>
<tr>
<td>Isopropyl Palmitate</td>
<td>0.08</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distearyldimonium Chloride</td>
<td>0.14</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dimethicone</td>
<td>0.04</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.

Cutlery

The SUP reference cutlery item is an average polypropylene utensil with mass taken from Öko-Institut eV (2017). It is assumed that injection molding represents process burdens. Transport to end-users is modelled via the 2200 Plastics products product group.

The SUNP reference product is a wooden utensil and is assumed to have the same weight as for SUP. It is modelled via the market for plywood, for indoor use, RER dataset (Ecoinvent, 2017c; due to lack of a global dataset). This is the lowest-density wood product available in Ecoinvent and is assumed to represent the typical low-grade wood that would be used for manufacturing of wooden utensils. Transport to end-use is via the 1629 Other wood products product group.

MU cutlery is an average steel utensil from Öko-Institut eV (2017). Material inputs are via the market for steel, chromium steel 18/8, hot rolled, GLO dataset (Ecoinvent, 2017d) and a 4-stroke impact extrusion process is assumed (expert consultation). The MU reference product receives washing burdens from the market for dishwashing dataset. Transport is via the 2500 Articles of base metal product group.

Table A.11 gives the Cutlery reference products modelled.
### Table A.11 Composition of reference products for modelling the Cutlery product group

<table>
<thead>
<tr>
<th>Material</th>
<th>Weight</th>
<th>Material</th>
<th>Weight</th>
<th>Material</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>PP utensil</td>
<td>2.60</td>
<td>Wooden utensil</td>
<td>3</td>
<td>Steel utensil</td>
<td>31</td>
</tr>
</tbody>
</table>

Note(s): All weights given in grams.

### Straw

SUP straws are modelled as a polypropylene extrusion process analogous to that for SUP cotton buds via the *market for earth tube heat exchanger, polyethylene, DN 200, GLO* dataset (Ecoinvent, 2017b). Transport burdens are also analogous, product weight is based on industry data.

The SUNP straw is modelled as made of kraft paper (50:50 bleached/unbleached paper assumed) analogous to SUNP cotton buds. Weight is assumed same as for the SUP reference product.

MU straws are modelled as a 50:50 market average between silicone and steel straws. The silicone item is modelled via the generic Ecoinvent silicone product dataset, while the steel straw model is analogous to that for MU cutlery.

Table A.12 gives the modelled Straws reference products.

### Table A.12 Composition of reference products for modelling the Straws product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
</tr>
<tr>
<td>PP straw</td>
<td>0.40</td>
<td>Paper straw</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Steel straw (50%)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Silicone straw (50%)</td>
</tr>
</tbody>
</table>

Note(s): All weights given in grams.

### Stirrers

SUP stirrers are modelled analogous to SUP straws and cotton buds. Assumed weight is an industry average from multiple sources.

SUNP stirrers are assumed made of wood and modelled analogous to SUNP cutlery. Weight is estimated.

MU stirrers are assumed to be analogous to MU cutlery (i.e. a steel spoon). Table A.13 presents the Stirrers reference products.

### Table A.13 Composition of reference products for modelling the Stirrers product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
</tr>
<tr>
<td>PP stirrer</td>
<td>0.60</td>
<td>Wooden stirrer</td>
</tr>
</tbody>
</table>

Note(s): All weights given in grams.

### Food containers

An average polystyrene clamshell container is modelled as the SUP reference item, sourced from Frankin Associates (2006). Transport burdens to end-user are via the *2200 Plastics products* product group.

The above reference is also used for the SUNP reference item – a wax-lined paperboard container. Transport burdens are via the *1702 Corrugated board and containers* product group.
The MU reference product is a reusable polyethylene tupperware container, its weight estimated (lid inclusive). Transport is analogous to the SUP food container; the MU product receiving washing burdens from the *market for dishwashing* dataset.

Table A.14 details the composition of Food containers reference products.

### Table A.14 Composition of reference products for modelling the Food containers product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>SUNP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
<td>Material</td>
</tr>
<tr>
<td>PS clamshell</td>
<td>5</td>
<td>Paperboard + wax box</td>
</tr>
<tr>
<td>Paperboard</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wax</td>
<td>1</td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.

**Drinks cups and lids**

The SUP reference drinks cup is a composite corrugated board cup with polyethylene lining and an LDPE lid. Total product weight is estimated, with shares of individual components from Vercalsteren *et al.* (2006). Transport burdens are for product group 2200 *Plastics products*. The product is assumed to be injection moulded.

The MU drinks cup is modelled via an LCA analysis conducted for KeepCup (Edge Environment, 2017). Transport burdens are analogous to the SUP product, washing burdens are received via the *market for dishwashing* dataset.

Table A.15 presents the Drinks cups and lids reference products’ compositions.

### Table A.15 Composition of reference products for modelling the Drinks cups and lids product group

<table>
<thead>
<tr>
<th>SUP</th>
<th>MU</th>
</tr>
</thead>
<tbody>
<tr>
<td>Material</td>
<td>Weight</td>
</tr>
<tr>
<td>Paper cup w/ PE coating</td>
<td>11</td>
</tr>
<tr>
<td>Corrugated board</td>
<td>10.34</td>
</tr>
<tr>
<td>PE</td>
<td>0.66</td>
</tr>
<tr>
<td>LDPE lid</td>
<td>3.00</td>
</tr>
<tr>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note(s): Indents represent modelled sub-compositions of individual components. All weights given in grams.
Appendix B: E3ME macroeconomic methodology

B.1 Specification of scenarios

Scenario One: Tax on non-recyclable plastic waste
Scenario One models the imposition of a tax at the point of generation of non-recyclable plastic waste generation\(^{113}\). The tax imposes an additional cost to each sector, dependent upon its magnitude of waste generated. Rate of taxation is 0.5€ per kg of plastic waste. The key responses in E3ME are:

- increased prices resulting from increased costs of production
- a reduction of waste generation (crucially through substitution of virgin plastic to alternative materials)

Scenario Two: Tax on production of the plastics sector
Scenario Two assesses the imposition of a tax at the point of production: a tax is applied proportional to the weight of plastic material produced. The tax imposes an additional cost to the plastics sector.\(^{114}\) Rate of taxation is 0.102€ per kg of plastic.\(^ {115}\) The key responses in E3ME are:

- increased price in the plastics sector resulting from increased cost of production
- increased prices across the economy from increased costs of production in industries using plastic as an input to production
- reduction in final consumer demand for plastic products
- reduction of intermediate demand for plastic

Given the level of sectoral disaggregation in E3ME, it is not possible to examine in detail the effects of heterogenous taxation across different plastics, or the taxation of additives to plastics. An important issue to consider in Scenario Two is potential production leakage.

Scenario Three: Border tax on select plastic external imports
Scenario Three considers the imposition of a tax at the point of import into the European Union. The border tax is applied to sheets of plastic\(^ {116}\) only, at a rate of 0.102€ per kg. The key responses in E3ME are:

- increased prices of imports (both final consumer and intermediate goods)
- increased prices across the economy from increased costs of production in sectors that use plastic as an input to production
- a reduction in final consumer demand for plastic products
- a reduction in intermediate demands for plastic

B.2 How the demand for plastics is reduced in response to higher prices

The users of plastics can be grouped into two broad categories: other firms that use plastics in their production process and households that consume plastics directly as final products. Firms purchasing

\(^{113}\) Collection of waste and materials recovery are both categorised under NACE 38 ‘Waste collection, treatment and disposal activities; materials recovery’. Wholesale of waste and scrap is found under NACE 47.6 ‘Other specialised wholesale’.

\(^{114}\) Given data limitations, estimates of non-recyclable plastic waste generation used Eurostat plastic waste generation data.

\(^{115}\) The tax is characterised as a production tax applied to the whole output of the plastics sector. If the production tax was charged for weight of plastic produced through polymerisation, the tax would be applied to NACE Sector 20 ‘Manufacture of chemicals and chemical products’.

\(^{116}\) Tax rate taken from suggestions on plastic packaging waste tax in EC (2016) ‘Study on Assessing the Environmental Fiscal Reform Potential for the EU28’.

\(^{116}\) Tax applied to a broad definition of plastic sheets: PRODCOM code range 22213010 to 22214280 and PRODCOM code 22292950.
plastics are referred to as intermediate demands, while consumer purchases are referred to as final demands.

Intermediate demand in E3ME is determined by the input-output relationships in the model. These input-output relationships are fixed and do not respond to changes in prices; therefore we had to manually adjust changes in intermediate demands in the model. The elasticities used to determine how much demand should change in response to higher prices are described below.

A tax on plastics reduces final demand across sectors if higher costs are passed on to users of plastics in the form of higher prices. This mechanism, however, does not capture non-price related changes, such as new technologies that allow firms to use plastic more efficiently or to use alternative materials. Given the importance of this effect, sensitivity analysis of this behavioural response is employed. The key parameters for the sensitivities are the relative cost of using alternative materials and technological constraints.

The model specification captures price substitution effects in consumers’ demand across expenditure categories, so there is no requirement for modelling adjustments to final consumption demand. A tax on plastics affects direct consumption of plastic and indirect consumption through goods that contain plastic or use plastic packaging.

**B.3 Assumptions**

The following substitution elasticities for plastic were used to adjust intermediate demand:

- Low: 0% (no change in the production function in response to higher plastics prices)
- Middle: 50%
- High: 150%

The assumed elasticity is key to characterising the economy’s response to the taxes, given that a large proportion of the output of the plastics sector is intermediate goods. The output of the plastics sector is very diverse and the substitutability of plastics for other materials can vary significantly. A more detailed study is required to properly assess the magnitude of production response to a tax on plastics. Issues affecting the size of the elasticity include:

- technological substitutability of plastic\(^{117}\)
- the cost of alternative materials
- non-price behavioural responses: e.g. the tax signal to encourage more sustainable production may be more important than the cost increase itself

The related assumption is how the production function changes given the assumed elasticity above. This assumption characterises how production inputs and technologies adapt. In these stylised scenarios, a one-to-one redirection of demand to the domestic waste management sector is applied. This approach characterises a circular economy response, where firms use recycled material/feedstock, in the same volumes and at the same (pre-tax) price as virgin plastic. Alternative responses in the economy include:

- a more than proportional reallocation to waste management; implying that a higher value of recycled material is required compared to using virgin material
- a reallocation to glass, metal, or wood products; a change in material use in production
- a reduction in total material demanded in production; higher input costs incentivise improvements in efficiency of material use

The above responses are not mutually exclusive to each other or to the current assumption. It is likely that responses in the economy would be a combination of increased use of recycled material, material substitution, and material efficiency improvements.

\(^{117}\) Denkstatt (2010) find that 16% of the plastics market is not-substitutable ‘without a significant change in design, function, service rendered or in the process itself, which delivers a certain service’.
Given the level of sectoral disaggregation in E3ME\textsuperscript{118}, substitution between different types of plastics has not been explicitly assessed. If the tax targets certain plastics, then substitution between plastics is an important consideration in the design of the tax.

Revenues from the plastics tax are directed towards government expenditure so that the scenarios remain ‘revenue neutral’ overall.

\textsuperscript{118} Plastics is part of NACE Rev.2 sector 22 ‘Manufacture of rubber and plastic products’.
Appendix C References


Plastics recyclers Europe (2016) 20 years later and the way forward- making more from plastics waste, Brussels: Plastics recyclers Europe.


Links between production, the environment and environmental policy

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1 Introduction

Motor vehicle manufacture is an economically important and politically significant sector. Links between the sector and the environment are many and significant: production is important in the scale of raw material and energy use; product design is important from the perspective of the circular economy with regards to end-of-life vehicle (ELV) waste; and product use is crucial because of localised noise & air pollution and emissions of greenhouse gases (GHGs).

The motor vehicles sector is one of Europe’s most important manufacturing sectors but also a major contributor to a range of environmental pressures. The life-cycle of motor vehicles involves significant use of resources and energy, as well as the creation of waste and pollutants at later stages in the supply chain.

A transformation of the motor vehicles sector is a necessary component of achieving the SDGs and EU-specific goals. The policy landscape surrounding the sector is complex given the political, socio-economic, and environmental importance of motor vehicles, especially in the Union. The potential power of the consumer to shape the sector provides a wealth of policy opportunities in addition to policy aimed at manufacturers.

A number of key technical terms and acronyms are used in this report to refer to specific technologies within a motor vehicle. These are explained in a short glossary below;

<table>
<thead>
<tr>
<th>Technical term</th>
<th>Meaning</th>
</tr>
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<tbody>
<tr>
<td>Powertrain</td>
<td>The drive unit of the vehicle, which provides the power for motion</td>
</tr>
<tr>
<td>ICE</td>
<td>Internal combustion engine – a vehicle powered through the burning of (typically fossil) fuel</td>
</tr>
<tr>
<td>BEV</td>
<td>Battery electric vehicle – a vehicle with a battery powering an electric motor, and no alternative source of power</td>
</tr>
<tr>
<td>PHEV</td>
<td>Plug-in hybrid vehicle – a vehicle with a battery, electric motor and a combustion engine, which can switch between the two to source power</td>
</tr>
<tr>
<td>HEV</td>
<td>Hybrid electric vehicle – a vehicle which is primarily powered via a conventional internal combustion engine, but which includes a small battery and electric motor, where the battery is charged through the recouping of energy that would otherwise be wasted (e.g. during braking)</td>
</tr>
<tr>
<td>FCEV</td>
<td>Fuel cell electric vehicle – a vehicle with an electric motor which receives power from a fuel cell, via an electrochemical reaction, typically from hydrogen and oxygen.</td>
</tr>
<tr>
<td>EV</td>
<td>Electric vehicle – a broad term, typically including BEV, PHEV and FCEV</td>
</tr>
<tr>
<td>PM10, PM2.5</td>
<td>Different forms of particulate matter, which are created from the burning of fossil fuels (and tyre and brake wear)</td>
</tr>
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2 Environmental footprint of the sector

2.1 Research questions
- How do these sectors benefit from the environment?
- How do they affect the environment?
- What is their resource use?
- The evolution in consumers and investors demand for increased transparency on environmental performance?

2.2 Overview
The motor vehicles sector is one of Europe’s most important manufacturing sectors but also a major contributor to a range of environmental pressures. The life-cycle of motor vehicles involves significant use of resources and energy, as well as the creation of waste and pollutants at later stages in the supply chain. Environmental impacts from the sector in the use phase of vehicles are considerable. Environmental pressures linked to motor vehicles include:

- Production emissions (e.g. car plants and steel production)
- Production resource use (extraction of critical raw materials – e.g. palladium for catalytic converters)
- Use phase emission (e.g. CO2, NO2 and fluorinated greenhouse gases – from mobile air conditioning, tire wear and tear leading to micro plastics emissions)
- Use phase resource use (e.g. fossil fuel use)
- Use phase land use conversion (e.g. urban, highway infrastructural demands & land conversion with biodiversity impacts)
- Use phase noise pollution (e.g. urban noise pollution with health impacts)
- Use phase environmental health impacts (e.g. non-communicable disease, physical inactivity)
- End of life pollution (e.g. ELV management or tire dumping)

The manufacture of internal combustion motor vehicle sector relies on the environment to source raw materials needed for manufacturing, as well as fossil fuels needed for energy. Raw materials needed for the manufacturing of cars include steel, iron and aluminium, and increasingly polymers (Tagliaferri et al. 2016, Schulz 2016). Terrestrial acidification is caused by the production of platinum-group metals during the production phase and the emission of sulphur dioxide during the use phase of the car (Hawkins et al. 2013). In general, extractive industries for raw materials and the production processes themselves are energy intensive.

Notable in the sector are the impacts of motor vehicles in their use phase; these are linked to fossil fuel use, GHG emissions and air pollution. One-fifth of EU’s total CO2 emissions come from road transport.120 Particulate matter (PM) is emitted during the production phase by metal supply chains and during the use phase by fuel combustion, brakes and tyres wear and road dust (Hawkins, et al. 2013, EEA 2017a, Rogge et al. 1993). Air pollutants including particulate matter (PM), sulphur oxides, nitrogen oxides, methane and ozone constitute the largest environmental health risk in Europe and lead to premature deaths and an increased incidence of various diseases (e.g. cancer and cardiovascular diseases) (Lim et al 2012, WHO 2015). Emissions of PM2.5s alone were responsible for around 399,000 premature deaths in 2014 (EEA, 2017c). During the life cycle of motor vehicles, toxic chemicals are released that can have harmful impacts on humans (AD Little 2016). Terrestrial eco-toxicity is mainly caused during the use stage by zinc emissions from tire wear and copper and titanium emissions from brake wear. Eutrophication of freshwater is caused by the disposal of sulfidic mine tailings (Hawkins et al 2013).

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120 COM(2017) 283 final
Through the introduction of EURO emissions standards over the past decade, there have been substantial reductions in air pollutants from the tailpipe of motor vehicles. According to the EEA\textsuperscript{121}, nitrogen oxides (NO\textsubscript{x}) from tailpipe emissions has fallen by 60\% over 1990-2015. Similarly, particulates (PM10s) from tailpipe have seen a 60\% reduction between 2000 to 2015. However, as mentioned above particulates from brake and tyre wear have remained largely constant over the past decade.

2.3 The role of fuel-efficient technologies

There remains much more that can be done to improve the efficiency of the internal combustion engine powered vehicle, and many of the technologies that are already available on the marketplace will become more prevalent in the coming years as manufacturers aim to meet the 2021 emissions targets.

Start-stop technology using advanced lead-based batteries is perhaps the most cost-effective way of achieving reductions of 5-10 per cent in CO\textsubscript{2} emissions (depending on whether the system can recapture braking energy). Ricardo-AEA has estimated that the cost per gram of CO\textsubscript{2} reduction is about half that of improving the fuel efficiency of the internal combustion engine, and less than a quarter of that for hybridisation.

Other options that are likely to be applied first include engine downsizing coupled with boost (e.g. a combination of turbo- and super-charging) and direct injection for petrol engines. For example, there has already been a 31\% reduction in g/km of CO\textsubscript{2} between 2010 petrol Ford Focus variants (at 159 g/km) and 2012 EcoBoost branded variants (at 109 g/km), achieved mainly using downsized engines (from 1.6 litres to 1.0 litres) with turbo-charging, direct injection and start-stop technologies.

Systems combined with increasing levels of hybridisation offer even greater potential benefits – e.g. a 52\% reduction in CO\textsubscript{2} emissions from the use of the 2010 petrol Toyota Yaris (at 164 g/km) to the 2012 Toyota Yaris hybrid (at 79 g/km). In the past, the high cost and time taken to produce and use carbon fibre has limited it to niche/small-scale and high-end applications in vehicles. However, recent research has made significant strides in both areas.

Alongside improvement to the ICE powertrain (i.e. the unit within the vehicle that provides motion) itself, there are several improvements to the rest of the vehicle that will improve overall vehicle efficiency. These include options such as aerodynamic benefits, lightweighting and low rolling resistance tyres.

2.4 Impacts of improved vehicle testing

Current standards for fuel efficiency of light duty vehicles (passenger cars and vans) have been assessed under the New European Drive Cycle (NEDC), however in recent years, the NEDC test cycle has come under scrutiny for failing to reflect the performance of vehicles under real world conditions. Through surveying a large sample of drivers about vehicle performance, the ICCT (2017b) has been able to estimate that the gap between the test cycle emissions and real-world emissions has been steadily increasing over the past decade, with a gap of around 9\% in 2001 expanding to 42\% in 2016.

In an aim to improve test procedures, the EU introduced the World Harmonised Light Vehicle Test Procedure (European Commission, 2017b). Since September 2017, new car models must pass the new type approval procedures, in a lab, to drive on European roads. The expectation is this improved testing procedure will close the gap between tested emissions and actual real-world emission. This should mean that any future technologies that are introduced reduce real world emissions, rather than simply reducing emissions reported in the testing cycle, and so ensure a real improvement in the carbon footprint of vehicles.

2.5 The electrification of vehicles

The electrification of motor vehicles presents an opportunity to reduce the environmental impacts of motorised transport. The global stock of electric vehicles (EVs) has been on the rise since 2010. European frontrunners are Norway (albeit not an EU Member State) and the Netherlands (IEA, 2016). Key barriers for

greater adoption of electric cars include battery costs and ranges (Haddadian et al. 2015), both areas where recent developments show promising signs (IEA, 2016). Additionally, adequate charging infrastructure, the lack of awareness, and a lack of consumer confidence have also been raised as barriers (IEA, 2016; European Commission, 2016c). Furthermore, there are various societal obstacles that hamper greater adoption, including for example perceptions about the safety of EVs, societal scepticism and lack of awareness about financial incentives (Haddadian et al. 2015).

Policy instruments that promote the adoption of electric cars can be divided into “technology push” policies and “market pull” instruments (IEA, 2016). Policies that push the technology support direct research, development and demonstration. Mechanisms from the second group include EV purchase incentives (e.g. sales tax exemptions, VAT exemptions, rebates at registration or sale), incentives for the use of EVs (circulation tax exemptions, waivers on fees, electricity supply reductions) and waivers on access restrictions such as access to bus lanes and tailpipe emission standards (IEA, 2016). A comparison of EVs markets in 30 different countries worldwide shows that financial incentives have a substantial impact on the adoption of EVs (Sierzchula et al. 2012). Additionally, the number of charging stations is a significant factor explaining the variation in EV adoption between countries (Sierzchula et al. 2012). See also the discussion on EV incentives below.

There are currently around 200,000 charging stations in the EU and it is envisioned that this needs to increase to 800,000 by 2020 (Teffer, 2017). The European Commission has allocated €800 million that will finance an accelerated roll-out of charging infrastructure for electric cars (Teffer, 2017) and has also established stakeholder platforms. The objective put forward by the Commission in the European Strategy for Low-Emission Mobility (EC, 2016c) is to make electrical charging as convenient as filling a tank such that an electrical car journey across Europe becomes possible.

Extensive research has been done on the environmental impact of electric vehicles (EVs) compared to that of internal combustion vehicles (ICVs). It is shown that EVs can deliver significant GHG emission savings compared to ICVs as long as the GHG intensity of grid electricity used to power the electrical vehicle is sufficiently low, the GHG emissions related to battery manufacture and disposal are brought down, and smart charging is adopted (Ma et al, 2012; Contestabile et al, 2012). The largest environmental impact of EVs is associated with the manufacturing stage; in specific the metals used in the battery pack can have toxicological impacts (Hawkins et al. 2012; Taglieferri et al 2016, AD Little, 2016).  

### 2.6 The potential impacts of connected cars and autonomous vehicles

As with many sectors of the economy, the increasing digitalisation of one or more aspects of the motor vehicle sector has the potential to significantly disrupt business activities and, in some cases, reduce a number of environmental impacts. Technologies are under development to increase the connectivity of vehicles, including internally (between the different operating parts of a vehicle) and externally (between the vehicle and other road users, as well as with the road environment itself) (McKinsey 2014). There may also be connectivity between vehicles and their manufacturers or other stakeholders who can utilise data from one or more vehicles in their activities. Vehicle connectivity will likely be linked to other innovations in the sector including electrification and alternative business models.

An opportunity which exists within the increased connectivity of vehicles is a varying automation level of driving functions. Within the spectrum of automation, a final level would allow vehicles to drive without a human driver. Automated driving is forecasted to bring a number of changes to the sector:

- A significant reduction in the rate of motor vehicle crashes – near zero crash risk could be achieved via deep neural network learning
- Increase the accessibility of mobility to those who are not typically able to drive (e.g. blind, disabled, young)
- An optimisation of congestion and traffic via vehicle connectivity
- A reduction of the socio-economic costs of traffic – as passengers can engage in alternative activities while driving
- Optimisation of driving speeds and breaking in order to reduce energy use
- Loss of employment in sectors focused on driving
New regulations and standards would need to be developed in order to ensure public/environmental health and safety. The connectivity and automation of vehicles will also bring new technological and data orientated challenges such as data privacy and cyber security linked to personal and fleet vehicles (Accenture, 2017). Further, liability uncertainties exist with respect to who would take responsibility for risks associated with automated drive.

It is not clear how automated vehicles will affect environmental impacts from motor vehicles in the future. Analysis of the travel, energy and carbon use impacts suggests that vehicle automation could reduce energy consumption and carbon emissions, however there is also a risk of a rebound effect, whereby efficiencies result in more travel overall, and thereby offsetting any net benefits (Wadud et al, 2016). In addition, the computing power required to enable autonomous driving could potentially substantially increase energy consumption by individual vehicles.

One scenario exercise carried out by Boston Consulting Group showed that vehicle automation in the city of Boston could bring about a number of benefits, including a reduction in the number of vehicles on the road, a reduction in total distance driven by the fleet, reduced travel time, reduced emission and a reduction in the number of parking spaces needed (BCG, 2017). In 2017, Tesla presented a prototype electrified and driverless HGV. Amongst reported benefits of the driverless model of the truck were that multiple vehicles could drive in convoy reducing drag.

The European Commission has recognised how quickly the technology is being developed and is already starting to act on connected and automated driving. In April 2017, the first conference on automated driving was held in Brussels, supported by two Horizon2020 projects. The transport strategy launched in May 2017 forecasts that driverless cars will be integrated with traffic by 2025.

2.7 Critical raw materials and the manufacture of motor vehicles

The manufacture of motor vehicles is reliant on the input of a range of resources to produce the necessary (and increasingly complex) components. Different vehicle designs, including conventional, hybrid and electric vehicles require different inputs. Critical raw materials are present in conventional, hybrid and electric vehicles. Key applications include (EC, 2018):

- Graphite (brake linings, exhaust systems, motors, clutch materials, batteries)
- Cobalt (lithium ion batteries)
- Platinum group metals (palladium, platinum and rhodium in auto-catalysts)
- Niobium (alloying agent in steel and nickel alloys)
- Rare earth elements (magnets, auto catalysts, and filters)

For conventional internal combustion vehicles, the recycling of parts and valuable materials contained in cars when they reach their end of life is not always optimised. One example is the loss of palladium in catalytic converters when end of life vehicles are exported outside of the EU. Research demonstrated that the recovery of platinum metal groups in the EU was less than 70%, even though 100% was technically feasible. Furthermore, recycling of auto-catalysts is around 50-60%, with an estimated €115 million worth of catalytic converters leaving the EU each year in end of life vehicles (EC, 2016b). Circular economy measures, including better recovery and recycling of auto-catalysts, present an economic opportunity for the EU automotive industry.

Growth in the market for HEVs and EVs will change the demand for specific materials necessary for their manufacture. EVs are likely to increase the demand for CRM – particularly those used in the manufacture of batteries. In 2015, the EU market for EVs required 510 tonnes of cobalt and 8330 tonnes of graphite. Currently, recycling of EVs is not expected to be widespread until 2025 – presenting a further risk of the loss of materials and their economic value.

Analysis of the use of critical raw materials and rare earth minerals in EVs suggests that although their availability is unlikely to hinder the development and uptake of vehicles, there is a need for the EU to reduce its dependence on the import of these minerals in the future (T&E, 2017a).

Policies can be used to support the effective recycling of materials in vehicles. Existing relevant measures include:
• Directive 2000/53/EU on end of life vehicles – requiring 95% reuse and recovery and 85% for reuse and recycling by average weight of a vehicle since 2015
• Directive 2005/65/EC on type approval of vehicles regarding their recyclability, which aims to ensure manufacturers allow parts to be reused, recycled or recovered
• Directive 2006/66/EC on barriers – covers automotive batteries
• Circular Economy Action Plan (COM/2015/614) – aims to reduce the leakage of raw materials from high value waste streams including end of life vehicles.

In order to ensure better recovery of materials from the motor vehicle sector, further policies may be necessary. Possible tools include a better information exchange on dismantling vehicles, clearer distinction between second hand and end of life vehicles, greater investment in recycling EV batteries, research into reducing the CRM content of motor vehicles and developing capacities for battery reuse (EC, 2018). A number of EU initiatives are already exploring the potential for battery reuse. The H2020 project SmartEnCity explores how batteries can be reused in electric taxis in Estonia (SmartEnCity, 2018).

2.8 Socioeconomic footprint of the sector

Motor vehicles are important thanks to their contribution to society and the economy in providing mobility, as well as contributing to economic growth and employment more widely. The sector can broadly be broken down into the production and sale of motor vehicles and associated supply chains, and as providing transportation services for both personal mobility as well as goods and services. In providing mobility, MVs enable companies to access their resources and transport their goods, as well as provide citizens with an access to goods and services, and participate in activities. It is an important source of revenue for various sectors of the economy and is a vital tool in the lives of everyday citizens. In the same time, the sector is a key driver for rising carbon emissions and air pollution, amongst other impacts.

Accounting for 6.8% of Europe’s GDP, the automotive sector provides over 12.5 million jobs; 3.3m in manufacturing, 4.3m in sales and maintenance, and 4.8m in transport. The sector’s current needs are highly dependent on the mining sector (for extraction of minerals), the oil and agricultural sectors (providing fuel for vehicles), and infrastructure development (roads and energy grids). This is exemplified through the feedback mechanism between the transport and industrial sectors in which industrial growth requires the construction of efficient modes of transportation which in turn feeds industrial growth.

While some congestion can serve as a signal of high economic activity, too much of it limits the capacity for markets to grow. Economists have measured these economic losses through hours lost in traffic jams (see box below which highlights an example from Belgium). Congestion costs Europe about 1% of its GDP every year and also causes heavy amounts of carbon and other emissions (EC, 2011a). Delays caused by congestion also result in increased stress levels, negatively impacting the health of the population. While degraded infrastructure may also result in economic losses, the cost of poor infrastructure and risks associated to motor vehicles is reflected in an average of 51 road deaths per million inhabitants in Europe in 2016, with significant variations across the continent.

Congestion impacts in Belgium

In Belgium the manufacturing of motor vehicles employs directly 15,000 people, sales and maintenance employs 85,000, while transport services employs 110,000 people. Cumulatively, the logistics and automotive sector represent 350,000 jobs and 8% of all jobs in the country. In Belgium, public spending on road infrastructure has decreased from 4.5 to 2% of GDP since the 1980s, well below that of its neighbours. This is reflected in a study conducted by the World Economic Forum, in which Belgium was ranked 22nd of 140 in terms of quality of infrastructure, and has fallen ten places in ten years, putting it well below its neighbours, Luxembourg, France, Germany and the Netherlands, which are all in the top 10. The Belgian vehicle fleet increased from 1 million vehicles in 1960 to over 5.7 million today. With over 80% of Belgian households owning at least one vehicle, studies found that over 50% of people interviewed drove five to six days per week. The small size of the country, continued urbanization and a high volume of commuters is giving rise to increasing congestion. In Europe, Belgium has shown the highest number of lost hours in traffic jams with an average of 44 hours spent in jams per year, which corresponds to between 1 and 2% of its GDP or 4 and a cost of 8 billion euros (OECD, 2013). Such figures
are calculated based upon the time lost in traffic, and the impact on productivity through human health implications. Accordingly, over a third of Belgians consider that their commute is inconvenient, with 70% of them feeling stressed at the workplace and exhausted after a day of work. Congestion is also inconvenient for employers, who face repeated delays. Transportation is responsible for 18% of GHG emissions in Belgium; and have grown 18% since 1990, while the overall emissions have fallen 8%. This is attributed to the demand growth for mobility and persistent dominance of diesel-fuelled vehicles – alternative fuelled were less than 1% of the fleet in 2016.

2.9 Environmental pollutant releases

Many sectors of the economy use and/or create pollutants during production processes, with the potential to damage human health and the wider natural environment. Data on pollutant releases is published in the European Pollutant Release and Transfer Register (E-PRTR\(^{122}\)). Data is classified by NACE code of the emitter, pollutant, location (including country), year, release medium and volume, and as such it is possible to track its evolution over time.

In the analysis below, we present data in volume terms, but also environmental impacts. These are calculated by applying coefficients reflecting the toxicity of different pollutants, taken from ReCiPe2016 LCIA, according to whether they were released via air, water or land. This allows the summation of different pollutant based upon the impact that they have on human health (measured in disability-adjusted life years, DALYs) and ecosystem health (measured as disappeared species per year, species.years).

\(\text{Figure 2.1 The health impacts of pollutant releases from the motor vehicles sector}\)

![Figure 2.1 The health impacts of pollutant releases from the motor vehicles sector](image)

Source: Author calculation, using data from E-PRTR and ReCiPe2016 LCIA.

Over 90% of the recorded releases of pollutants in the E-PRTR database do not have the required information (on chemical released) to allow them to be included in the analysis. From the data that is present, there is a clear increase in environmental impacts from 2014 onwards (see Figure 2.1). This is a result of substantial increase in emissions from Germany (see Figure 2.2)– primarily in terms of CO2, but also carbon monoxide (from the use of energy in the sector). Carbon monoxide releases also increase in Poland from 2014 onwards, related to the processing and production of metals for vehicles.

\(^{122}\) http://prtr.eea.europa.eu/#/home
3  Current trajectory – direction of travel

3.1  Research questions addressed

- Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?
- What has driven changes over time (post-2000), and what changes are expected to occur in the future?
- How have the answers to all of these questions changed over time, and how are they forecast (modelled) to change in the future?
- How do SMEs differ from other firms in their answers to all of these questions?
- Are there examples of striking differences to any of these questions between Member States (and if so, why)?

3.2  Overview

Changes to the motor vehicle industry since the millennium have been driven by a number of factors at the European and global level.

Globally, socio-economic and demographic changes have led to increasing demand for vehicles. Although these changes are most evident outside of Europe, notably in emerging markets such as China and India, growth in demand has been highly relevant to EU manufacturers. Between 2015 and 2025 the number of cars on the road is anticipated to grow from 1.1 billion to 1.5 billion (WEF, 2016). In the EU, the motorisation rate is already very high at 573 vehicles per 1,000 inhabitants in 2015, compared to 83 per 1,000 in China – reflecting big differences in the maturity of markets (Vieweg et al, 2017).

Emerging markets also increasingly contribute to car production. In 2016, for the first time, China overtook the EU as the world’s biggest producer of cars. In 2001, the EU accounted for 35% of world motor vehicle production; by 2016 this had decreased to 23%, though in absolute terms the EU’s production has grown over the same period (ACEA, 2018a).

The state of financial markets in Europe is a key determinant of the output of the motor vehicle industry. The financial crisis in 2008 had a strong impact on car registrations in the EU. Following the financial crisis car manufacturers in Europe were faced with surplus production, as the already mature domestic market was faced with austerity and tightening credit availability. In general, motor vehicle registrations tend to correlate with GDP growth in the EU (ACEA). Europe’s motor vehicle industry has reported declining profits so far this century. In 2007, the European automotive industry had record profits of €15 billion. By 2012, this became a financial loss of €1 billion. Overcapacity and competition in a saturated market keep profits down. Data suggests that in 2016 the industry returned to profitability (Strategy&, 2017).

Investment in research and development have brought some changes to motor vehicles in the last two decades, and represent significant sources of expenditure. In 2015, it is estimated that the ACEA members invested €41.5 billion in research – equal to more than 5% of revenues. Key areas of research are safety, digitalisation/connectivity and environmental performance (ACEA, 2017b). Electrification and autonomous vehicles are widely discussed, but other technologies such as light weighting, powertrain improvements and emission abatement systems have been more relevant in delivering improvements in environmental performance so far.

The international agenda on climate change and the 2016 Paris climate agreement are highly relevant for the motor vehicle industry, but it is still unclear how industry and mobility will respond to high level environmental objectives. The EU is committed to agreements to reduce emissions as part of the Kyoto Protocol. As motorised transport is one of the major contributors to CO₂, growing pressure to reduce emissions will also have to address transport.

There has been limited action to address transport emissions though with no net impact – reflected in the overall increase in emissions from transport since 1990. Nevertheless, evidence suggests that EU emissions standards have reduced average tailpipe emissions from passenger vehicles. This can be seen by contrasting increased vehicle activity (based on passenger km) against total stock emissions for cars (EC, 2017e).
Though emissions reductions can be achieved in conventional engines, more stringent targets on emissions reduction will likely lead to more investment in e-mobility, including electrical and hybrid power trains. However, a rapid change is not anticipated; by 2020 it is expected that conventional combustion engines will still account for more than 90% of vehicles on the road (McKinsey, 2013) – this reflects the fact that stock turnover is not particularly rapid, with an average age of in-use vehicle of 10.7 years across the EU as a whole.123

As discussed above, the emissions scandal involving VW, and subsequently other OEMs, had a significant impact on the industry. For VW the immediate implications were financial and political, as its business reported losses and chief executives faced criminal charges. For the wider industry this has had potential spillover effect on the demand for diesel vehicles, and helped to create a political environment which is more conducive to supporting alternative powertrains. In Germany, one implication of the scandal was that the highest court in Germany, the Bundesverwaltungsgericht (Federal Administrative Court), made provisions for German cities and municipalities to ban diesels (and some polluting petrol vehicles). A number of German cities including Hamburg, Dusseldorf and Stuttgart will introduce bans shortly (Bundesverwaltungsgericht, 2018).

3.2.1 The makeup of the market and the role of SMEs

The motor vehicle industry is dominated by a number of OEMs headquartered in Europe, as well as in third countries; notably the USA, Japan, South Korea and increasingly China. Of the ten largest OEMs, four are based in the EU: Groupe PSA, Renault, Fiat Chrysler Automobiles, and Volkswagen AG. The ACEA (European Automobile Manufacturers Association) represents 15 EU car, bus and truck manufacturers, accounting for over 95% of EU’s production capacity. Despite concentration of production in a few large corporations, SMEs do play an important role in the industry. For example, more than 3,000 businesses, of which 2,500 are SMEs, provide parts to car manufacturers. About 75% of a vehicle’s original equipment components and technology is sourced from automotive suppliers (European Parliament, 2013).

Despite the continued dominance of the main OEMs, a number of factors also present opportunities as well as risks for business – both in terms of SMEs and other non-traditional actors. These factors include developments in emerging markets, the adoption of new technologies, and changing preferences about vehicle ownership.

New technology in the automotive sector is opening opportunities for different kinds of businesses. Tesla provides a notable example; the United States’ youngest car manufacturer, founded in 2003, has the US’ best-selling electric car the model S, which is now also the best-selling motor vehicle in Norway (Statista, 2018a). Tesla has been a frontrunner in the development of technology such as affordable battery packs (60-100kWh), home charging infrastructure (Tesla Supercharger), and semi-automated drive (Tesla autosteer). Tesla also has promoted changes to selling strategies, opting for online sales and company owned showrooms, as opposed to the dealership model favoured by traditional OEMs.

The emergence of EVs could also pose a risk to some of the parts suppliers in the conventional auto manufacturing industry. In general, EVs contain fewer components and moving parts; on average 1,400 parts in an ICE powertrain compared to 200 in a BEV powertrain. Data on the manufacture of ICEs compared to EV motors suggests that in a year a single employee could produce 350 engines or transmissions, compared to 1,600 EV motors respectively (ING, 2017). This is the case because the production of many new electronic parts can be automated. In the UK, a major labour union warned of potential job losses if workforces were not retrained and redirected towards the development of EVs (Financial Times, 2018). While supply has previously focused on providing parts such as gear boxes, exhaust pipes, and injectors, new products will supersede these, such as battery manufacture, regenerative braking systems and electric motors (T&E, 2017b).

Non-OEMs, including technology developers such as Google, Apple, Amazon, Baidu and Uber, are investing in the automotive sector, and represent a diverging market. These actors are particularly focused on

different activities within the value chain, including automated vehicles and shared mobility (McKinsey, 2016a).

Alongside new technology, emerging markets have also presented opportunities for new actors in the motor vehicles sector. This has been clearly evident in China, where the market is more diversified in terms of actors, represented by 76 OEMs and 184 vehicle assemblers (PwC, 2016). China’s domestic automotive market has grown by more than 10% per year for a decade, and since 2009 China has been the world’s largest market for vehicles, with sales of 28 million units in 2016 (including 517,000 EVs). China is also the largest manufacturer of vehicles, accounting for 30% of global production (Statista, 2018b). China’s vehicles sales are anticipated to grow by 5% annually to 2020 (McKinsey, 2016b).

A number of government policies have sought to support domestic car manufacture in China. The Chinese government offers a subsidy of up to 9,800 USD towards all electric vehicles\(^{124}\). The “Made in China 2025” strategy aims to transition from low cost mass production towards higher value-added manufacturing – it targets ten key sectors including motor-vehicles, and specifically EVs (Chinese Ministry of Industry and Information Technology, 2015). Targets within the Made in China 2025 strategy include to:

- Increase sales of pure electric vehicles to 1 million units by 2020 and 3 million units by 2024
- Increase overseas sales of Chinese manufactured vehicles relative to domestic sales to 10% - in 2016 overseas sales account for 2.5% of the overall production volume in China (Statista, 2018b).

A number of established OEMs, i.e. those from Europe, Japan and the US, are present in China, but in general were previously only able to manufacture cars locally by entering in joint ventures with Chinese manufacturers – sometimes called the 50:50 rule. In April 2018, the Chinese Government announced that this rule would be abolished by 2022. The EV market in China is currently dominated by domestic manufacturers, including BAIC, BYD, and JAC. The Ministry of Industry and Information Technology defines the EVs which are eligible for subsidies. Arguably, these subsidies create a barrier for imported vehicles, which are not eligible. It was announced in 2016 that subsidies would be phased out by 2020 (Export.gov, 2017). Overall, growth in the Chinese EV production is seen both as a measure to reduce air pollution in urban areas, and a support to the global competitiveness of Chinese manufacturers.

3.2.2 Differences across the Member States

There are big differences between member states both in terms of the development of their motor vehicles sector and the nature of mobility more widely. Europe’s domestic market for vehicles and production activities varies considerably. In general, the western European market is very mature, and car purchases mostly represent replacement. Germany represents Europe’s biggest market for vehicles and is also its most important manufacturer – accounting for around 20% of European sales and 19% of global production in 2015. Productive capacity is more than double that of each of Spain, France and the UK, the next biggest producers (Germany Trade & Invest, 2016).

Most EU countries demonstrate a high rate of motorisation. The highest rates in 2016 for the EU are Luxembourg (662), Italy (625), and Malta (615), with the lowest rates in Romania (261), Hungary (338) and Latvia (341) (Eurostat, 2018a). Some newer member states, such as Poland, Bulgaria and Romania, still have a growing rate of motorisation (see Figure 3).

There is also a difference in modal share between countries. In some MS, such as Portugal and Lithuania, motor vehicles accounted for close to 90% of transport in 2015 (Eurostat, 2018b). In contrast, the member state with the highest share of rail transport were Austria and the Netherlands, with 12.0% and 10.8% respectively (EEA, 2017b). According to a Eurobarometer survey, those in EU15 tended to use a car more than those living in EU13 countries (57% compared to 45%) and, also tended to use less urban public transport (16% compared to 27%) (EC, 2014a).

Member states display significant differences in the age of the vehicle populations. Across the EU28 cars are on average 10.7 years old, however some countries have much older fleets. For example, in Estonia, Latvia,
Lithuania, Poland and Romania the average passenger vehicle is more than 15 years old\textsuperscript{125}. This is linked to the fact that there is a sizeable flow of second-hand cars from the EU15 to EU13 countries, with Germany accounting for 42% of all imported cars (EC, 2014b). In some MS imported second-hand vehicles account for more than 60% of the annual registration of new cars\textsuperscript{126}. The trade of second-hand vehicles suggests that the uptake of EV and other low emission vehicles will be slower in some MS than others. Following “diesel-gate”, some analysis has suggested that incentives to “trade-in” diesel vehicles in MS such as Germany will result in the export of these vehicles to CEE countries, thereby relocating the air pollution associated with highly emitted vehicles prolonging their impacts and distorting markets (T&E, 2018a).

\subsection*{3.2.3 How the demand for motor vehicles is evolving}

Demand for motor vehicles can be measured in a number of ways; common methods include the overall motorisation rate (measured in the number of cars per 1000 people), and in the rate of new registrations (number of new passenger cars per year). In the EU the motorisation rate varies from over 660 in Luxembourg to 260 in Romania. The EU average is around 500 (Eurostat, 2018a). In most MS, the size of the passenger car fleet has grown in the last 5 years.

In Europe’s mature vehicle market future demographics are likely to influence the demand for motor vehicles. The EU Reference Scenario 2016 (EC, 2016d) suggests that after 2040 overall growth in passenger transport will slow because of stagnant population sizes across the EU. In Germany, the population is expected to decline from 81 million to 77 million between 2014 and 2040 (Shell, 2014). This could lead to an overall reduction in vehicle demand (Friedrich Ebert Stiftung, 2015).

There is significant variation between member states in the ratio of petrol to diesel cars. For example, in 2016 nearly 79.6% of new cars registered in the Netherlands were petrol, whilst in Croatia 77.2% of new cars were diesel (Eurostat, 2018a). Eurostat data shows that between 2014 and 2016 there was a growth in the share of vehicles using alternative fuels. However, on average in 2016 alternative fuels only made up around 3% of new registrations, and significantly less than this of the total vehicles on the road. Data for early 2018 shows that electric and hybrid vehicles account for 3.2% of new registrations (see Figure 3.1).

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{figure31.png}
\caption{New passenger cars by fuel type in the EU Market share (\%) new registrations Q1 2018}
\end{figure}

Source: ACEA, 2018

\textsuperscript{125} http://www.acea.be/statistics/tag/category/average-vehicle-age
\textsuperscript{126} True for BG, CY, CZ, GR, LV, MY, PL, RO and SK. p.iii

3.2.4 How vehicles are changing over time

According to the ICCT (2017c), the average fuel efficiency of new vehicles in the EU in 2016 is around 118gCO₂/km on the NEDC test cycle, which is 30% lower than in 2000. This is still 25% higher than the 2021 target of 95gCO₂/km.

For passenger cars, research from Ricardo AEA (CCC, 2015), suggests there are a number of fuel-efficient technologies that can improve fuel efficiency of conventional ICE vehicles. However, these technologies alone are unlikely to be deployed fast enough to meet the EC's target for 2021 and are insufficient to meet the proposed targets for 2025 and 2030. Meeting these will require the deployment of advanced powertrains though a combination of hybrid, plug-in hybrid and battery electric vehicles.

For heavy good vehicles, there has not been much progress over the past decade in part due to a lack of targets on fuel efficiency. Additionally, there are challenges in the lack of incentive for fuel efficiency within the market for haulage services:

- Fuel expenses are covered by the clients as part of standard contracts
- The haulage sector has a large number of SME operators that lack the capacity to finance investments in more fuel-efficient rolling stock
- Technical challenges in supporting advanced powertrains

In the short to medium term, the deployment of fuel efficiency technologies to conventional ICE powertrains will need to play a substantial role in ensuring HGV reach the proposed targets of 15% by 2025 and 30% by 2030. Data on available fuel-efficient technologies for heavy duty vehicles from the ICCT (2017e) shows potential fuel efficiency improvement of 27% in the short to medium term and as much as 43% by 2030. The proposed standards for HGVs, announced in 2018, require emissions of new vehicles to be 15% lower in 2025 than 2019, and reach 30% below 2019 levels by 2030. The ICCT analysis suggests that this should be possible through the deployment of fuel-efficient technologies, although it is likely that advanced powertrains will make some inroads into the market (for example BEVs for use cases that require relatively low mileage, and FCEVs in use cases where long running distances are required).

3.2.5 Modal shift in transportation demand

One approach to improve the sustainability of the motor vehicle sector is to support a modal shift through investment in alternative, less polluting modes of transport. Promoting multi-modality is one of the measures supported in the European Strategy for Low-Emission Mobility (EC, 2016c).

In many areas, there is very strong evidence demonstrating economic, social and environmental benefits of alternative transport modes, and the need to develop policies to support a shift in the modal share of transport away from personal motor vehicles towards a greater share of public and mass transit systems (e.g. trains, buses, taxis, and ferries) as well as soft mobility (e.g. walking and cycling). Potential benefits from reducing the modal share of motor vehicles includes reduced emissions, reduced congestion, and improved public health. Some of these benefits are also outlined in the European Commission’s Strategy for Low Emissions Mobility (COM/2016/501).
For public transport systems, net efficiencies are often determined by urban population densities and travel distances. Investments in infrastructure for alternative transports have arguably declined in comparison to other points in history. This is notably the case for rail transport, which has a declining share for journeys travelled. Today rail travel in Europe accounts for 8% of the passenger-km and 17% of the tonne-km, but just 2.5% of the emissions (Vieweg et al, 2017) (see Figure 3.2).

Short distances for typical journeys could easily be covered on foot or by bicycle. In mechanical terms, the bicycle presents the single most efficient form of transport available today, but remains underutilised for many journeys. Soft mobility provides additional benefits in comparison to other forms of transport as a result of being zero emission, low noise, low cost, active/healthy, and placing low demands on infrastructure or public investment. Some European cities, such as Utrecht, Copenhagen and Amsterdam, benefit from high rates of cycling. Research by the European Cyclists Federation (ECF) demonstrated that cycling creates economic benefits of €513 billion – with the most significant benefits relating to health.

Promoting soft mobility cannot be achieved by single measures, but is more often the result of a range of initiatives, which simultaneously reduce the incentives to drive and make it easier to walk or cycle, resulting in changes to social norms.

Innovative solutions relevant to motor vehicles, including electrification, social enterprise and the digital connectivity of vehicles, can be just as relevant to other modes of transport. E-bikes for instance may help to overcome barriers to cycling, such as steep gradients in local geographies or old age. Likewise, docked and dockless bike sharing systems provide citizens with the convenience of cycling without owning a bike. Mobile applications and big data can also contribute to personal and network efficiency of public transit systems, reducing journey times and operating costs.

Overall, the evidence suggests that a significant modal shift away from motorised transport is not taking place in the EU, although there are measurable differences between Member States. This can be seen in Figure 3.3, which demonstrates that overall demand for passenger transport has grown on average in the EU28; however demand for car transport has grown at a greater rate. Some Member States, notably Italy and the Netherlands, show an overall decline in the demand for car transport (EEA, 2015).
3.2.6 Car sharing and alternative business models

Car sharing systems present one opportunity to reduce the environmental impact of the motor vehicle sector. Car sharing has the potential to improve system efficiency by reducing structural waste and idle capital. In the EU, cars are parked 92% of the time, with 1% of the time spent in congestion, 1.6% looking for parking, and only 3% of the time spent driving. Furthermore, the average European car has 5 seats, but only carries 1.5 people per trip (EMF, 2015). Sharing vehicles through different systems can help to reduce the inefficiency illustrated by these statistics.

Research has shown that car sharing can reduce vehicle ownership and vehicle kilometres travelled among users (Shaheen et al. 2012). Case studies in the Netherlands, Portugal and Ireland demonstrate that car sharing brings significant CO₂ emission savings (Nijland and van Meerkerk, 2017; Baptista et al. 2014; Rabbitt and Ghosh, 2016). Other benefits associated with car sharing are reduced pressure on parking spaces, more equitable access to jobs and public services, and congestion reduction (ITF 2017). A potential benefit of service models for motor vehicles is that the quality of rolling stock can be increased – this could for example make an electric vehicle accessible to a user who otherwise might not be able to afford to own one outright.

In general, car sharing and approaches which provide access to motor vehicles without ownership will not necessarily help to reduce the environmental impacts and dependencies of the motor vehicle sector. Car sharing could be beneficial where it:

- Reduces the idle stock of motor vehicles (material efficiency)
- Reduces the number of vehicles on the road (spatial efficiency – to reduce congestion)
- Reduces the need for individuals to own personal motor vehicles (economic efficiency)
- Facilitates the use of more fuel efficient or zero emission vehicles (energy efficiency)
- Facilitates multi-modal transport, as individuals can choose more readily to drive at different stages in their journey or from day to day (system efficiency).

In 2015, there were 2.1 million users sharing 31,000 vehicles in Europe\textsuperscript{127} (Boston Consulting Group, 2016). Forecasts on car sharing growth suggest that there may be over 15 million car sharing users by 2020 in Europe (see Figure 3.4).

There are various car sharing systems including business-to-consumer (B2C) services, where a car sharing operator distributes cars through a network, and peer-to-peer (P2P) services, where short-term access to a private vehicle is provided (Shaheen et al. 2012). In the traditional B2C model members can access the car for short-term daily use by making a reservation and paying a charge based on the time and miles they drive (Shaheen et al. 2012). Shared cars may be picked up at a fixed station (traditional system) or used for one-way trips in a free-floating system without the need for reservation.

Various policies can be put in place by local governments to promote car sharing including financial assistance for the start-up of car sharing systems, raising awareness on the benefits of car sharing, provision of public parking spaces for shared vehicles, establishing development requirements on spaces for car sharing, and fleet-sharing agreements between local governments and car share businesses (Deloitte, 2015; Dentel-Post, 2012). A combination of these policy approaches allows for these options to reinforce one another (Dentel-Post, 2012).

Uncertainty exists over the extent to which car sharing schemes can actually lead to net reductions in car use and/or the number of cars on the road at any time.

Analysis by McKinsey suggests that shared mobility will slow the annual rate of car sales but that overall car sales would continue to grow. It argues that there will be a decline in private car sales, particularly in dense urban areas including many cities and suburbs in Europe and North America. However, reduced private sales would be (partly) offset by increased sales in shared vehicles, which would have a higher turnover due to higher utilisation. The higher turnover of shared vehicles will also facilitate the introduction of new and

\textsuperscript{127} Data covers Europe, Russia and Turkey
\textsuperscript{128} Data covers Germany, UK, France, Italy, Switzerland, Austria, Netherlands, Sweden, Spain, Belgium, Norway, Denmark, and Sweden
more efficient technology. The major driver of sales would continue to be linked to macroeconomic development in emerging markets, i.e. outside of the EU (McKinsey, 2016a).

A review of existing studies on car sharing suggests that car sharing could deliver a net reduction in car use. Ride sharing apps for example can support a reduction in the number of vehicles on the road and the number of vehicle kilometres driven, as well as supporting a shift to more multimodal travel. Car sharing schemes (either point to point or free floating) can help to reduce car ownership – an estimated 5 – 15 cars can be replaced by a single car added to a car sharing fleet. Additionally, while long distance car sharing schemes do compete with rail and coach services, they can also help to increase car occupancy and reduce per passenger kilometre emissions (T&E, 2017c).

There is some evidence which questions the benefits of these kinds of services, in particular how they interact with other modes of transport and travel behaviours in general. For example, study of ride hailing services in the cities of Boston, Chicago, New York, Seattle, San Francisco, Los Angeles, and Washington, D.C. showed that these new services were adding additional journeys and often replacing trips taken by public transport or soft mobility (e.g. cycling or walking) (see Figure 6) (Clewlow and Mishra, 2017).

Figure 3.5 How ride-hailing users would travel if Uber or Lyft were unavailable in seven US cities

Source: Clewlow and Mishra, 2017.

3.3 The impact of changing consumer preferences

This section primarily focuses on personal motor vehicles and its conclusions can thus not necessarily be extended on to other types of motor vehicles such as buses, trucks, motorcycles. Changing consumer preferences have been driving a shift in the type and characteristics of motor vehicles, as well as their ownership model. Nonetheless, current trends and all major predictions forecast continued growth in motor vehicle demand (PwC, 2016). In terms of consumer preferences, the three major trends that can be identified are:

- Demand for low emission vehicles continue to grow at the expense of diesel-fuelled vehicles.
- Advanced driver assistance systems (ADAS) and the move toward autonomous driving is driving an innovation rush across both the traditional automobile industry and high-tech firms – shedding light on an increasing number of alternative business models and cross-sectorial partnerships.
- The combination of alternative business models, socio-economic factors and changes in urban landscapes and public awareness is contributing to changes in ownership trends, as well as to a limited extent, changes in demand.
Since the “diesel-gate” scandal, carmakers have rushed towards the production of “cleaner” electric vehicles (EV). Nearly all major automakers have announced EV plans for the next few years. Pressure on the industry has intensified since a growing number of cities and governments have stepped up efforts to phase out diesel and petrol vehicles altogether in a near to medium-term.

Furthermore, the recent European Commission proposal for post-2021 CO₂ standards incentivizes EV sales by awarding less stringent CO₂ reduction targets to manufacturers that exceed Zero Emissions Vehicle sales targets. Financial incentives at the national level will also contribute to demand for low emission vehicles (ICCT, 2017a). Some predictions suggest that electric car sales will reach 28 percent, and those of electric buses could reach 84 percent, of their respective global markets by 2030 as a result of declining battery costs and large-scale manufacturing. The rapid growth of electric buses (outpacing electric cars) stands out particularly because “in almost all charging configurations [they have] a lower total cost of ownership than conventional municipal buses by 2019” (Bloomberg New Energy Finance). Nonetheless, in certain instances, the decline in diesel car sales has been offset by greater demand for petrol-fuelled cars, with important environmental implications (ACEA, 2017a). Moreover, underinvestment in electric vehicle recharging infrastructure across Europe, and the strong appeal of petrol and hybrid vehicles are still hindering the switch to electrically-chargeable cars on a large scale.

ADAS features such as automatic emergency braking and Vehicle to Vehicle (V2V) communication are increasingly being installed in vehicles. This is mostly driven by falling technology prices and the popularity of such features with consumers, governments and insurers. The associated long-term result will likely be the availability of fully autonomous, or self-driving vehicles. This spur in innovation is gradually blending the tech and automotive sector with the effect of reshaping the landscape of actors. While large technology firms like Apple and Google are developing vehicle sensors and ride-sharing platforms to gain competitive advantages in autonomous driving and capitalize on mobility as a service, some OEMs are moving to strategic partnerships with Blockchain/tangle technologies to cope with demand and grid integration for EVs (Deloitte, 2018). Furthermore, increased demand for EVs is driving suppliers higher up the supply-chain (in the traditionally oil and gas sector) to diversify their portfolio and contribute to infrastructure development (such as installing charging stations) (CleanTechnica, 2017).

Germany provides some valuable insights in how changes in the socio-economic situation of young adults in the country is linked to a decrease in car ownership, in contrast to older age groups. Contrary to the popular narrative, Germany is experiencing two major trends in regard to motor vehicles:

1. There is a trend towards one car per driver; car sharing within households is decreasing, not growing; and
2. There is a trend towards using privately owned cars less; i.e. there is a trend toward owning instead of using, as well as an increase in multimodality of young drivers (Kuhnimhof, 2017).

In spite of the growing political support for soft and shared mobility, and an increasing number of commuters in Europe, these trends reveal a limited impact on motor vehicle demand. Driving public awareness campaigns, as well as a number of local and institutional initiatives, are stressing the negative impacts of air pollution, and promoting the significant health and economic value of increasing bike-friendly infrastructure and public transport, including electric buses (European Commission, 2016a).

While growing support for the use of public transport through fiscal incentives and improved services may contribute to the growing number of commuters in Europe and theoretically contribute to a decrease in the demand for individual motor vehicles, no evidence suggests any significant impact on demand currently (Eurostat, no date).

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129 For example, BMW plans to build a fully electric version of the Mini at its Cowley plant near Oxford from 2019. Volvo has announced that from the same year, all its new models will have an electric motor. VW has earmarked €70bn (£62bn) to produce battery-powered versions of all models by 2030.

130 https://www.iscapeproject.eu/about/

131 http://www.cleanairbxl.be/

132 https://ec.europa.eu/transport/themes/urban/cleanbus_en
In summary, other than the switch to EVs as a result of incentives and air pollution awareness, environmental preferences -per say- may not have a significant impact on the demand. The shift in ownership due to socio-economic factors and thanks to the sharing economy could potentially improve the efficiency of motor vehicles during their use phase, while increased infrastructure and technological advances may better integrate EVs into our energy systems. Despite some identifiable trends in terms of urban mobility, these fall short of having any significant impact on the overall demand for motor vehicles at this point.

3.4 Sustainability in the motor vehicles sector

The initial focus of regulations on motor vehicles was to improve passenger safety. Increasing attention has been given to regulating the environmental performance of motor vehicles, notably fuel consumption and the emission of specific pollutants. In this way, regulations have been applied to reduce vehicles’ consumption of fuel, their emission of pollutants including CO₂ and nitrous oxides, as well as addressing other issues such as noise pollution.

Regulations in European countries have focused on gradually increasing the stringency of emission limits for new vehicles – as defined through the Euro standards (1 – 6). However, existing standards and monitoring systems came under scrutiny after “diesel-gate” revealed several motor vehicle manufacturers were deliberately cheating emissions testing procedures and exceeding legal limits on pollutants. Unreliable information on CO₂ emission values for vehicles poses a barrier to the effectiveness of measures. As mentioned, evidence shows that CO₂ emissions in tests compared with real-world values are increasingly diverging. Between 2006 and 2016 the gap between test values and real-world values increased from 14% to 42% (ICCT, 2017e). This evidence suggests that improvements to emissions are potentially smaller than reported.

Since September 2017, new car models have to pass new type approval procedures, both via the on the road Real Driving Emissions (EC, 2017b) and in lab World Harmonised Light Vehicle Test Procedure (EC, 2017c), in order to drive on European roads. Further regulations exist for mobile air-conditioning units (Directive 2006/40/EC), and for reducing noise pollution from motor vehicles (Regulation (EU) No 540/2014).

At national, sub-national and city levels there are further regulations which can be applied in order to reduce the environmental impact of vehicles. At city level, in recent years there has been increasing interest in the application of the low emissions zones (LEZs). In Europe, many cities and towns have established LEZs. Brussels Capital Region, for instance, introduced a LEZ in January 2018. LEZs also provide a basis for regulating older vehicles which were introduced before more stringent emissions standards existed. Even stricter zero emissions zones (ZEZs) are foreseeable in the future.

Regulations on emissions from vehicles present a policy option which could help to dramatically reduce negative impacts from car exhaust emissions. These measures might also have complex implications for the development of the sector more widely, for example favouring the development of low emission, hybrid and electric vehicles, and how the production of these vehicles remains competitive on the global market. The development of LEZs reflects the extent of the air quality crisis in some European geographies, and the potential for municipal policy makers to develop more stringent policy than EU-wide standards.

Sustainability may have a different meaning depending on whether one considers the sustainability of motor vehicles industry, or sustainability of mobility as a whole. At the European level there are a number of measures in place to define what sustainability might mean in different contexts.

In November 2017, the Commission launched a new Clean Mobility Package containing a number of measures with a particular focus on reducing the emissions from motor vehicles, and in order to “boost the demand and supply of clean mobility solutions” (EC, 2017d). Earlier in May 2017, the Commission launched the Europe on the Move Communication (COM/2017/0283). Linked to the Communication there are a number of legislative proposals covering issues such as road infrastructure charging (Eurovignette and European Electronic Toll Service), haulage services and goods vehicle leasing. The European Strategy for Low-emission Mobility provides a broader overarching strategy for sustainable mobility, focusing on increasing the efficiency of the transport system overall, promoting the electrification of transport, moving
towards zero emissions vehicles, and engaging local authorities in promoting more sustainable modes of transport including public transport and active travel, in order to reduce pollution and congestion.

Motor vehicles are addressed in the 7th EAP, in the following areas:

- To integrate the need to transform to an inclusive green economy in wider policy areas (transport). (§:11)
- Address transport-related air pollution and CO2 emissions (§:19, 22, 33, 47)
- Reduce unsustainable land use change linked to transport (§:23)
- Integration of transport with sustainable urban planning and green infrastructure (§:87, 95)

Key issues for sustainability in the motor vehicles industry will relate to how cars contribute to EU and national level environmental objectives in a number of areas, including GHG emissions, air pollution, noise pollution and land use change. Transport sector emissions have continued to rise in recent years, suggesting that the industry is so far failing to contribute to over-arching emissions reduction targets. Road transport is responsible for 93% of the transport sector’s emissions. The European Strategy for Low-Emission Mobility establishes a non-binding target for the transport sector to reduce emission by 60% compared to 1990 levels by 2050 and be on a pathway towards zero emissions (EC, 2016c).

In contrast to these objectives, the EU Reference Scenario 2016 (EC, 2016d) forecasts that emissions in the transport sector (excluding international maritime) will decrease by 8% between 2010 and 2050. The biggest reductions from road transport are expected to be driven by emission standards on new cars.

3.5 Internalising the external costs of the sector

Fair and efficient pricing is a key element in developing a sustainable mobility future for Europe. In the case of road transport and motor vehicles, it is widely acknowledged that significant costs are not internalised - consequently the polluter pays principle is not applied and transport users do not pay for the costs they impose on others (EEA, 2016).

The European Commission has previously researched the extent to which the costs of transport are internalised and carried out analysis on potential remedial tools. Supporting better transport pricing has been promoted in a number of policy documents from the European Commission, including the 2011 White Paper on Transport, and the 2016 European Strategy for Low-Emission Mobility (EC, 2016c). The 2016 strategy suggests that there should be a move towards distance-based road charging based on km driven in order to correct externalised costs.

A number of studies have been conducted by contractors for the Commission to assess the size of the external costs for transport in the EU and to provide a handbook on external costs, which provides methodologies for carrying out an assessment. Handbooks on external costs were published in 2008 (EC, 2008a) and updated in 2014 (EC, 2014c). The European Commission Staff Working Document (SWD/2013/269 final) provides a summary of measures that can internalise or reduce transport externalities. For different transport modes, and specifically for road transport, it identifies measures to support the internalisation of costs at different governance levels (EC, 2013).

Europe’s major motorways within the trans-European Transport Network (TEN-T) are one area where the issue of external costs is particularly acute. TEN-T recommends that the full costs on public health and the environment should be used to determine road tolls, with the 60% addition proposed by the EP Transport Committee as a minimum. It also recommends compulsory HGV tolls with a minimum rate on the entire EU road network, for vehicles with a weight over 3.5 tonnes, as well as additional surcharges for environmentally sensitive areas (TEN-T, no date).

Some studies have attempted to assess the costs of road transport;

- Per passenger km costs of cars and aviation are about four times that of rail transport. The total external costs of transport in the EU (including Switzerland and Norway) were €500 billion in 2008, roughly 4% of GDP. In addition, annual congestion costs of road transport delays are between €146 and €243 billion (around 1 to 2% of GDP). Overall road transport accounts for 93% of the external costs – this is explained by both the larger modal share of road transport and the higher per passenger km external costs. The
research also shows that the marginal costs for road transport are much higher compared to rail. This is particularly the case for urban road transport (CE Delft et al. 2011).

- In 1998, external costs of transport caused by environmental damage (noise, local air pollution, and climate change) and accidents were estimated to be 4% of GDP. For road transport the level of cost recovery was 30% compared to 39% for rail. Overall road traffic accounted for 83% of the external costs (ECMT, 1998, EEA, 2016).

- Estimates of costs of road infrastructure suggest that costs are between €600,000 and €800,000 per motorway kilometre. Costs for HGVs varied according to weight but were between €0.40 and €0.19 per kilometre (EC, 2008b).

It should be said that high levels of public and corporate investment in infrastructure to support motor vehicles have, to an extent, created a lock-in to a built environment in which the private ownership of cars is unreasonably favoured and facilitated. Research into environmentally harmful subsidies (EHS) has shown that in the EU vast sums of money support fossil intensive motorised transport. Of the €112 billion in fossil fuel subsidies that exist in Europe, 44% (or €49 billion) relate to the transport sector (ODI, 2017). In Germany, subsidies for fossil-based transport amounted to €28 billion in 2012 (Umwelt Bundesamt, 2016).

3.6 Current European policy measures

Across Europe there are schemes in place to promote the uptake of advanced powertrain vehicles, in most cases these are subsidies for purchases, either through tax reductions and exemptions, or direct contributions towards a vehicle purchase. Other incentives include investments in infrastructure and zoning measures at the local level. Research has shown that there is a strong correlation between the level of support for the uptake of EVs and the level of uptake (ICCT, 2016). A 2015 paper which assesses global policies to support electric mobility outlines a number of relevant measures:

- Incentives in the form of direct subsidies and tax credits for electric vehicles
- Incentives in the form of rebates to return old vehicles
- Feebates combining incentives and repressive elements
- Quotas for OEM to produce a minimum number of zero emissions vehicles
- Penalties and taxes on conventional vehicles
- Rules to reduce vehicle fuel consumption per km
- Support for R&D
- Investments in charging infrastructure
- Road tax exemptions
- Road use measures – such as free use of bus lanes or parking

The study examines which type of measure would be most effective from a consumer perspective based on surveys in 20 countries across five continents. The study concluded that charging networks on motorways were the most important driver, while cash incentives were an additional driver. It also notes the risk of free riders with cash incentives, as many EV purchases may have taken place anyway (Lieve, 2015).

The ACEA provide an overview of the tax incentives in place for electric vehicles in the EU MS (ACEA, 2018b). This overview focuses on fiscal measures only and does not include other types of policy. It shows that most MS have some form of fiscal incentive in place for EVs, Only Croatia, Estonia and Lithuania do not have any kind of fiscal measure in place. The box below provides an overview of the measures in place in France, Germany and the Netherlands.

<table>
<thead>
<tr>
<th>MS measures to support the uptake of alternative powertrains (T&amp;E, 2016a).</th>
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<tbody>
<tr>
<td>France – has been supporting EVs since 2009. France now has the largest EV and BEV market in Europe. France has a CO2 based bonus/malus scheme for new car purchases. This is currently €6,300 (though was previously €10,000). Additional benefits are available when old diesel cars are scrapped. France has also invested in EV infrastructure, and local measures also provide further incentives.</td>
</tr>
<tr>
<td>Germany – is aiming to have 1 million EVs on the road by 2020, and offers a number of incentives for</td>
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</table>
different types of powertrain (BEV, PHEV, and FCEVs). EVs are exempt from annual circulation for the first 10 years.

Netherlands – has the highest share of EVs amongst new registrations in Europe, accounting for 8%. In 2015, the Dutch Parliament introduced a target to make 100 per cent of new car sales zero emissions by 2025. Generous subsidies for PHEVs in particular. EVs have exemptions from registration taxes, and from annual circulation taxes if emissions are less than 50g CO₂/km.

As well as national level tools, local incentives also play a role in supporting the uptake of EVs. Municipalities have an interest in supporting the uptake of EVs because they can reduce noise and air pollution levels, issues which are particularly acute in urban areas.

Tools available to local level policy markers include:

- Urban planning and the provision of infrastructure – this can include charging infrastructure, reserved parking areas, as well as the designation of low emissions zones in urban centres (e.g. Brussels)
- Public procurement and fleets – authorities have the potential to support the uptake of EVs directly by utilising alternative powertrain vehicles as part of public services (e.g. Stockholm)

Analysis of “electric vehicle capitals”, those cities with a high uptake or sales share of electric vehicles, by the ICCT aims to assess the drivers in these cities. The study covers 14 cities in Europe, North America and China, covering a third of global EV sales. It notes that cities combine typical incentives, such as subsidies and charging infrastructure, with additional local benefits. Examples include waiving tunnel tolls in Norway, use of carpool lanes in Los Angeles, and exemption from registration lotteries in Beijing and Shanghai (ICCT, 2017d).

Another study argues that in order to for the EU to reach its 2050 emissions reduction target at least half of the EU fleet will need to be fossil free by 2050. This will require EV sales to already make up 50% of sales in 2030 (considering the rate of renewal of the car fleet). It calls for an EU wide Zero Emission Vehicle (ZEV) target for 2025, or conversely a cap on ICEV sales which would be gradually lowered. Such a scheme would be comparable to the Californian model or the Chinese ZEV quota discussed previously.

Charging infrastructure

To support the electrification of motor vehicles, it will be important to ensure there is sufficient charging infrastructure in place. To achieve this will require the deployment of a range of different types of charging infrastructure to meet the varied needs of EV users.

For most use cases, it is expected that EV users will charge through privately owned infrastructure using a combination of household and workplace charging. These chargers will typically slow charging with a power of around 7kW and would take around 4 to 8 hours to fully charge a typical EV passenger car (~25kWh battery size). The expectation is that these would support a grazing type pattern of charging where the vehicles are charged slowly when vehicles are left idle either at home or at work.

The charging needs to support longer journey’s, are expected to be met through a public charging infrastructure network using a combination of slow public chargers and rapid charging stations. Slow public chargers will be in public places such as on street parking spaces and retail car parks and have a power of up to 22kW which allows for a full charge time of around 1hr. Rapid charging stations will be used on motorways to provide fast charging necessary to cover longer distances. These are being deployed with a power of 50kW allowing a typical EV to achieve an 80% recharge in 30 minutes. There is an expectation that the power of rapid charging stations will increase over time with plans already announced by a number of car OEMs to develop a charging network of 400 ‘ultra-fast’ chargers of up to 350kW (Daimler, 2016).

This approach to charging infrastructure highlights some of the challenges to adoption of EVs in that it requires a change in consumer behaviour as the BEV charging does require more consideration of time and a switch to regular topping up of battery charge which is a contrast to traditional refuelling of an ICE vehicle.

Smart charging

The rollout of EV passenger cars across Europe will lead to an increase in overall demand for electricity. In analysis for the European Climate Foundation (Harrison, 2018) scenarios with a high take up of electrified
vehicles (50% of stock by 2050) would result in an increase in EU electricity demand by just under 20% by 2050. Without any demand management, this overall increase in demand would likely occur at peak time and thus require substantial increases in electricity generation capacity to meet demand.

As such there are potential benefits to providing charging infrastructure with demand management, commonly known as smart charging. This approach looks to smooth out the demand for charging over a longer time window avoiding periods of peak demand but still ensuring that EVs are fully charged at the end of their charging window. As part of the same ECF study, they found that for Germany, unmanaged charging would increase peak demand from by 3 GW in 2030 and 22GW in 2050.

The basic form of smart charging assumes only unidirectional charging however, there are more advanced smart charging systems with bidirectional charging which are commonly known as vehicle to grid (V2G). V2G allows smart charging to actively switch between charging the EV battery and drawing charge from the EV battery to manage grid demand. This allows V2G to offer further grid management services beyond smoothing of demand offered by unidirectional charging.

By allowing battery charge from EVs to feed into the grid they can provide grid balancing services in periods of peak demand. V2G also synergises well with a decarbonising grid with increasing reliance on RES, as if EVs can be charged at times of excess RES generation this better utilises renewable capacity, and there is the potential for electricity to be fed back onto the grid when generation is low. As a results V2G could help support further rollout of RES onto the grid by reducing curtailment and enhancing grid stability.

From a consumer perspective, V2G charging could offer financial benefits to EV owners directly. This is already demonstrated through plans in UK though a collaboration of energy supplier OVO and Nissan (Ovo Energy, 2017) to offer trial V2G in home chargers for Nissan Leaf owners in 2018, with the expectation that through charging when electricity prices are low (high RES and low demand) and feeding back to the grid when prices are higher (peak demand) that EV owners could receive sufficient compensation to effectively charge their vehicle at home for free.

This is broadly in line with analysis in Harrison (2018), which expects a net benefit for residential EV users of using a V2G system. However, the analysis does make clear that the scale of the benefit is very sensitive to the price of ancillary grid services.

3.7 The socioeconomic implications

Given that reducing emissions and air pollution from transport are high on the political agenda in Europe and globally, the sector is experiencing various trends with important socio-economic implications. Most importantly, the potential positive impacts associated with the electrification of the sector would be maximized if fuelled from renewable energy sources and if access to clean technology and services are framed in an equitable manner.

The manufacturing stage is vulnerable to employment loss due to changing skill requirements. Furthermore, driving-focused sectors such as taxis and freight transportation are at risk from growing competition from sharing platforms and in the long run, autonomous vehicles. While sharing platforms may reshuffle employment, research and innovation, as well as adapted manufacturing and infrastructure development, may in turn heavily contribute to new job creation.

In terms of social implications, these trends will need to be followed with consequential educational and training support to minimize large-scale employment loss, as well as ensure that the cost of infrastructure development is not disproportionately placed on poorer segments of the population. Ensuring an equitable access to electric vehicles and new business models providing mobility as a service must be integrated in the ongoing transition to avoid deepening social and economic inequality gaps.

Nonetheless, such changes are providing valuable complimentary opportunities in public–private partnerships and contributing to the energy transition. This is particularly important as powering the sector’s electrification must be done with renewable energy in order to avoid aggravating or displacing environmental costs. Enabling the electrification of motor vehicles should be done in a way that does not shift the social, environmental and economic burdens to other segments of the population or geographies.
4 Future Policy Priorities

4.1 Research questions

- What are the links between these sectors and the Sustainable Development Goals and the different targets?
- The links between production in these sectors and consumption including consumption orientated policy tools?
- The evolution in relationships with investors and financial markets including identification of environmental issues as a risk or liability for business?
- What is the role of research and innovation in affecting the links?
- Are there some differences and good practices to be learnt from non-EU countries?
- What are the opportunities for these sectors (including jobs, growth and investment opportunities) provided by the environment and by environmental policy?
- What are the potential business evolutions (business model, product specificities, etc.) driven by environmental policies or voluntary initiatives influenced by the environment in a given sector?
- How does environmental policy affect the links between the sector and the environment?
- How does the environment and environmental policy affect the link between these sectors and growth, jobs and investment?

4.2 Links to the SDGs/EAP

Although motor vehicles are not addressed explicitly within the Sustainable Development Goals (SDGs), the development of sustainable mobility systems is an intrinsic facet of several of the goals. Relevant targets include:

- SDG 3 on health (3.6 road traffic injuries, 3.9 mortality attributed to ambient air pollution),
- SDG 7 on energy
- SDG 9 on resilient infrastructure
- SDG 11 on sustainable cities (11.2 sustainable transport systems, 11.6 air quality)
- SDG 12 on sustainable consumption and production (12.c Rationalize inefficient fossil-fuel subsidies)
Figure 4.1 Illustration of Sustainable Development Goals and Targets relevant to the motor vehicles value chain

Mapping the SDGs against the motor vehicles sector value chain

The manufacture of internal combustion motor vehicle sector relies on the environment to source raw materials for manufacturing. Growth in the market for HEVs and EVs will change the demand for specific materials necessary for their manufacture.

**+ve contribution**

12.2 By 2030, achieve the sustainable management and efficient use of natural resources

7.2 By 2030, increase substantially the share of renewable energy in the global energy mix

9.1 Develop quality, reliable, sustainable and resilient infrastructure

3.6 Halve the number of deaths/injuries from road traffic accidents

10.7 Facilitate, safe, regular and mobility of people, 11.2 safe, affordable, accessible and sustainable transport systems

9.4 Upgrade infrastructure and increase retrofit capabilities

**-ve contribution**

14.1 Prevent and significantly reduce marine pollution

12.3 Rationalise inefficient fossil-fuel subsidies

4.4 Ensure youth and adults have relevant skills and training, 9.3 increase role of SMEs

3.9 Reduce deaths and illnesses from air pollution. Goal 13 Take urgent action to combat climate change and its impacts.

3.4 Reduce by one third premature mortality from non-communicable diseases, 11.6 Reduce the adverse per capita environmental impact of cities, including by paying special attention to air quality

**Design and product development**

- Tire wear and tear remains a major driver of microplastics. No obvious upstream measure to address this issue has been identified.

- EU external costs of transport were EUR 500 billion in 2008 ~4% of GDP. Annual congestion costs EUR 146 and 243 billion

- Increasing automation in the automotive manufacture industry and driving services (e.g., haulage) present the risk of job losses and risks of SMEs if measures e.g., for vocational retraining and financing are not implemented

**Raw material supply**

**Production**

**Sales and service**

**Use-phase**

**Secondary markets**

**ELV**

A key challenge for the automotive industry will be to ensure efficient material management, particularly recycling and reusing new battery technology.
4.3 Future investment requirements

The continued growth of zero and low emissions vehicles, with some estimates reaching 50 million EVs by 2027, is dependent upon significant investments in infrastructure, battery technology and EV development (Navigant Research, 2018). The latest Bloomberg New Energy Finance Energy Outlook suggests that in Europe, EVs will account for 13% of electricity generation by 2040 and that by 2030 the growth of EVs will have pushed the cost of lithium-ion batteries down 73%. This will be particularly important in balancing the grid and capitalizing on low wholesale electricity prices. While very few estimates have been released in terms of total investment need in the sector, both the public and private sectors have increasingly pledged financial support; with global carmakers alone pledging over $90 billion towards electric vehicles so far (Reuters, 2018a). In terms of the development of charging infrastructure, some projections call for upwards of $80 billion in investments globally between now and 2025 (Smart Energy International, 2017). In the ambitious deployment of advanced powertrains modelled in Harrison (2018), by 2050 there would need to be 121 million residential charging points, 30 million workplace charging points and another 30 million public parking points across Europe. In addition, to support long distance travel, a rapid charging network of 107,000 rapid chargers would be needed. In the same scenario, total investment in charging infrastructure totals €200bn over the period 2017-2050 to meet the demands for the EU fleet.

This scale of rollout presents a considerable challenge and will likely need a combination of public and private provision to deliver. To date much of the provision in Europe has been provided by the private sector and this trend appears set to continue, with providers of conventional fuels infrastructure (BP, 2018, Shell 2017) in the UK entering the market for EV charging. However, in the early stage of charging infrastructure deployment, different OEMs operating within Europe have used different standards and technologies which limits the true coverage of the network. Inter-operability and ease of use are likely to remain a problem in such a piecemeal system.

Governments play a major role in the process of creating and securing a viable investment climate. In China, the government has pledged to invest $5 billion in new car-charging points by 2020 and aims to have one-fifth of the 35 million annual vehicle sales be alternatively-fuelled by 2025. Similarly, the Indian government has outlined plans aiming that every car being sold in India after 2030 be electric. In contrast, the regulation efforts by the European Union appear small in scale accounting for a total investment up to €800 million into new car-charging points until 2030 (WEF, 2017, Global Risk Insights, 2018). When looking at the private sector, major oil companies and private equity firms are already investing in electric vehicle developers, while equity funds are starting to look for opportunities in the infrastructure networks needed for charging the EVs (The Investor, 2018). In order to unlock multiple revenue streams for EVs, thereby boosting financial returns, some investors await the emergence of viable business models for EV batteries used as smart distributed grid storage (vehicle to grid (V2G) or vehicle-to-building (V2B)). In the meantime, investors are primarily focused on established generation technology (solar & wind) and increasingly energy storage.
4.4 **Policy recommendations**

4.4.1 **The aims of policy**

The first issue to address when setting out policy recommendations related to motor vehicles is to set out the goals, and understand how those goals might be achieved. The over-arching aim of environmental policy in this sector is clear; to reduce the environmental impact of the manufacture and operation of motor vehicles, through improvements in the way that such vehicles are manufactured and used.

The follow-on issue is then how such outcomes can be brought about. There are, broadly, three ways that reducing the environmental impacts of the manufacture and use can be achieved;

- Ensuring that production of components and vehicles is carried out in the locations where environmental damage can be most limited
- The development and deployment of incremental improvements in vehicle technology
- Wholesale technological shifts which substantially alter the manufacturing and use processes (and therefore can fundamentally shift the environmental impacts)

The first addresses the fact that, with regards to global production, some regions (such as the EU) have tighter environmental legislation than others, and therefore production carried out in these regions will typically have a smaller environmental impact than production in other locations. In terms of motor vehicles, the economics of transportation mean that most components are manufactured either within or in close proximity to the European market; the notable exception currently is batteries for use in electric vehicles, which are currently primarily produced in China and shipped into the EU.

The second and third both relate to the deployment of new technologies, but with the potential for different magnitude of environmental impacts. The second category would include the incremental technology improvements to motor vehicles that have been seen since the initial introduction of EU emissions standards, such as regenerative braking, which serve to improve the fuel efficiency of vehicles (and therefore reduce the environmental impacts of use). The third group covers more substantial technology shifts, such as the replacement of internal combustion engines with electric motors (which completely alters the environmental impacts of use, removing the combustion of fossil fuels within the vehicle), but also technologies which change how vehicles are used, such as the increased take-up of shared mobility, and (in the longer term) the deployment and take-up of autonomous vehicles.

4.4.2 **What lessons can be learnt from Norway (and from the greater demand for EVs there)?**

Norway is the country with the highest number of electric cars per capita in the world. In 2017, over 50% of new cars sold were electric or plug-in hybrids. These distinctive results can be attributed to generous tax breaks and advantageous services (i.e. free parking, charging and access to bus lanes) as well as increasing road tolls (Norsk elbilforening). While tax breaks (since 1990) have allowed electric cars to be sold at the same price as fossil fuel vehicles, lower running costs have helped sway Norwegian consumers. Moreover, up until 2017 measures such as unconditional free municipal parking and access to bus lanes also incentivised the shift to zero emission vehicles. Since 2017 local governments have been able to decide on the extent of incentives such as access to bus lanes and free municipal parking. Moreover, free toll roads are in the process of giving place to differentiated prices depending on CO2 and NOx emissions in order to avoid running into financial deficits in governmental budgets. Nonetheless, the Norwegian government’s non-binding goal that by 2025 all cars sold should be zero (electric or hydrogen) or low (plug-in hybrids) emission provide a clear and ambitious political framework that provides stability for long-term investments. In parallel to this, the government has undertaken a program to provide at least two multi-standard fast charging stations every 50 km on all main roads in Norway. While Norway’s tax breaks can be partially credited to the country’s high revenues from oil and gas production, it nonetheless exemplifies the use of a greener tax system and a well-designed policy package that is based on the polluter pays principle. The demand growth led the country to experience shortcomings in terms of supply from car manufacturers, which points to the fact that policy-makers must make sure that manufacturers can meet rapid growth (T&E, 2018b). Moreover, this example points to the need to ensure that both the electricity and battery components come from renewables and sustainable sources respectively. It can also be expected that as a
growing number of these batteries make the way onto the market, there will be a need to recycle and circulate the materials within the EU to avoid fuelling environmental degradation elsewhere.

4.4.3 How can fiscal instruments (i.e. fuel taxes, vehicle registration and circulation charges) support the decarbonisation of motor vehicles in Europe?

Market based instruments can be used to reduce the environmental impacts and dependencies of motor vehicles across their product life cycles. MBIs relevant to motor vehicles include:

- Production taxes (e.g. extraction charges for raw materials)
- In use vehicles specific taxes (i.e. registration taxes, annual circulation taxes, and company car taxation)
- In use energy specific taxes (i.e. fuel taxes)
- In use road charges (e.g. tolls, congestion charges, and parking fees)
- End of life charges and deposits

Market based instruments are one types of measure which can correct market failures, by which the costs from motor vehicles are poorly internalised. The existence of such instruments and their design can also be used to favour the manufacture or purchase of less polluting vehicles, or can lead to less polluting in-use behaviour, including an overall reduction in the distance travelled and diversion of traffic away from problematic areas. At the European level, the directive on passenger car related taxes (COM/2005/261) and the energy taxation directive (2003/96/EC) provide the main fiscal orientated instruments relevant to motor vehicles.

In practice, fiscal measures are national competences, and MS governments determine the design of fiscal instruments addressing motor vehicles. There is significant variation in practice on vehicle and transport fuel taxes between member states. The most important fiscal measures on motor vehicles are the taxes on the registration of newly purchased vehicles, circulation taxes, and fuel taxes (on petrol and diesel). Various factors determine rates in the MS.

The ACEA Tax Guide 2018 (ACEA, 2018c) provides a complete overview of the fiscal instruments currently in place in the EU-28 on motor vehicles. Analysis of 14 MS using 2018 data shows that revenues from a range of different fiscal instruments for motor vehicles varied between 2.2% of GDP in Spain and 4.9% in Belgium. Revenue generation from transport taxes (excl. fuel) varied from 0.05% GDP to 1.49% GDP across the EU-28 in 2011. When revenues from transport fuels are included, the variation is from 1.31% GDP to 3.01% GDP. Evidence suggests there is potential for reform of these taxes according to environmental criteria.

The design of specific measures as well as the combined policy mix can help to support sustainability mobility systems. Taxation policies can help to determine new motor vehicle purchases based on environmental criteria – for example vehicle emissions. New car purchases are important, as though they represent a small portion of the whole fleet in any given year they will remain in the car fleet for years and determine future impacts – for example when sold on second hand markets.

Analysis of different taxation systems in MS demonstrates that well designed fiscal measures can support a reduction in CO2 emissions. For example, in 2013 five MS reduced the annual emissions of new cars (gCO2/km) by more than 5%, whereas five MS reported reductions of less than 3%. Analysis of the fiscal instruments in place in these countries demonstrated the effectiveness of measures in reducing the CO2 emissions of the new car fleet (T&E, 2014). The best performing MS for CO2 emissions from new passenger cars are the Netherlands, Portugal and Denmark, with an average of 105.9, 104.7 and 106.1 gCO2/km respectively in 2016. As elaborated previously, evidence on the growing gap between lab and real-world test results for emissions standards of new cars presents a barrier to reducing the carbon intensity of motor vehicles during the use phase (T&E, 2017b).

133 Covering VAT on vehicle sales, services etc., fuels and lubricants, sales and registration taxes, annual ownership taxes, driving license fees, insurance taxes, tolls, custom duties and other taxes.
Increasingly EU MS have adopted vehicle taxation systems based on CO₂ emissions or fuel consumption. All MS apply VAT to new car purchases, but some apply exemptions such as those for EVs. The rates of VAT vary between countries, but a legal minimum requirement is for MS to have VAT of 15%. Most MS also apply a one-off tax on the purchase of new vehicles in the form of a registration and sales tax, as well as circulation taxes or periodical taxes. The rates for these taxes are usually determined by the characteristics of the car (weight, engine capacity, fuel consumption, CO₂ emissions).

Purchasing taxes are seen to be a key tool in reducing emissions from vehicles, providing a strong upfront signal to buyers. The Netherlands, Denmark and France, are noted for having high incentives for low emission vehicles. Recurring taxes on ownership, such as road taxes based on engine power, size or cubic capacity can also help to determine vehicle purchasing behaviour, but are potentially less effective as they occur in the future, and they also have no impact on usage.

Fuel taxes, including those on petrol, diesel and LPG, are applied in all MS. The energy tax directive provides minimum rates for fuel taxes, but there is significant variability between MS in the rates charged above the minimum rate.

Company cars represent a significant number of new car purchases – for instance in Germany 65% of all new passenger vehicles are registered to companies. Company car taxation can also be used to support a shift to low emissions vehicles. Most company car taxation is taxed as income, and few MS tax company cars on the basis of CO₂ emissions. The current design of company car taxation in most countries is based on the list price of a vehicle, however the list price doesn’t represent the financial benefit of private use of a company car. Consequently, company cars represent a subsidy to their users, and the size of the subsidy increases with the size of income or the car purchased. Poorly designed company car taxation can encourage people to purchase more polluting cars and use their cars unnecessarily.

4.4.4 Transport policy as a means to improve air quality

Much of Europe’s population are exposed to poor air quality and air pollution in different forms – representing a significant environmental and social problem. In urban areas exposure to air pollution is particularly acute – with up to 30% of Europeans exposed to air pollution levels exceeding EU air quality standards, and around 98% of Europeans in cities exposed to air quality deemed damaging by the WHO. This drives a range of impacts such as increased mortality, medical costs, loss of economic productivity etc. Economic health related impacts from air pollution are estimated at €766 billion (EC, 2017f).

Air pollutants are emitted by a range of anthropogenic activities and natural phenomena. Key sectoral contributions include transport, energy production, households, commercial activities, agriculture and waste management. Both road and non-road transport are major source sectors of air pollution of different forms. Road transport is the biggest emitter of NOₓ and the second largest emitter of BC. Other pollutants linked to transport include carbon monoxide, NMVOCs, particulate matter and sulphur dioxide.

Air pollution has been the target of a range of policy actions in the EU, including measure to address transport related pollution.

European legislation to address air pollution (including transport related measures) includes:

- Target values for ambient concentrations of pollutants (Directive 2008/50/EC)
- Limits on total emissions at the national level (Directive 2016/2284/EU)
- Transport specific measures such as the EURO 1-6 standards (Regulation 2016/2284/EU)
- Fuel quality requirements (Directive 98/70/EC)

Increasingly, initiatives have also been carried out at local or regional levels, such as low emissions zones or congestion charges to reduce pollution in urban areas. Successful reductions from transport represent the biggest share of reductions since 1990 (EEA, 2017d). However, some specific sectors in transport such as aviation and maritime shipping have increased their contributions.

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134 Although not all registered company cars are also used for private use.
4.4.5 How does road freight differ from passenger transport, what are the major challenges faced by the sector in reducing its environmental impact?

Road freight, including national and international haulage, is an important market for motor vehicles in Europe displaying some differences to passenger transport. Road freight transport increased by 4.5% between 2015 and 2016 to over 1,800 billion tonne kilometres. It has also now recovered from a decline experienced in the 2007/2008 financial crisis (Eurostat, 2017). HGVs and vans represent 75% of land freight in Europe, transporting an estimated 14 billion tonnes of goods each year. There has been no significant modal shift from motorised freight to alternative modes, such as rail transport, the share of freight held by roads has remained unchanged in the last decade (EC, 2017g).

Although HGVs represent just 4% of the road fleet, they are responsible for 30% of on road CO$_2$ emissions (ICCT, 2015). By 2030, it is forecast that HGVs will account for 40% of road transport emissions.

There has been considerable growth in some MS, notably Romania, Lithuania and Poland. In 2011, the European Commission estimated that by 2050 road freight would grow by 80%. Although the passenger car market in Europe is saturated, the road freight market is expected to grow in the future. Oil demand growth from road freight vehicles continues, and at a rate faster than other major economic sectors (IEA, 2017). Europe remains a major manufacturer of trucks and vans. In 2016 the EU exports of heavy-duty trucks generated a positive trade balance of €4.5 billion (ACEA).

A number of factors have the potential to define how road freight develops in the future and its environmental dependencies in the future. These may include but are not limited to:

- The utilisation of improvements in logistics operations so that the efficiency of networks can be improved, and vehicle utilisations can be increased
- The growing demand for on-line retail (both B2C and B2B) which can disrupt or fragment conventional supply chains (Gazzard, 2014)
- The optimisation of freight vehicle size, weight and capacity to increase efficiency (Gazzard, 2014)
- Greater adoption of readily available technology such as light weighting, aerodynamics, global positioning systems (GPS) and radio frequency identification (RFID) (IEA, 2017)
- Adoption of advanced technology such as alternative drive chains (e.g. electrification, biofuel, gas, and hydrogen), as well as automated drive technologies
- Socio-economic factors which may influence decision making for the haulage industry

Policies targeting road freight can support a reduction in environmental impacts from road freight. However, at the EU level, there are currently no legislative requirements for reducing the CO$_2$ emissions of heavy goods vehicles. Proposed amendments to the directive on clean and energy efficient road transport vehicles (COM/2017/653) outlines the limitations of the existing Directive, including the lack of incentives to move to low emission freight vehicles.

One US study suggested that while fuel prices were low, there was little incentive for HGV operators and drivers to invest in new technology (Boriboonsomsin, 2015).

Other than carbon emissions, the EURO standard (IV) for tailpipe emissions has successfully reduced the levels of pollutants emissions from heavy duty vehicles – notably NO$_x$ and particulate matter.

The haulage industry itself, which relies on motor vehicles for the provision of a service, will likely be impacted by changes to the sector. The adoption of driverless vehicles for road freight in the future will likely reduce the demand for drivers. In OECD countries, drivers represent 6.4 million jobs, of which up to 4.4 million jobs could lost if there is widespread deployment of automated technology. Research by the ITF suggests that drivers would be particularly vulnerable to extended unemployment as a result of low levels of education and automation in other sectors (ITF, 2017). SMEs, make up around 85% of the EU haulage market – usually with ten or fewer trucks, and face several barriers to the uptake adopting more fuel-efficient vehicles, including (T&E, 2016b):

- More fuel-efficient vehicles are often more expensive, with limited access to finance for SMEs to invest in improving their fleet
- Inadequate information is available to SMEs on the right vehicles to use, with brand loyalty rather than fuel efficiency often determining purchases.
- Incentives are split between trailer and carrier owners which are often separate from each other. E.g. fuel costs may be covered by contractors, or carriers are unable to invest in improvements to trailers.

In general, the uptake of advanced power trains for HGVs is expected to be slower than for light weight vehicles. This could be the result of the physical and economic differences between the two types of transport – i.e. more expensive, heavier vehicles which cover longer distances.

In the short to medium term, the deployment of fuel efficiency technologies to conventional ICE powertrains will need to play a substantial role in ensuring HGV reach the proposed targets of reducing emissions in new vehicles by 15% by 2025 and 30% by 2030, compared to 2019 levels. Data on available fuel-efficient technologies for heavy duty vehicles from the ICCT (2017e) shows potential fuel efficiency improvement of 27% in the short to medium term and as much as 43% by 2030.

As with passenger vehicles, reducing emissions from road freight will also rely on developments to road infrastructure. To support electrification of HGVs, will require a greater reliance on public infrastructure either a rapid charging network (as proposed in the announcement of the Tesla Semi (Reuters, 2018b)) or an electric road system (such as the catenary road system currently trialled by Siemens in Sweden (Siemens, 2016)) or hydrogen refuelling stations.

### 4.4.6 Summary of recommendations

As outlined at the start of this chapter, reductions in the environmental impact of motor vehicles can be achieved through shifting the location of production or the deployment of new technologies, whether incremental or transformative. In the near term, the largest gains in terms of environmental impact that can be realised in the sector relate to shifting demand away from internal combustion engine vehicles and towards electric vehicles, and ensuring that the new components in electric vehicles (most notably the battery) are produced.

To that end, the analysis has highlighted a number of ways in which greater deployment of EVs could be achieved, in order to reduce the environmental impact of the sector, and how the environmental impact of EVs could be minimised:

- Market-based instruments to reduce the cost of e-mobility options, such as the tax breaks on purchases provided in Norway.
- Stronger signalling through a wider range of rates applied for in-use taxation (e.g. reflecting CO₂ emissions from use).
- A continued tightening of the supply-side measures, primarily CO₂ and EURO standards, to reduce GHG and particulate emissions.
- Policy to align incentives in the HGV market, including improved access to finance for small operators.
- Continued measures, such as the European Battery Alliance, to onshore the production of EV batteries.
4.5 Impacts of Future Policy

4.5.1 The environmental impact of battery manufacture in the EU vs China

Understanding the environmental footprint of the sector, and therefore how the transition to e-mobility impacts on the aims of the SDGs and the EAP, requires an understanding of the environmental impact of new components. Most notably, in the context of a shift to EVs, there is a substantial environmental impact associated with the manufacture of the batteries used in these vehicles. Currently, battery cell manufacture takes place in China; however, as the take-up of EVs in Europe increases, and policy initiatives such as the European Battery Alliance gain traction, it is expected that production of cells will switch to Europe. A key question then is the impact that this switch will have on the environmental footprint of EVs and e-mobility.

In order to assess the environmental impacts of automobile (traction) battery production between EU and China, an LCA study using a detailed battery model was conducted. The scope of the study was limited to the production phase of one battery pack, which served as functional unit.

*Figure 4.2 Flow diagram of the traction battery pack model used for this study*

Source: Ellingsen et al. (2013).

Note(s): BSM=Battery Management System; BMB=Battery Management Board; IBIS=Integrated Battery Interface System

For compiling the life-cycle inventory of a battery pack, Ellingsen et al. (2013) collected primary data on production inputs and outputs of a lithium-ion nickel-cobalt-manganese traction battery was used. This data was sourced from a Norwegian battery manufacturer (Miljøbil Grenland). For the sake of meaningful comparison, it was assumed that the battery inventory does not differ between the EU and China.

The Ecoinvent v3.4 database was used for compiling life-cycle inventories. As Ecoinvent does not offer EU and Chinese datasets for all required life-cycle inputs, a hierarchy of geographic specification was used in order to ensure at least broad representativeness.

Table 4.1 gives the geographies used, which indicates how accurate the compiled inventory is for representing the two regions under comparison – 68% of datasets for the Chinese inventory are for the “Rest of World” geography, however the “China” and “Rest of Asia” markets cover most important inventory components such as mining and electricity use. Thus, while the inventory is not entirely specific to the region of China, it gives a general impression of the impacts of a battery pack produced in Asia.

Finally, due to lack of representative datasets, use of secondary aluminium and copper is modelled as inputs of primary metals. As this is done for both EU and China, no large deviations should result.
Table 4.1 Geographic breakdown of Ecoinvent v3.4 market datasets used for life-cycle inventory compilation

<table>
<thead>
<tr>
<th>EU market datasets used</th>
<th>China market datasets used</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rest-of-Europe &amp; Europe without Switzerland</td>
<td>86%</td>
</tr>
<tr>
<td>Rest of World</td>
<td>1%</td>
</tr>
<tr>
<td>Global</td>
<td>6%</td>
</tr>
<tr>
<td>IAI Area, EU27 &amp; EFTA</td>
<td>4%</td>
</tr>
<tr>
<td>Other</td>
<td>3%</td>
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<tr>
<td>Total = 259 datasets</td>
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</tbody>
</table>

Note(s): “Other” for EU includes data from individual countries such as SE, DE and US, where no other regionally-representative dataset exists. “Global datasets used are identical for EU and China — i.e. for these inventory components, there are no other Ecoinvent datasets available. A “dataset” is the smallest element considered in the life cycle inventory analysis for which input and output data are quantified, as per ISO 14040.

Ultimately, in the Interpretation phase, conclusions are drawn from the inventory analysis and the environmental impact assessment. On that basis, recommendations for decision-makers can reasonably be made, taking into due account data limitations.

Note(s): 100% = equal impacts; less than 100% = Chinese battery pack has higher impacts. Difference from 100% = how much more/less impactful the EU battery pack is. In general, differences of less than 10% are likely not significant, given the uncertainty of the Ecoinvent data. “CTU” is a chemical toxicity unit (similar to a CO2-equivalent); Sb is the element antimony.

Figure 4.3 gives the impact categories used and demonstrates that across most indicators, battery packs produced in Europe exhibit a lower environmental impact compared to China. In particular, the impact on human health toxicity, as well as water pollution in Europe, is less than one-third that of China. In addition, the climate change impacts in Europe are approximately half that of China. The same is also true in terms of air pollution. There are certain impact categories where Europe does not fare as favourably compared to China. Overall traction battery production in Europe requires only marginally less water and land use and involves as much resource use impacts. However, resource use impacts need to be interpreted with caution. The largest component in the resource use category would be the Battery Management System, which has multiple electronic (and thus rare-earth element) inputs. All such electronic inputs are only available as “Global” datasets in Ecoinvent. Thus, the available data does not allow a comparison between EU and China.
The model detail allows examining each individual battery component separately. This can help establish whether certain components can be produced in China without adding any significant net environmental burdens compared to European production.

*Figure 4.4 Ratio of impact assessment results for a battery pack produced in the EU vs China, broken down by battery pack component*

Note(s): 100% = equal impacts; less than 100% = Chinese battery pack has higher impacts. Difference from 100% = how much more/less impactful the EU battery pack is. In general, differences of less than 10% are likely not significant, given the uncertainty of the Ecoinvent data.

The cooling system exhibits the most significant fluctuation among examined impact categories. Whereas land use, air pollution and climate change impacts are distinctly more favourable in Europe, most other environmental impacts are more or less the same in both examined regions. Most notably, the production of cooling systems in China seems to require less water than in Europe, primarily due to lower water consumption in the aluminium production process.

Conversely, battery management system (BMS) production is relatively consistent among all examined impact categories. Further, it also displays approximately the same environmental impact for production in both Europe and China. It is absolutely critical to bear in mind that these results can largely stem from the fact that Ecoinvent’s database does not offer regionalised datasets for electronics components, which are over 50% of the BMS by mass, and can be expected to contribute strongly to impacts due to inputs of rare-earth elements.

Next, packaging production demonstrates tangibly lower environmental impacts for Europe in six out of eight categories. Resource and water use impacts, on the other hand, are essentially the same in Europe and China.

Last but certainly not least, the environmental impacts from battery cell production in Europe are also substantially lower than those in China. In five out of eight categories the European environmental impact is barely half of the Chinese. What is more, in three out of those five categories the European impact is between 12% and 24% that of the Chinese. In other words, producing battery cells in Europe leads to much lower impacts in terms of human toxicity and water pollution, as well as reasonably lower impacts on air pollution and climate change.

This finding is significant, insofar as battery cells comprise almost two-thirds of the battery pack by weight. As a result, they account for the better part of the overall environmental burdens caused by traction battery production in both Europe and China. As demonstrated in the table below, the only major exception here is in the resource use category, where the production of BMS displays higher environmental impacts,
compared to the battery cell. The EU battery cell accounts for a relatively lower share of impacts compared
to the Chinese cell, owing to less impacts of EU electricity usage and mining.

Figure 4.5 Breakdown of impact category contribution of different battery pack components

<table>
<thead>
<tr>
<th></th>
<th>One battery pack</th>
<th>Battery cell</th>
<th>Packaging</th>
<th>BMS</th>
<th>Cooling system</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate Change</td>
<td>3.84E+03</td>
<td>2.80E+03</td>
<td>6.27E+02</td>
<td>2.95E+02</td>
<td>9.80E+01</td>
</tr>
<tr>
<td>Water use (m3)</td>
<td>9.50E+01</td>
<td>8.18E+01</td>
<td>8.70E+00</td>
<td>3.14E+00</td>
<td>1.27E+00</td>
</tr>
<tr>
<td>Water pollution - ecosystems (kgP-Eq)</td>
<td>3.65E+00</td>
<td>2.51E+00</td>
<td>3.40E-01</td>
<td>7.50E-01</td>
<td>5.00E-02</td>
</tr>
<tr>
<td>Water pollution - human health (CTUh.m3.y)</td>
<td>1.11E+05</td>
<td>7.26E+04</td>
<td>9.97E+03</td>
<td>2.69E+04</td>
<td>1.51E+03</td>
</tr>
<tr>
<td>Air pollution (kgPM2.5-Eq)</td>
<td>5.61E+00</td>
<td>4.53E+00</td>
<td>4.30E-01</td>
<td>5.70E-01</td>
<td>6.00E-02</td>
</tr>
<tr>
<td>Land use (kgSoilO.)</td>
<td>7.02E+03</td>
<td>5.12E+03</td>
<td>6.64E+02</td>
<td>1.06E+03</td>
<td>6.69E+01</td>
</tr>
<tr>
<td>Resource use (kgSb-Eq)</td>
<td>1.18E+00</td>
<td>5.50E-01</td>
<td>3.00E-02</td>
<td>1.41E+00</td>
<td>1.68E-03</td>
</tr>
<tr>
<td>Human health toxicity (CTUh)</td>
<td>3.94E-03</td>
<td>2.17E-03</td>
<td>4.58E-04</td>
<td>1.25E-03</td>
<td>6.62E-05</td>
</tr>
</tbody>
</table>

Note(s): Results for 1 functional unit (battery pack), being a summation of each row. In general, differences of less than 10% are likely not significant, given the uncertainty of the Ecoinvent data.

In summary, the results generally give credence to traction battery manufacturing in the EU as opposed to China, at least insofar as environmental impacts are concerned. The battery cell is by far the largest contributor to environmental impacts for all impacts apart from resource use. For resource use, the lack of geographical differentiation in Ecoinvent does not allow a true comparison between EU and China, given that electronics are a significant user of rare-earth elements. Water use is the only category where the EU performs worse than China, specifically to do with water use in aluminium production. Whether this is a true result or a quirk of the Ecoinvent data requires more detailed analysis. In any case, worse performance in a single impact category does not strongly change the conclusions drawn.

This suggests that moving battery cell production from China to the EU would have beneficial environmental impacts, and therefore should be considered as a potential area for further policy action.

4.5.2 Modelling the impacts of taxes on the take-up of electric vehicles

Understanding the implications of individual policy measures on the take-up of different technology options requires a substantially different approach to modelling compared to that encompassed in the macroeconomic model E3ME. The decision-making process is made ‘bottom-up’, i.e. individual consumers make technology choices based upon a number of factors, including (but not limited to) the price of competing options. In European Commission Impact Assessments, this modelling is carried out using the TREMOVE model\textsuperscript{135}. Modelling such technology choices ‘top down’, based upon simple price elasticities (e.g. the change in the use of ICEs from a change in fuel duty) will give only limited insight, as it is not able to explicitly consider the relative costs of alternative technologies and how such costs evolve over time (and with deployment).

E3ME does include a bottom-up technology module related to passenger cars; FTT:Transport. While this has been applied in academic literature (see for example, Mercure et al, 2018), it has not been used in the context of formal European Commission modelling. The FTT:Transport model is not calibrated to match TREMOVE, and has a different underlying theoretical framework; it is therefore likely that the two models would show substantially different impacts from the same policy introduced into the two frameworks. On this basis, no formal modelling has been done in this study to assess the responsiveness of take-up of EVs to

\textsuperscript{135} See https://www.tmleuven.be/en/navigation/TREMOVE for more details.
changes in taxation policy, and it is our recommendation that such analysis should be undertaken in TREMOVE, to ensure consistency with the European Commission’s approach to transport modelling in other research.

Instead, in the analysis that follows, changes in the deployment of fuel-efficient technologies and advanced powertrains (focusing on EVs) is introduced exogenously into the available modelling tools. Such an approach does not allow modelling of the efficacy of specific policies, but does allow an exploration of the socioeconomic and environmental impacts of achieving such an accelerated shift. This includes the impact of the transition on tax revenues (e.g. how total fuel tax revenues are altered by a shift towards EVs).

4.5.2 Modelling the environmental and economic impacts of more efficient vehicles and the transition to advanced powertrains

To explore the impact of the introduction of more fuel-efficient vehicles and the deployment of advanced powertrains into the European fleet, we model the outcomes of a number of potential uptake scenarios through new sales.

- **Reference (REF)** – A baseline where there are no further introductions of advanced powertrains of fuel-efficient technologies from 2017 onwards
- **Fuel efficient technology (TECH)** – No advanced powertrains are introduced after 2017 but fuel-efficient technologies are introduced.
- **Electric vehicle (EV)** – Advanced powertrains are rapidly introduced in the short to medium term alongside fuel efficient technologies with a phase out of ICEs by 2035

To model the impacts of these uptake scenarios requires a modelling framework which uses two models, CE’s vehicle stock modelling tool (VSM) and CE’s macroeconomic model (E3ME), to model the economic impact.

The vehicle stock model calculates vehicle fuel demand, vehicle emissions and vehicle prices for a given mix of vehicle technologies. The model uses information about the efficiency of new vehicles and vehicle survival rates to assess how changes in new vehicles sales defined in the uptake scenarios, affect stock characteristics. The model also includes a detailed technology sub-model to calculate how the efficiency and price of new vehicles are affected, with increasing uptake of fuel-efficient technologies. For the detailed results of the vehicle stock modelling, see appendix B.

The outputs of the VSM are then used as inputs to E3ME, an integrated macro-econometric model, which has full representation of the linkages between the energy system, environment and economy at a global level. The high regional and sectoral disaggregation (including explicit coverage of every EU Member State) allows modelling of scenarios specific to Europe and detailed analysis of sectors and trade relationships in key supply chains (for the automotive and petroleum refining industries). E3ME was used to assess how the transition to low carbon vehicles affects household incomes, trade in oil and petroleum, consumption, GDP & employment.

For this modelling, we have chosen to model these scenarios for Germany only. Germany is a large EU member state, where motor vehicle production plays a large role in the economy. It has extensive and robust data available on sales and the characteristics of the stock.

4.5.3 Economic modelling results
Over the long term, both scenarios show positive impacts on GDP relative to the reference scenario. Figure 4.6 shows the relative changes out to 2050. The main driver of the increase in GDP is the reduction of fossil fuel demand in the scenarios. As Germany is dependent on imports of crude oil and manufactured fuel to meet the demands for road transport, a reduction in demand for oil in the German economy leads to an improvement in the balance of trade and so leads to an increase in spending on good and services on goods with a higher domestic content.

Figure 4.6 Impact of decarbonisation scenarios on GDP relative to Reference scenario

This is partially offset by high prices for motor vehicles, which raises the costs for consumers and diverts spending away from other parts of the economy and towards the manufacture of motor vehicles. In the case of Germany, as the economy has a strong position in the manufacture of motor vehicles, this diverting of spending would still be to a mostly domestic sector. For other member states which do not have a large vehicle production sector this would have a larger negative impact. However, we see in the short term it has a noticeable negative impact on GDP as the increase in the cost of motor vehicles lead ahead of fuel savings over the lifetime of the vehicle.

Even then, we would expect the reduction in oil imports to dominate as over the medium to long term, as the reduction in the lifetime cost of owning and refuelling a vehicle with fuel efficient technologies or an advanced powertrain outweighs the initial cost of the technology.

Finally, the investment in charging infrastructure boosts investment in the electricity sector which, again due to a high volume of domestic content, boosts domestic output. As mentioned in the infrastructure section above, this infrastructure is paid for through higher prices which offsets some of the positive economic impact.

In explaining the relative impacts of the scenarios, we can also see the dominance of the oil import effect drives this as by 2050 the reduction in oil in TECH is around half that of EV and overall the GDP impact of the TECH scenario is slightly less than twice the impact of EV by 2050. In the short to medium term out to 2030, the impact is broadly similar, as the difference in oil imports between the scenarios is small enough to be offset by the slightly higher vehicle costs for advanced powertrains.

Table 4.2 Economic and environmental impacts for different vehicle uptake scenarios, in Germany in 2050
### Battery Assumptions

<table>
<thead>
<tr>
<th></th>
<th>50% of cells imported</th>
<th>All cells domestically produced</th>
<th>All cells imported</th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP (%)</td>
<td>0.4%</td>
<td>0.8%</td>
<td>1.3%</td>
</tr>
<tr>
<td>Employment ('000s)</td>
<td>132</td>
<td>229</td>
<td>342</td>
</tr>
<tr>
<td>Investment (%)</td>
<td>0.5%</td>
<td>1.1%</td>
<td>1.4%</td>
</tr>
<tr>
<td>Tailpipe CO2 emissions from passenger cars (MTCO₂)</td>
<td>-47</td>
<td>-99</td>
<td>-99</td>
</tr>
<tr>
<td>Fossil Fuel demand from passenger cars (Mtoe)</td>
<td>-699</td>
<td>-1459</td>
<td>-1459</td>
</tr>
<tr>
<td>Electricity demand from passenger cars (Mtoe)</td>
<td>0</td>
<td>244</td>
<td>244</td>
</tr>
</tbody>
</table>

**Notes(s):** All values represented as difference from baseline REF scenario

*Figure 4.7 Employment impact by sector in the EV scenario*

In terms of employment, we see a net increase in jobs in both scenarios as with the overall economic impact on GDP. However, underneath the aggregate impact the benefits of the transition are not felt equally across all sectors of the economy as shown for the EV scenario in Figure 4.7. The largest increase is in the service sectors of the economy, fuelled by additional consumer expenditure on goods and services other than fossil fuels. Manufactured fuels show a decline in jobs from lower fossil fuel demand although this is quite small due to the low employment intensity in refining. Similarly, the electricity sector sees a modest increase in jobs from the additional electricity demand for electric vehicles.

The shift in motor vehicles and electrical equipment reflect the changes in the automotive supply chain due to electrification, as production moves away from internal combustion engines sourced from the motor vehicle sector to batteries and electric motors sourced from electrical equipment sector. Overall the net impact on the automotive supply chain is positive reflecting the higher value of advanced powertrains.
4.5.4 The impact on tax revenues

In Germany, as with most European countries, there is a tax on fossil fuels to pay for road maintenance or to the general pool of taxation. With a transition to fuel efficient vehicles and advanced powertrains reducing consumption of fossil fuels, the government sees a loss in taxation from fuel use.

From our modelling, we see that the losses in fuel duty have a very small impact on overall government tax revenues with a reduction of €30bn in the EV scenario in 2050 relative to reference out of a total tax revenue of €1.9 trillion. However, due to the overall increase in economic activity as described above, we see an increase in revenue from other taxes (such as income tax) that mitigates the lost fuel duty entirely.

However, that is not to say that policy makers should not look to replace fuel duty with an alternative tax on road use as policies such as road pricing could help to ensure drivers are paying the true cost of road use, including negative externalities such as congestion and noise pollution.

4.5.5 The impact of the location of battery manufacturing

The results of the scenario are sensitive to a number of assumptions about the transition to electrified vehicles. One of largest considerations is the location of battery cell manufacture. Currently battery cell manufacture is largely situated outside of Europe, but it is expected as the production of electric vehicles expands that such activity will start to take place within Europe. Although it is unclear exactly where in Europe this will take place, in this analysis, we have taken a simple assumption in the central case that Germany imports 50% of the battery cells it needs.

To explore the potential range of impacts, we have also explored scenarios variants where Germany produces all battery cells domestically or imports all battery cells. In Figure 4.8, we show the GDP impact of the scenarios under these different assumptions of battery production.

From the modelling, we see that the location of battery cell production could have a substantial impact on overall economic effect of a shift to advanced powertrains. However, even in the most pessimistic case where all battery cells are imported which would have a negative impact on the German trade balance, the overall impact of the transition is positive for the German economy.

4.5.6 Environmental impacts of the transition towards EVs
The transition to EVs has the potential to substantially reduce the in-use emissions associated with motor vehicle use. CO2 emissions are 33% lower in the EV scenario compared to Reference by 2030 (73MTCO2 vs 109 MTCO2 in the reference), and by 2050 CO2 emissions are 93% lower in the scenario, at only 7MTCO2 (compared to 106MTCO2 in the reference) (see Figure 4.9).

Figure 4.9 MT CO2 emitted by the German motor vehicle fleet

Other emissions related to the operation of internal combustion engine vehicles include nitrous oxides (NOx) and particulate matter (primarily PM10s and PM2.5s). While there is a link between rates of CO2 and other emission releases, historically separate technologies have been used to reduce non-CO2 emissions, with reductions legislated through the EURO emissions standards. As such, the modelling presented here assumes that CO2 and other emissions evolve independently of one another, and reductions in NOx and PM10s are associated only with a shift away from ICEs, rather than being explicitly linked to CO2 emissions standards. In practice, this would be achieved through the removal of existing technology designed to limit these pollutants as fuel efficiency improved (i.e. as NOx emissions fell due to improved fuel efficiency, aftertreatment technologies would be removed from the vehicle to reduce manufacturing costs).

The result is that reductions in NOx and PM10s occurs more slowly than the reduction in CO2, as it is linked only to the deployment of EVs (with no impact from the improved fuel efficiency of new ICE vehicles). By 2030, NOx emissions are 7% lower than in the reference case, while PM10s are reduced by 9%. However, by 2050, as many more EVs are deployed into the vehicle fleet, emissions of these pollutants are reduced substantially; NOx is reduced by 88%, to less than 8MT, while PM10 emissions are reduced by 89% to almost zero (see Figure 4.10).

Note also that both NOx and PM10 emissions fall substantially in the reference case; this is due to the fact that all new vehicles sold conform to the latest Euro 6 standards, and these vehicles enter the stock to replace older vehicles which conformed to earlier, less restrictive, Euro standards.
4.5.7 Limitations to the modelling

In the modelling, it is important to consider the potential impacts that have not been considered. Most significant is the potential rebound effect of a lower cost of driving. As electrification of passenger cars reduces the overall cost of driving, this may incentivise people to drive further or more frequent or shift away from public transport to private car use. This would lead to reduction in the environmental and economic benefits of the transition outlined, and would exacerbate other negative externalities such as congestion and noise pollution. As a result, the existence of this rebound effect though not quantified still suggests a clear role for policy to ensure that the true costs of driving are paid by the road user most likely through a new form taxation of road use.

Secondly, beyond the battery sensitivities, we do not consider the impact of the transition from traditional motor vehicles to advanced powertrains on the trade in motor vehicles as we assume that countries maintain their current market share and transition to new market conditions at the same rate. However, clearly this may not be the case.

Finally, on autonomy and shared ownership, as outlined in the literature review, there is a large amount of uncertainty around the net impact of these developments on the environmental and economic impacts. For example, with autonomy, there could be a reduction in energy use if autonomy leads to more efficient driving and less congestion, however, this could all be offset by a rise in demand for private car use as the convenience of autonomy, leads to a lower cost of driving and leads to a net increase in private car use and thus higher fuel demand.

The impacts as assessed for Germany will not entirely reflect the impacts of the rest of the EU and would be expected to represent the ‘top end’ of potential impacts relative to the rest of the EU, where the manufacture of motor vehicles is not such a large part of the economy. However, regardless of the magnitude, the key drivers of the economic results hold true for the EU28 as much as for Germany. The clearest example is with oil imports, which all member states are reliant on to fuel their passenger car fleet and it is reduction of imported oil which drives the economic impacts. This result is borne out in Harrison (2018) where impacts of decarbonising transport across the EU, we see smaller scale impacts for the EU28 with a similar decarbonisation scenario delivering a GDP impact of +0.5% by 2050 while reducing tailpipe CO2 emissions from passenger cars by around 90%.
Appendix A LCA methodology

A.1 Goal and scope definition

The goal of the study was to assess the difference in impacts in the life-cycle of traction battery packs manufactured in the EU vs such manufactured in China. The scope is cradle-to-gate production of a Nickel Manganese Cobalt Oxide (NCM) traction battery and its related background processes (transport, infrastructure). Other electric vehicle components are excluded. The functional unit is the production of one battery pack, with its composition by mass given in Table A.1, following Ellingsen et al. (2013) who provide this based on primary data on production inputs and outputs from the Norwegian battery manufacturer Miljøbil Grenland.

According to Peters et al. (2017) who undertake a review of Li-Ion LCA studies, the Ellingsen inventory is amongst the few battery LCA studies which are more recent and provide a detailed product inventory based on primary data. In order to ensure representativeness, it is assumed that the battery pack modelled has the same composition in both EU and China.

Table A.1 Modelled battery pack composition by mass. Indents denote subcomponents, bolded are the main parts of the battery pack. Taken as per Ellingsen et al. (2013).

<table>
<thead>
<tr>
<th>Component</th>
<th>Mass [kg]</th>
<th>% mass</th>
</tr>
</thead>
<tbody>
<tr>
<td>One battery pack</td>
<td>250.00</td>
<td></td>
</tr>
<tr>
<td>Battery cell</td>
<td>150.00</td>
<td>60%</td>
</tr>
<tr>
<td>Anode</td>
<td>59.00</td>
<td></td>
</tr>
<tr>
<td>Negative current collector Cu</td>
<td>34.00</td>
<td></td>
</tr>
<tr>
<td>Negative electrode paste</td>
<td>25.00</td>
<td></td>
</tr>
<tr>
<td>Battery grade graphite</td>
<td>24.00</td>
<td></td>
</tr>
<tr>
<td>Cathode</td>
<td>65.00</td>
<td></td>
</tr>
<tr>
<td>Positive current collector Al</td>
<td>7.50</td>
<td></td>
</tr>
<tr>
<td>Positive electrode paste</td>
<td>58.00</td>
<td></td>
</tr>
<tr>
<td>Positive active material</td>
<td>54.00</td>
<td></td>
</tr>
<tr>
<td>(NiCoMn) hydroxide</td>
<td>52.00</td>
<td></td>
</tr>
<tr>
<td>Nickel Sulphate</td>
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<td></td>
</tr>
<tr>
<td>Cobalt Sulphate</td>
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</tr>
<tr>
<td>Manganese Sulphate</td>
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<td></td>
</tr>
<tr>
<td>Electrolyte</td>
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</tr>
<tr>
<td>Lithium hexafluorophosphate</td>
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<tr>
<td>Separator</td>
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<td>Cell container</td>
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<td></td>
</tr>
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<td>Tab, aluminium</td>
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<tr>
<td>Tab, copper</td>
<td>0.39</td>
<td></td>
</tr>
<tr>
<td>Multilayer pouch</td>
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<td></td>
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<td><strong>Packaging</strong></td>
<td>81.00</td>
<td>32%</td>
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<tr>
<td>Module packaging</td>
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<tr>
<td>Module fasteners</td>
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<tr>
<td>Outer frame</td>
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<td></td>
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<tr>
<td>Inner frame</td>
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<tr>
<td>Bimetallic busbars</td>
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<td></td>
</tr>
<tr>
<td>End-busbar, Al</td>
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<td></td>
</tr>
<tr>
<td>End-busbar, Cu</td>
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<td></td>
</tr>
<tr>
<td>Module lid</td>
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<td></td>
</tr>
<tr>
<td>Battery retention</td>
<td>9.00</td>
<td></td>
</tr>
<tr>
<td>Strap retention</td>
<td>0.78</td>
<td></td>
</tr>
<tr>
<td>Lower retention</td>
<td>3.10</td>
<td></td>
</tr>
<tr>
<td>Propagation plate</td>
<td>4.10</td>
<td></td>
</tr>
<tr>
<td>Battery tray</td>
<td>24.00</td>
<td></td>
</tr>
<tr>
<td>Tray w fasteners</td>
<td>19.00</td>
<td></td>
</tr>
<tr>
<td>Tray lid</td>
<td>5.00</td>
<td></td>
</tr>
<tr>
<td>Tray seal</td>
<td>0.01</td>
<td></td>
</tr>
<tr>
<td><strong>Battery Management System</strong></td>
<td>9.40</td>
<td>4%</td>
</tr>
<tr>
<td>IBIS</td>
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</tr>
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<td>IBIS fasteners</td>
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<td></td>
</tr>
<tr>
<td>Low Voltage system</td>
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<td></td>
</tr>
<tr>
<td><strong>Cooling system</strong></td>
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<td>4%</td>
</tr>
<tr>
<td>Radiator</td>
<td>9.10</td>
<td></td>
</tr>
<tr>
<td>Manifolds</td>
<td>0.40</td>
<td></td>
</tr>
<tr>
<td>Clamps &amp; fasteners</td>
<td>0.24</td>
<td></td>
</tr>
</tbody>
</table>
A.2 Life-cycle inventory compilation

The life-cycle inventory for the entire product system studied is modelled via Ecoinvent v3.4 (Wernet et al., 2016) and with the consequential system model, as is generally recommended for studies with decision support in mind (Ekvall and Weidema, 2004). In brief, consequential modelling deals with the consequences of changes in demand for products from unconstrained suppliers (such that can respond to changes in demand, i.e. those that can be expected to change). Under this system model, products receive the full burdens of their inputs and emissions but also receive “credits (impacts forgone i.e. subtracted) for any by-products produced that can substitute primary inputs (e.g. waste heat for electricity generation). For a fuller discussion on consequential modelling, refer to Ekvall and Weidema (2004) and Wernet et al. (2016).

Ecoinvent differentiates datasets based on geography (e.g. aluminium production in different countries). The goal of the study was to model battery packs produced in EU vs China but Ecoinvent does not provide all datasets required in both geographies. Thus, we have utilised a hierarchy of Ecoinvent geographies, with the aim of maximal representativeness.

For the EU this is:
Rest-of-Europe or Europe without Switzerland → IAI Area, EU27 & EFTA → Rest of World → Global → “Other” (individual countries where no other regionally representative dataset exists)

For China this is:
China → Rest of Asia → Rest of World → total nr of observations

In general, EU modelling is well-representative, while for China this is less so. However, modelling of China is covered by “China” and “Rest of Asia” datasets for most important inventory components (e.g. mining, electricity use), so the inventory compiled is broadly representative of the life-cycle of a battery pack produced in Asia.

Finally, inputs of secondary aluminium and copper are modelled as inputs of primary metals due to lack of representative datasets. This is done for both EU and China in order to avoid any deviations.

A key limitation of the approach is the modelling of the electronic components of the battery management system (BMS), for which only a single “Global” dataset existed, based on which no difference can be modelled between EU and China. While the BMS is not very large in terms of mass, it has the 2nd GHG intensity after the battery cell according to the source study (Ellingsen et al., 2013). We can expect that this underestimates resource use impacts as well as BMS electronics are the main user of rare-earth minerals in the battery pack.

A.3 Life-cycle impact assessment

For life-cycle impacts modelling, we select a subset of indicators covering all 4 main categories of the ILCD 1.0.8 2016 midpoint method. The International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC, 2011) recommends several models for life-cycle impact assessment which is why the method has been preferred for this study. The impact categories selected are:

- Climate change – GWP100 (kgCO2-Eq)
- Ecosystem quality – Freshwater ecotoxicity (CTUh.m3.yr) and eutrophication (kgP-Eq)
- Human health - Respiratory effects, sum of organics and inorganics (kgPM2.5-Eq)
- Resource use – land use (kg Soil organic carbon) and minerals, fossils & renewables (kg antimony-Eq)

---

136 “Dataset” in this context follows the ISO 14040 definition of a “unit process” – the smallest element considered in the life cycle inventory analysis for which input and output data are quantified.
• We add an extra 5th category "Water use", being a summation of water use in the life-cycle inventory (m3)

While many LCA studies exist comparing different vehicle technologies, their comparison is difficult and shows wide divergence. Nordelöf et al. (2014) review 79 vehicle LCA studies, which can be divided into two main types – well-to-wheel (WTW) studies (covering just the life-cycle of the fuel, including its use phase) or “full” LCAs (which also include the production of the vehicle itself).

A.3.1 LCA evidence from well-to-wheel
Reviewing WTW studies finds important determinants of GHG performance to be electricity production, degree of vehicle electrification and driving modes. Comparing BEVs powered with different electricity mixes modelled after a large EU study, grid GHG intensities above 900 g CO2-eq/kWh (roughly that of oil-fired power production in the study) lead to BEVs emitting more than the reference average EU vehicle. The study shows that a decarbonized grid leads to substantially lower BEV WTW emissions. For degree of vehicle electrification, BEVs, PHEV (plug-in hybrids) and E-REVs (extended range EVs) are compared to a reference EU vehicle, with petrol as a liquid fuel and an EU-average grid mix for electricity consumption. Given an EU-average electricity mix and compared to the reference EU vehicle’s 143 gCO2-eq/km intensity, all electric vehicles perform better across the range of uncertainty in the results, with BEVs performing best with a maximum 76 gCO2-eq/km intensity, while PHEVs maximum intensity is worst at 126 gCO2-eq/km.

With regards to driving modes, the study cites Raykin et al. (2012) who compare the impacts of different traffic conditions for HEV, PHEV and ICE (internal combustion engine) large family cars under scenarios of city driving (low speeds, many stops, high congestion), suburban driving (less congestion and higher speeds) and highway driving (no stops, high speeds). The Nordelöf review further includes BEVs (representing small family cars) in the comparison (apart from in the highways scenario). Electric vehicles show clear advantages for city driving where the benefits of regenerative breaking can be maximized and due to the fact that electric vehicle engines turn off when idling. This is most pronounced for BEVs which are found to be 2x less GHG intensive per km than the PHEVs (the next best option) and 6x less intensive than petrol ICEs, this difference further increasing in a suburban scenario where PHEVs become more GHG intensive due to more reliance on liquid fuels. However, in a highway driving scenario (with BEVs not included due to infeasibility), the best-performing electric option (PHEVs) is only 13% less GHG intensive per km than an ICE engine.

It is worth noting that a large amount of WTW studies are based on laboratory-tested fuel consumption figures (as per the New European Driving Cycle). A review by Fontaras et al. (2015) shows that such figures do not correspond well to real-life driving conditions with a divergence of 30%-40%, a conclusion also highlighted in ICCT (2017). Further still, the charging time-of-day of EVs can influence their GHG footprint due to the patterns of electricity supply in response to demand (Messagie et al., 2014 find a factor of 2 difference in the GHG intensity of the Belgian grid). Again, focusing on Belgium, Rangaraju et al. (2015) find charging BEVs in off-peak periods (e.g. at night) is beneficial for GHG emissions compared to charging in periods of peak demand.

A.3.2 LCA evidence from full life cycle studies
The so-called equipment cycle (impacts from production of the vehicle itself) is important for the full life-cycle performance of vehicles and its inclusion provides a more comprehensive view of the vehicle’s impact. The Nordelöf et al. (2014) review shows that in general (85% of the 79 studies examined), WTW GHG emissions dominate the full vehicle life-cycle, including for ICEs. The impact of the vehicle’s production increase with increasing electrification (i.e. largest for BEVs) due to the increasing importance of additional components, chiefly the battery pack, but WTW emissions still remain the most significant contributor. Nonaka and Nakano (2010) note that the balance between vehicle production and WTW emissions varies between countries and is sensitive to the assumptions on average driving distance, but that BEV and PHEVs always increase in benefits compared to ICE vehicles with longer lifetime distances driven.

At the full vehicle life-cycle level, van Mierlo et al. (2017) demonstrate in a Belgian analysis that BEVs and PHEVs have lowest GHG performance across a range of options including diesel, CNG and biomethane ICEs.
BEVs have lowest life-cycle impacts for photochemical oxidant formation. BEVs and CNGs have comparable life-cycle PM emissions, though when considering local (at point of use) and external occurrence (e.g. due to mining), BEVs are best-performing out of all compared vehicles (due to zero tailpipe emissions). Time of charging of EVs is also shown to be important for overall emission performance. Van Miero et al. also conclude that the battery pack is the most important component in terms of impacts in the vehicle production stage.

Peters et al. (2017) conduct a review of 113 battery LCAs (36 shortlisted) and find that depending on battery chemistry, the impacts of a lithium-ion battery can vary between 40 and 350 kg CO₂ per kWh of battery capacity, with an average of 110 kg CO₂/kWh battery capacity cited. GHG emissions are the principal focus of most vehicle LCA studies, but for the vehicle production stage, other impacts are also important. The Peters et al. review provides such information, though based on a more limited amount of data. Resource use is typically cited as significant, due to vehicle reliance on metals and battery reliance on rare earth minerals. In LCA, this is typically measured as the “Abiotic depletion potential”, which captures depletion of metals, rare earth minerals and fossil fuels. Overall, Peters et al. find that fossil fuel demand is the main driver in this impact category. However, the principal data source used by most reviewed studies is Notter et al. (2010), which is based on outdated characterization factors which place higher weight on fossil fuel depletion. Nordelöf et al. (2014) note for the same study that if more recent life-cycle characterization factors were to be used, higher weight would be placed on earth minerals and on e.g. metals such as copper and nickel. Further, common LCIA methods do not cover rare earth metals such as lithium. In Notter et al. (2010) in particular, BEVs have 35% higher overall impacts from resource depletion compared to ICEs in a full vehicle life-cycle assessment.

Related to resource use are the toxicity impacts of resource extraction. Again, less data is available but overall, batteries which lack certain metals (nickel, cobalt) perform better due to less impactful production but also end-of-life handling (lithium iron phosphate batteries perform best in the Peters et al. review). Overall for non-GHG impacts, Peters et al. note high uncertainties due to the limited amount of data obtained in the review. However, normalizing all considered LCA impacts to a common endpoint (all impacts are converted to a comparable unit), the study ranks resource depletion as the largest environmental impact, followed by acidification and human toxicity; GHG effects are fourth. Though endpoint normalization in LCA is uncertain and caveat-prone, it is nonetheless useful to illustrate the importance of non-GHG impacts of battery production.

A.3.3 LCA and autonomous vehicles

Autonomous vehicles are an emerging and fast-growing trend but detailed vehicle-level LCA studies are in general lacking. Gawron et al. (2018) is one such study and compares the full life cycle of a high-automation (Level 4) CAV (connected and automated vehicle) to ICE and BEV vehicles. The study is based on a detailed model of the CAV platform including onboard sensing and computer subsystems and their use. CAVs have increased weight and power consumption due to onboard equipment which increases their impacts, but this must be weighed against efficiencies from efficient driving algorithms but also larger energy consumption from faster highway speeds. In the study, the added fuel efficiency of CAVs is 5-22% (14% median).

The baseline scenario in the study is a BEV vehicle compared to a BEV with medium-sized mounted CAV subsystems. Compared to the baseline, the CAV subsystems add 22 kg weight and 240 W power consumption, the onboard computer contributing respectively 45% and 80% to the aforementioned and 43% to the subsystems’ GHG emissions. At the LCA level of the vehicle platform (without CAV fuel efficiency), BEV+CAV vehicles have comparable total life-cycle emissions. With CAV fuel-efficiency considered, the final autonomous vehicle estimate is 35,700 kg lifetime CO₂-eq or a 6% reduction compared to a non-autonomous BEV. ICE vehicles outfitted with autonomous capabilities have 40% higher lifetime GHG emissions.

The study outlines the following main conclusions:

- Electricity grid GHG intensity is important – a 2% increase in grid intensity in the study increased CAV GHG impacts by 1.6%. This highlights the importance of charging BEVs with less-GHG intensive grid electricity.
Results are sensitive to fuel efficiency gains from CAVs, highlighting the importance of eco-driving algorithms, which can add 7-16% efficiency compared to human driving. Increased aerodynamic drag from externally mounted components contributes to lessened fuel efficiency. Though in the future said components are likely to be downsized, in the near-term their impact would be significant.

The added mass and power consumption of onboard computers as well as additional components is almost 50% of total CAV subsystem GHG emissions. The argument for downsizing as above holds here as well.

While Gawron et al. demonstrate that CAVs may add GHG benefits, especially if improvements are made to the main GHG-intensive vehicle subsystems, their study is based on simple use phase assumptions. Impacts due to change in vehicle demand from CAVs may have different impacts depending on the envisioned scenario, as demonstrated by Wadud et al. (2016). The study is different from Gawron et al. analysis in that it disregards vehicle-level impacts (the direct impacts of automation) but instead assesses the impacts that automation has on overall traffic demand and behaviour. The factors considered include multiple fuel efficiency gains from downsizing vehicles, eco-driving, congestion impacts, increased highway speeds, dematerialised performance and stripping down of safety features (due to safer driving). Also included are behaviour-modifying impacts such as reduces cost of driver’s time (time not spent driving is used for other productive activities), increased travel from new user groups (e.g. young, elderly) and new mobility service models (e.g. ride-sharing and on-demand autonomous transport). The study is not LCA-based but rather based on literature-derived ranges for different parameters plus assumptions on their penetration in different scenarios.

The authors compare 4 scenarios – in all 4 travel demand increases, while energy intensity and demand vary between scenarios (from a best to a worst-case). In the best case (practically all energy use benefits of CAVs are realised), the net road transport energy intensity decreases by 40%. However, in the worst-case (small to no realisation of CAV energy use benefits, near-same travel demand), net road transport energy intensity increases by nearly 100%.

### A.3.4 Implications of electric and autonomous vehicle LCA studies for policy

The review highlights several issues that have important policy implications:

- Electricity grid intensity is a major determinant of GHG intensity at the vehicle level – charging electric vehicles (autonomous or otherwise) with low-GHG electricity decreases their overall GHG burden due to the WTW (fuel production and use) phase having strongest precedence in total vehicle GHG performance. In a WTW context, BEVs are overall less GHG-intensive than hybrid vehicles and ICEs under similar grid intensity assumptions. Charging conditions are important, with off-peak charging being more beneficial with respect to emissions.

- Driving conditions are important and EVs typically display highest gains compared to ICEs in urban (and somewhat so in suburban) settings. Differences in performance erode in a highway setting due to limited opportunity for regenerative breaking and due to no idling of ICE vehicles.

- Non-GHG impacts are important when considering vehicle production in addition to WTW – the battery pack & related component are the strongest determinants of non-GHG impacts, with resource depletion and toxicity (to humans and ecosystems) being most important. The end-of-life management of traction batteries (esp. recycling) is important for mitigating said effects. Different battery chemistries can lead to wide-ranging impacts and innovation in this area toward less impactful materials can help alleviate impacts, as well as toward increased energy density.

- Autonomous vehicles have some potential for lessened GHG emissions from increased fuel efficiency at the vehicle level, still potentially not fully-tapped due to room for innovation. However, the impacts of CAVs are much more dependent on tangential issues such as fuel-efficiency gains from design changes and eco-driving, as well as from changes in driving demand. In the nearer-term with less full automation, benefits from eco-driving may be realized given the right incentives for manufacturers but increasing levels of automation can potentially increase single-vehicle demand to a point where it outweighs benefits from fuel efficiency due to substantially more vehicles on the road. Policies for steering demand toward shared and on-demand mobility rather than privately owned & used vehicles can contribute to alleviating this effect.
In summary, battery electric vehicles outperform other fuel technologies in terms of GHGs but with the electricity grid GHG intensity being a major determinant and with off-peak charging being important. This outperforming is most pronounced in an urban (and somewhat so in a suburban) setting and is eroded on highways due to limited potential for regenerative breaking and no idling of ICES. The battery pack & related components are the prime source of non-GHG impacts in BEVs, with end-of-life management of batteries being important for mitigating these. Finally, autonomous vehicles can offer GHG benefits due to increased fuel efficiency and eco-driving (not yet fully exploited and with room for innovation). However, overall driving demand is a strong determinant in the sign of the net GHG effect – without policies for lessening the overall number of vehicles on the road (e.g. via shared and on-demand mobility), autonomous vehicles may in fact lead to an increase in GHG emissions due to higher accessibility and desirability of driving.
Appendix B  E3ME macroeconomic methodology

To explore the impact of the introduction of more fuel-efficient vehicles and the deployment of advanced powertrains into the European fleet, we model the outcomes of a number of potential uptake scenarios through new sales.

- Reference (REF) – A baseline where there are no further introductions of advanced powertrains of fuel-efficient technologies from 2017 onwards
- Fuel efficient technology (TECH) – No advanced powertrains are introduced after 2017 but fuel-efficient technologies are introduced.
- Electric vehicle (EV) – Advanced powertrains are rapidly introduced in the short to medium term alongside fuel efficient technologies with a phase out of ICEs by 2035, as shown in Figure B.1.

Figure B.0.1 New Sales mix in the EV scenario

<table>
<thead>
<tr>
<th>Year</th>
<th>ICE</th>
<th>HEV</th>
<th>PHEV</th>
<th>BEV</th>
</tr>
</thead>
<tbody>
<tr>
<td>2017</td>
<td>96%</td>
<td>4%</td>
<td>3%</td>
<td>1%</td>
</tr>
<tr>
<td>2020</td>
<td>94%</td>
<td>6%</td>
<td>7%</td>
<td>3%</td>
</tr>
<tr>
<td>2025</td>
<td>77%</td>
<td>23%</td>
<td>4%</td>
<td>2%</td>
</tr>
<tr>
<td>2030</td>
<td>50%</td>
<td>25%</td>
<td>17%</td>
<td>4%</td>
</tr>
<tr>
<td>2035</td>
<td>20%</td>
<td>53%</td>
<td>47%</td>
<td>0%</td>
</tr>
<tr>
<td>2040</td>
<td>15%</td>
<td>73%</td>
<td>85%</td>
<td>0%</td>
</tr>
<tr>
<td>2045</td>
<td>20%</td>
<td>85%</td>
<td>15%</td>
<td>0%</td>
</tr>
<tr>
<td>2050</td>
<td>20%</td>
<td>97%</td>
<td>3%</td>
<td>0%</td>
</tr>
</tbody>
</table>

Note: HEV = Hybrid Vehicle, PHEV = Plug-in Hybrid Vehicle & BEV = Battery Electric vehicle

The vehicle stock model calculates vehicle fuel demand, vehicle emissions and vehicle prices for a given mix of vehicle technologies. The model uses information about the efficiency of new vehicles and vehicle survival rates to assess how changes in new vehicles sales defined in the uptake scenarios, affect stock characteristics. The model also includes a detailed technology sub-model to calculate how the efficiency and price of new vehicles are affected, with increasing uptake of fuel-efficient technologies.
B.1 Results from vehicle stock modelling

In Figure B.2, we see the impact on fossil fuel consumption (petrol and diesel) for each of the scenarios. By 2030, fuel efficient technologies could reduce demand for fossil fuels by 26% relative to the reference case, while in the EV scenario a larger reduction, of 33%, is achieved. By 2050, the additional impact of fuel-efficient technologies tails off as technologies reach saturation in the vehicle stock; demand for fossil fuels is reduced by 45% compared to the reference case. However, when considering the advanced powertrains shift, we see much larger reductions as PHEV and BEV vehicles come to dominate the vehicle stock and as such emissions drop by 93% relative to reference.

In addition, we do see a small reduction in fuel demand in the reference scenario, due to both the recent improvements in fuel efficiency of ICEs filtering through the whole vehicle stock and the declining size of the passenger car fleet reflecting expected trends in vehicle demand in Germany (Shell, 2014).

In Figure B.3, we show the additional electricity demand needed in the EV scenario to power the increasingly electrified fleet as PHEV and BEV take an increasing share of the vehicle stock. By 2050, this reaches 244 million tonnes of oil equivalent. This increase in electricity demand is much smaller than the
reduction of oil demand reflecting the higher efficiency of the electric motor versus the internal combustion engine.

In Figure B.4, we see the impact of the scenarios on stock CO2 emissions at the tailpipe closely reflects the reductions in fossil fuel consumption. However, this takes a very narrow view of the emissions relating to passenger cars most notably are the implied emissions from the production of electricity used to power zero emission vehicles. However, analysis looking at the full WTW emissions (in the next section) shows that even under carbon intensive power generation mixes of some EU member states, the transition to zero emission vehicle leads to a reduction in overall CO2 emissions.
B.2 Investment Requirements

For the EV scenario where there is a substantial rollout of advanced powertrains considerable investment in charging infrastructure is required to support the vehicle fleet. The cost and density of infrastructure required follows the assumptions outlined in Harrison (2018). In Figure B.5, the scale of the infrastructure rollout is shown alongside the total investment cost requirements. By 2050, Germany requires 38m charging points requiring total investment of €63bn.

For the economic modelling, as well as evaluating the scale of the infrastructure investment required, it is also necessary to ensure that the additional investment is paid for. For home and workplace charging, we assume that the cost of the charging infrastructure is borne by the owner of the vehicle when the vehicle is purchased. As such we added the cost of home and workplace chargers to the vehicle purchase price. For public and rapid charging, which are likely to be installed in motorway service stations, car parks and supermarkets, would be paid for by the retail sector and this cost get past through into prices for the retail sector.
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Links between production, the environment and environmental policy

*Water treatment and supply – sector study*
This report is an Annex to the Final Report *Links between production, the environment and environmental policy*, ordered and paid for by the European Commission, Directorate-General for Environment, Contract ENV.F.1/FRA/2014/0063. The information and views set out in this study are those of the authors and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

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1 Introduction

Water treatment & supply has been identified in this study as a sector of interest, on the basis that it is an environmental services sector, with a strong influence on other sectors and strong links to the SDGs. In addition, it has a strong dependence upon ecosystem services, and a number of policy challenges exist around the abstraction, reuse and supply of water, to both households and industry.

When considering the environmental impact of water treatment & supply, it is perhaps not helpful to think of it as a conventional economic sector; since its whole aim is the abstraction, supply and recirculation of a scarce resource. The sector therefore has a significant environmental impact, and there are substantial opportunities for environmental policy to alter this impact, including in reducing abstraction levels through increased reuse, and management of the environmental impacts where abstraction does take place.

Water supply plays a key role in the economic system, well beyond the impacts that are typically considered in a socio-economic analysis. It is a key input to the agricultural sector; Europe’s plant and animal agricultural sector would not be able to operate without consistent access to water, and its importance is substantially underestimated if looked at purely from an economic perspective. In addition, while the supply of water may be only a minor input to most other sectors of the economy from a cost accounting perspective, it is fundamental to meeting basic human needs. For example, any private enterprise which employs workers would find it very difficult/expensive (or even impossible) to operate without access to clean water for sanitary and drinking uses.

Water access is the specific focus of SDG 6: ensure access to water and sanitation for all. It is also a key element of the 7th EAP, in terms of the focus on a resource-efficient economy, with specific mention of more efficient use of water resources.

The focus of this report is specifically on access to water in Europe. It draws heavily upon an interim output of the wider study, which looked at the water investment gap in Europe, to explore the economic aspects of the sector – specifically, what are the requirements made of the sector by existing EU water legislation, what is the size of the investment ‘gap’ required to be bridged to ensure that all existing legislation can be met, how might such a gap be met, and what would be the socioeconomic impacts of addressing that gap.

The funding of water treatment and supply, to provide the wide socioeconomic and environmental benefits attributed to improved access to clean water, has become an increasing focus in Europe over recent years.
2 The environmental footprint of the sector

Research questions:
- How do these sectors benefit from the environment?
- What is the investment need in Europe to ensure that all existing relevant legislation is met?
- How far away are current plans from the required level of investment?

2.1 Overview
The water treatment & supply sector, while directly drawing on natural resources itself through the extraction of water, also has links with other sectors with substantial environmental impacts. Water is a key input to both agriculture and some manufacturing processes. At a broader level, although most of the rest of the economy is not heavily reliant on water in value terms as an input to production, there are clear implications from a lack of access to clean water that make it an essential input to almost all sectors of the economy; for example, services firms operating out of offices spend very little money on water supply services, and they form a very small part of the total costs of inputs to these firms; however it would be very difficult for these businesses to operate without access to water for drinking and hygiene purposes.

As such, a steady supply of clean, potable water is essential for the successful operation of the European economy. European legislation already aims to provide such a supply, adhering to minimum standards. However, while investment in water supply and use is essential on social and environmental grounds, these arguments do not necessarily translate into a compelling financial case for investment. There is evidence that current investment levels are sub-optimal relative to the economic, social, and environmental benefits which would accrue from additional sector investment (OECD-WWC-Netherlands, 2017; EIB, 2016). This is well recognised by EU policy makers and poses the central question as to how public institutions in Europe work to increase investment levels.

2.2 The investment gap for infrastructure in Europe
Public investment in infrastructure in many Member States has fallen, both as a percentage of public spending and as a percentage of GDP. In the EU-15 infrastructure spending fell by more than 1% of GDP between 1970 and the financial crisis in 2007-2008.

The period after the financial crisis led to a slowdown in investment amongst developed countries. Gross fixed capital formation (GFCF), which provides a measure of investment in the economy, fell below the level seen in the pre-crisis period (1995-2006). The difference was €150bn in 2015Q3 across the EU Member States. Overall, investment was 2% lower than the pre-crisis average in the EU (and 2.2% lower within the Euro area) (EIB, 2016b).

Forecasts suggest that other macroeconomic challenges, such as an aging population and the consolidation of public budgets, will lead to further reductions in spending on infrastructure (Heise et al, 2014).

Addressing Europe’s investment gap was central to the Juncker Plan (Investment Plan for Europe) and the development of the European Fund for Strategic Investments (EFSI), which aims to leverage €315bn of investment across the EU economy.

2.3 Assessments of the investment gap for the water sector
Water infrastructure faces challenges in terms of financing. However, existing attempts to assess the investment gap of the European water sector are scarce. This section summarises the information covered in this scoping study including global, European and Member State level assessments.

2.3.1 Global assessments
Although the literature regarding investment needs in water infrastructure in Europe is limited, there have been several attempts in a global context to estimate current spending and the amount of capital required to meet the world’s water supply and sanitation needs. These attempts were typically a part of a broader
process of assessing the global needs for infrastructure including transport, energy, and telecommunications.

The first comprehensive study reviewed was undertaken by Fay and Yepes (2003), who valued the global infrastructure stock in 2003 at $15 trillion, of which 7.5% ($1.125 trillion) accounted for water supply and sanitation infrastructure. The corresponding investment needs between 2005 and 2010 implied an increase in spending by 2.1%, 1.5%, and 0.4% per annum in Low-Income, Middle-Income, and High-Income countries respectively.

Using a sectoral bottom-up method, OECD (2006) estimated that the OECD countries together with Brazil, China, India, and Russia would need to invest $2,380bn in total in the 2000-2030 period on water supply and wastewater treatment. Current expenditure on water infrastructure in 2005 were estimated at around $576bn, which would need to increase to $772bn by 2015 and $1,038bn by 2025. Furthermore, Volume 2 of the same report suggests that OECD countries will have to increase their investment in maintaining, upgrading, and replacing existing infrastructure in the water supply and treatment sector by almost 50 percent (OECD, 2007).

The McKinsey Global Institute (2016) estimated the global infrastructure needs for water and sewage to be $500bn annually between 2016 and 2030, while the current investment in 2013 was $200bn, leaving a gap of $300bn. Bhattacharya et al. (2016) calculated the projected investment for the same period to be $0.9 trillion per year (the gap is due to methodological differences and the great complexity of such estimations).

### 2.3.2 European assessments

Very few studies have explored in detail the investment gap for the water sector in the European Union. In most cases attempts to do this are done by extrapolating from best available data, or have a focus on a specific aspect of the water sector. A summary of the relevant assessments focusing on the EU water sector are given below:

**EIB – Restoring EU Competitiveness**

The European Investment Bank publishes reports analysing and forecasting their financing activities in the EU and third countries. As part of the EIB’s engagement with the Investment Plan For Europe, launched in 2015 and including the EFSI (see below), the EIB carried out assessments of potential investment gaps which might hinder Europe’s competitiveness (EIB, 2016). The EIB carried out an assessment of the investment needs for water with respect to:

- Water risk management (including scarcity and flooding)
- Water supply and treatment
- Compliance with water legislation
- Water infrastructure in urban areas
- R&D for competitiveness in European water technology

A summary of this assessment is given in Table 2.1.

**Table 2.1 Summary findings from EIB, 2016**

<table>
<thead>
<tr>
<th>Investment need/objective</th>
<th>Annual investment (€bn)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Required</td>
</tr>
<tr>
<td>Water security (including flood risk management)</td>
<td>15</td>
</tr>
<tr>
<td>Compliance and rehabilitation of Europe’s water infrastructure</td>
<td>75</td>
</tr>
<tr>
<td>Enhancing waste management/materials</td>
<td>8</td>
</tr>
</tbody>
</table>
**Additional needs for resilient and efficient urban infrastructure**

<table>
<thead>
<tr>
<th>Recovery</th>
<th>40</th>
<th>13</th>
<th>27</th>
</tr>
</thead>
<tbody>
<tr>
<td>TOTAL:</td>
<td>138</td>
<td>48</td>
<td>90</td>
</tr>
</tbody>
</table>

Source: EIB, 2016

In general, these values were based on extrapolating data for a single Member State or for another region (e.g. the U.S.) across the whole EU28 using a conservative conversion (EIB, 2017b). The EIB acknowledges

“There are some limitations of this assessment, for example:

**COWI - Urban Waste Water Directive (UWWD)**

COWI (2010) estimated the costs of compliance with the UWWD in all EU Member States, in order to assess the financing gaps. To do so, they apply cost functions derived from their own model, FEASBLE. This includes investment costs for additional collection and treatment infrastructure, but neither the costs of renovation of existing systems nor the costs related to sludge treatment and disposal are included. The information sources are national registries, additional information at the Member State level and a survey. The study estimates that the total expenditure required in 2006 to reach full compliance with the UWWD in the EU was €45.3bn. 63% of this amount (€28.3bn) was due to investment costs for waste water collection (Article 3 of the UWWD), 28% (€12.5bn) to the advanced treatment required by Article 5, and the remaining 10% (€4.5bn) to costs for generic treatment (Article 4). COWI also estimated re-investment needs for the Member States who have full registry file data, including both the annual re-investment needs due to the current infrastructure and those related to the new infrastructure still to be built.

**Eurostat - Protection expenditures for waste water**

Eurostat collects annual data for all EU Member States on protection expenditures related to wastewater, which include pollution prevention, sewerage networks, wastewater treatment, treatment of cooling water and all other activities and measures aimed at wastewater management. According to the last available figures for EU28:

- In 2013, the environmental protection expenditure on waste water management was €14.1bn by governments and €44.3bn by private and public companies specialised in sewerage, waste collection, treatment and disposal, and remediation activities.
- In 2013, the environmental investment in waste water management was €5.4bn by governments and €13.8bn by private and public companies specialised in sewerage, waste collection, treatment and disposal, and remediation activities.
- In 2013, the environmental expenditure for waste water management was €6.1bn by governments and €30.5bn by private and public companies specialised in sewerage, waste collection, treatment and disposal and remediation activities.
- In 2015, the total market output of companies working in the field of wastewater management in the EU28 was €44.5bn.
Note that a more detailed analysis of trends in environmental protection expenditures (for this and other sectors) is carried out in another Annex report to this study.

**Ecorys - Drinking Water Directive (DWD)**
Ecorys (2016) calculated that the overall annual investment needed to provide drinking water in the EU28 was €46.5bn in 2014 and €630bn between 1998 and 2014, including costs due to ‘normal’ pipeline network (e.g. maintenance costs). This figure is based on the expenditure costs for drinking water in 6 Member States (as calculated by VEWA, 2015, through a survey) and extrapolated to the whole EU using data on total population and differences in income per Member State. Using a range of sources, including a survey and published studies, Ecorys (2016) estimated that around 16.5% of the total costs (€8.3bn in the EU28) can be attributed to the implementation of the DWD, i.e. €109bn over the DWD life span. According to a study carried out in Germany by Aquabench, which was used in the Ecorys (2006) report, costs related to drinking water provision are due to:

- Taxes, levies, fees, concession fees, Water abstraction charges (7%)
- Metrology / quality control (3%)
- Building management (5%)
- IT technical support processes (15%)
- Resource Management / Water procurement / Extraction / Processing (18%)
- Treatment of drinking water (18%)
- Imputed Costs, such as the pipeline system and overall amortization (33%)
- Other costs, such as travelling to international events (1%)

**IEEP - Metering**
According to IEEP et al. (2012) the installation of metering in all irrigated EU land could cost around €0.2bn and full-scale implementation of metering in the whole of EU would costs €3.0bn (calculated based on the French experience).

**Investment in water quality and quantity**
Spit et al. (2017) estimated the costs related to public expenditures for water quality, quantity and waterways, based on the values calculated by T auw and Twynstra Gudde (2015) for the Netherlands, which are extrapolated to the EU28 using the benefit-transfer methodology. Such costs are additional to the costs of the drinking water sector mentioned above, because these expenditures reduce the costs for the water sector by decreasing treatment costs and other costs for the water sector. The authors calculated that investment in water quality for the EU28 are around €39.5bn per year and those in water quality approximately €8.8bn per year. They also estimated that investment needed for the maintenance of water ways in the EU28 are between €45.6bn and €131.7bn per annum. Finally, they calculated the operational expenditure of economic sectors (intake, treatment, recirculation and discharge treatment costs) as the sum of the additional treatment required for abstracted water (€11bn) recirculation of water (€4bn) and the cost to discharge water having appropriate quality (€19bn).

### 2.3.3 Member state assessments
Within this scoping study several Member State level assessments of the investment needs for national water sectors have been identified. It is possible that more national level assessments have been carried out across the EU-28 but it was not possible to review all national literature in the scope of this study.

Investment needs naturally vary between EU Member States. In central and eastern Member States, maintenance costs are perhaps the most pressing investment need. In western Member States with fewer pressing compliance issues, such as the Netherlands, prioritisation is a more involved process of weighing up costs/benefits, with approaches such as detailed impact assessments based on risk matrices employed. Network coverage is a prominent issue in multiple Member States, particularly in relation to urban wastewater standards, which necessitates large capital investment (D. Simidchiev, Hydrolia).
Spain
The Directorate General for Environment and the Centre for Hydrographic Studies (2017) summarised the foreseen investment in Spain required to meet the requirements of the Water Framework Directive (WFD) and water demand in general (regulation and transportation works), as well as the measures needed to mitigate the effects of floods and droughts, including the investment required by the Flood Risks Management Plans. The investment needed to increase water supply is estimated to be €3.4bn between 2016 and 2021, €2.8bn between 2022 and 2027, and €32.3bn between 2028 and 2033. The investment foreseen for flood prevention is €0.4bn in 2016-2021, €0.1bn in 2022-2027 and €0.04bn in 2028-2033.

According to a recent study (SEPAN, 2017), the investment need for water infrastructure in Spain between 2017 and 2021 will be €12.0bn, including €0.6bn for water provisioning, €4.4bn for water treatment, €4.9bn for water distribution and €2.1bn for water regulation. 30% of this investment is needed to comply with the Water Framework Directive and the Flood Directive.

United Kingdom
The infrastructure investment planned for the water sector in the UK totals £5.03bn (€5.64bn) in 2016/17, £4.99bn (£5.60bn) in 2017/18, £4.66bn (£5.23bn) in 2018/19, £4.03bn (£4.52bn) in 2019/20, £0.5bn (£0.5bn) in 2020/21 and £0.1bn (£0.1bn) afterwards. Between 9 and 11% of these amounts will finance water and sewerage projects of above £50m (£56m), i.e. big projects such as the Thames Tideway Tunnel, a major new sewer system that is being built to protect the tidal River Thames from pollution and should be functioning by 2023. In addition, around £0.5bn (£0.6bn) will be invested in flood defences per year over 2016 and 2021, and £1.39bn (£1.56bn) of investment are planned after the 2020/21 programming period. In terms of past investment, according to Phippard S. (2015), investment from privatised water and sewerage companies during the first WFD cycle in the EU have been €30bn, of which €6.1bn was specifically to meet environmental obligations. These costs are recovered by billing of customers and result in a reduction of phosphate and ammonia input to water bodies.

Bulgaria, Hungary and Romania
A recent report from the EU Court of Auditors concluded that Bulgaria, Hungary and Romania will need to invest approximately €6.0bn by 2020 to improve access to quality water and to comply with the DWD. These countries received €3.7bn in regional development and cohesion funds, which played a significant role in improving drinking water supply, but further investment is needed from the Member States. In particular, only 62% of the Romanian population is connected to the public water supply system; in Bulgaria 60% of water is leaking from the supply network before reaching the final consumer, and in Romania the equivalent figure is 40%.

Bulgaria
In Bulgaria, the water network is extensive in terms of coverage, but often fails to meet modern engineering requirements, and is deteriorating more quickly than current levels of investment are able to address. The Bulgarian strategy for the water supply and sanitation (WSS) sector (Republic of Bulgaria’s Ministry of Regional Development, 2014) calculates that about BGN 12.2 (€6.2bn) will be needed between 2014 and 2023 to meet the national WSS requirements. BGN 6.7–7.2bn (€3.4 – 3.7bn) will be required for improving water treatment. Urgent investment needs for renewal and replacement investment in water supply (abstraction, treatment, transmission and distribution) are estimated at BGN 5.0bn (€2.6bn), of which BGN 0.4bn (€0.2bn) are costs related to water supply compliance costs. BGN 4.4bn (€2.3bn) are needed for wastewater collection, and BGN 2.8bn (€1.4bn) are required for wastewater treatment. EU funds can cover 30-40% of the required investment for WSS, while the remaining 60-70% will have to be financed by the government and utilities. Stakeholders generally agree with the assessment but believe that the cost estimates may be slightly higher. To comply with the EU water acquis, plus sectoral effectiveness as per international benchmarks, stakeholders believe that the required investment is in the range of €5-6bn (D. Simidchiev, Hydrolia).
Table 2.2 Non-exhaustive summary of data assessing the investment gap in European water infrastructure

<table>
<thead>
<tr>
<th>Source</th>
<th>Geographic</th>
<th>Sector focus</th>
<th>Actual investment estimate (€)</th>
<th>Investment needs estimate (€)</th>
<th>Methodology</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>OECD, 2006</td>
<td>OECD plus Brazil, China, India, and Russia</td>
<td>Water supply &amp; wastewater treatment</td>
<td>€462.8bn* by 2005</td>
<td>€1,915bn* in total from 2000-2030 €620bn* by 2015 and €833.4bn* by 2025)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mc Kinsey, 2016</td>
<td>Global</td>
<td>Water supply &amp; wastewater treatment</td>
<td>€180.7bn* in 2013</td>
<td>€450bn* annually between 2016-2030</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bhattacharya et al., 2016</td>
<td>Global</td>
<td>Water supply &amp; wastewater treatment</td>
<td></td>
<td>€813bn* annually between 2016-2030</td>
<td></td>
<td></td>
</tr>
<tr>
<td>COWI, 2010</td>
<td>EU</td>
<td>Urban waste water treatment</td>
<td>€45.262bn required in 2006 to reach full compliance with the UWWD</td>
<td></td>
<td>Cost functions developed using the FEASBLE model, developed by COWI</td>
<td>This figure only includes new investment</td>
</tr>
<tr>
<td>ESIF, 2014</td>
<td>EU (2014-2020)</td>
<td>Water supply</td>
<td>ERDF will cover 36% of the investment and CF the rest of it, which will benefit more than 12 million EU citizens</td>
<td></td>
<td>This will bring water supply to 56% of the EU population, which is currently without</td>
<td></td>
</tr>
<tr>
<td>Source</td>
<td>Period</td>
<td>Sector</td>
<td>Investment/Expenditure</td>
<td>Notes</td>
<td></td>
<td></td>
</tr>
<tr>
<td>--------------</td>
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<td>------------------------</td>
<td>-----------------------------------------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ESIF, 2014</td>
<td>EU (2014-2020)</td>
<td>Wastewater treatment</td>
<td>ERDF will cover 51% of the investment and CF the rest of it, which will benefit almost 17 million population equivalent</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>EIB, 2016</td>
<td>EU (2007-2013)</td>
<td>Municipal and industrial water/wastewater</td>
<td>€30bn</td>
<td>Secondary resources €90bn a year for the period 2014 to 2020</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Material recovery</td>
<td>€3bn</td>
<td>€8bn per year</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Urban water infrastructure</td>
<td>€13bn</td>
<td>€40bn</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Water technology (R&amp;D)</td>
<td>€4bn p.a.</td>
<td>€7bn per annum by 2020,</td>
<td></td>
<td></td>
</tr>
<tr>
<td>GWI, 2016</td>
<td>EU</td>
<td>Water supply &amp; wastewater treatment</td>
<td>Annual increases (2-5% range) in investment to 2020, resulting in an average expected yearly investment of €33bn</td>
<td>Not open access.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ecorys, 2016</td>
<td>EU</td>
<td>Drinking water provision</td>
<td>Costs related to the supply of water: €46.5bn in 2014, €630bn between 1998 and 2014; of this €8.3bn is due to compliance with the DWD</td>
<td>The expenditure costs for drinking water in 6MS, as provided by VEWA (2015) (which obtained it through a survey), are used to extrapolate the overall figure for the EU, based on information on total population and differences in income and these figures also include the ‘normal’ pipeline network (e.g. maintenance costs).</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Source</td>
<td>Region</td>
<td>Sector</td>
<td>Investment/Gap Details</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>-------------------</td>
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<td>---------------------------------------------</td>
<td>---------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Split et al., 2017</td>
<td>EU</td>
<td>Public investment in water quality, quantity and waterways</td>
<td>Investment in water quality: €39.5bn per year; investment in water quantity: about €8.8bn per year; investment in the maintenance of waterways: between €45.6bn and €131.7bn per year. The values calculated by Tauw and Twynstra Gudde (2015) for the Netherlands are extrapolated to the EU28 using the benefit-transfer methodology. These costs are additional to those calculated by Ecorys (2016) as the cost of provision of drinking water.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>EIB, 2017</td>
<td>Italy</td>
<td>Water sector</td>
<td>Investment gap in Italian water sector growing by around €3bn per year.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ASCE, 2016</td>
<td>USA</td>
<td>Water supply &amp; wastewater treatment</td>
<td>The total investment gap through 2025 is expected to be €94.9bn* and €137.4bn* by 2040 if left unaddressed. The national economic model provides a comparison for scale.</td>
<td></td>
<td></td>
<td></td>
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</tbody>
</table>

*Note: The currency was converted from US Dollars to Euros using the corresponding yearly average exchange rate.
2.4 Trends in environmental pollutant releases

Many sectors of the economy use and/or create pollutants during production processes, with the potential to damage human health and the wider natural environment. Data on pollutant releases is published in the European Pollutant Release and Transfer Register (E-PRTR\textsuperscript{137}). Data is classified by NACE code of the emitter, pollutant, location (including country), year, release medium and volume, and as such it is possible to track its evolution over time.

In the analysis below, we present data in volume terms, but also environmental impacts. These are calculated by applying coefficients reflecting the toxicity of different pollutants, taken from ReCiPe2016 LCIA, according to whether they were released via air, water or land. This allows the summation of different pollutant based upon the impact that they have on human health (measured in disability-adjusted life years, DALYs) and ecosystem health (measured as disappeared species per year, species.years).

In the water treatment and supply sector, as with other sectors analysed, only a small proportion of all recorded releases have a specific pollutant attributed to the release, and therefore for only around 10% of

Figure 2.1 Sectoral observations available in E-PRTR

![Figure 2.1](image)

data points are we able to calculate environmental impacts (see Figure 2.1).

Source: Author calculation, using data from E-PRTR and ReCiPe2016 LCIA.

\textsuperscript{137} \url{http://prtr.eea.europa.eu/#/home}
The impacts upon human and ecosystem health from water treatment and supply pollutant releases broadly mirror each other, reflecting the fact that the mix of pollutants released do not change much over time in a way that would damage one group more than the other (see Figure 2.2). There was a spike in pollutant releases in 2011 and 2012, reaching 442 DALYs in 2012. There is also a lesser spike in health impacts in 2015. Both are driven by large CO2 emissions. In 2011 and 2012 these emissions are from the...
Czech Republic; in 2015 from Germany (see Figure 2.3), in both cases related to on-site energy production.

Non-CO2 releases are dominated by methane, but in relatively small volumes (see Figure 2.4).
3 Current trajectory – direction of travel

3.1 Research questions

- How does environmental policy affect the links to the environment (i.e. what has been the impact of policy so far?)
- How does the environment and environmental policy affect the links between these sectors and growth, jobs and investment?
- The links between production in these sectors and consumption including consumption orientated policy tools? (i.e. to what extent does demand-side policy affect the sector)
- The evolution in consumers and investors demand for increased transparency on environmental performance? (are consumers demanding water with less environmental impact?)
- Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?
- What has driven changes over time (post-2000), and what changes are expected to occur in the future?
- How have the answers to all of these questions changed over time, and how are they forecast (modelled) to change in the future?
- Are there examples of striking differences to any of these questions between Member States (and if so, why)?
- What is the potential in Europe of developing further the ‘water tech’ sector (presumably focussed around minimising environmental impacts)?

3.2 The role of compliance with EU water legislation

The investment needs regarding water in Europe are strongly linked to the implementation of EU water law. Meeting EU objectives requires investment. Therefore, a key consideration in examining investment needs is the extent of compliance with EU water law, i.e. where there is non-compliance this might indicate that further investment is needed and where there is compliance this might indicate that there is not an EU legal driver for further investment.

In considering the EU water acquis, focus should be on those measures which have the largest potential consequence for spending. These are:

- The Urban Waste Water Treatment Directive 91/271/EEC (UWWTD), which requires investment to collect and treat waste water.
- The Bathing Waters Directive 2006/7/EC (BWD), which has required significant improvements/investment in sewage treatment to meet its objectives
- The Drinking Water Directive 98/83/EC (DWD), which requires investment for water distribution and treatment
- The Water Framework Directive 2000/60/EC (WFD), which sets objectives for all water bodies that may require investment to address pressures impacting on these objectives.

It is important to note that the compliance deadlines for the UWWTD, BWD and DWD have all passed and, therefore, non-compliance is strict legal non-compliance. For the WFD, the use of exemptions based on cost has effectively allowed Member States to delay meeting these objectives. Such exemptions can only be used for three River Basin planning periods and therefore all objectives should be met by 2027.

The stakeholders presented a unanimous opinion that the EU water acquis and compliance with EU policy is by far the principle driver of investment. While some projects can have better economic returns than others, or provide higher added social and environmental benefits, regulatory compliance is the primary consideration for decision-making. Following compliance, maintenance of sustainable services and higher efficiency are a major aim, through improving connectivity, infrastructure optimisation and aiming to cutting losses.

This section explores the extent of compliance with water sector directives and comments directly on conclusions on investment needs that can be drawn from this.
The Urban Waste Water Treatment Directive

The most recent (8th) implementation report on the UWWTD was published in 2016, covering 25 Member States (excluding Croatia, Italy and Poland, and based on 2012 data, so that some transition periods were still ongoing). The report found that “Compliance rates at EU-15 level have been found to be, in general, very high. At individual Member State level, rates of 95-100% are quite frequent. Results are much lower in EU-13, especially in sensitive areas. However, significant progress has been observed since the time of the last report.” The report concluded that to reach full compliance, action is needed to address the following gaps:

- 11 million population equivalent (p.e.) (2%) have to be connected and treated, or addressed through individual or other appropriate systems;
- 48 million p.e. (9%) of the urban waste water already connected have to meet the performance of a secondary treatment; and
- 39 million p.e. (12%) of the urban waste water already connected have to meet the performance of a more stringent treatment.

The Communication notes the important role of the European and Structural and Investments Funds to support investment. In the programming period 2007-2013, around €17.8bn from Cohesion Policy funds had (at the time of publication of the Communication) been allocated to UWWT infrastructure in 22 Member States.

The Communication noted that Member States reported on 8,600 projects related to infrastructure for waste water collection (about a third) and treatment (about two thirds of projects), as well as for individual or other appropriate systems, to be carried out between 2014 and 2027. Most focus on ensuring compliance. The total forecasted investment needed for new projects necessary to reach full UWWTD compliance was estimated at €22bn, equally distributed between collection and treatment infrastructure, with 25% of this co-funded from EU funds. Investment is also needed to extend and maintain current systems. However, the Communication also noted that “the waste water management in the goods and services sector represents more than 600,000 jobs, an annual production value of more than €100bn and an annual added value of about €42bn (investment, maintenance, operation, export of technology and knowledge).” Figure 3.1 shows the current and expected annual investment across different parts of the EU.
The investment required varies between Member States in absolute and per capita terms (see Figure 3.2) and will increase in some Member States and decrease in others.

### 3.2.2 Bathing Waters Directive

The BWD requires designated bathing waters to meet two microbial standards to ensure health is protected. The directive also includes guide values which are more stringent but should be aimed for. The latest report on compliance covers the 2016 bathing water season.

In 2016, 96.3% of bathing waters met minimum BWD quality standards (up slightly from 96.1% in 2015). 85.5% of bathing water sites met the BWD most stringent ‘excellent’ water quality standards (up from 84.4% in 2015). The EEA report highlights that “bathing water policy is one of the success stories in EU water policy and important to protect human health and the environment.” Therefore, the report concludes that this major improvement in compliance has been driven by investment in waste water treatment (both for the UWWTD and the BWD, such as ozone treatment or ultraviolet light disinfection).
There is only a small level of non-compliance left, where bathing waters were rated as poor. It should be noted that one appropriate response in the BWD is to close such sites to bathing. However, clean-up is also appropriate. However, the remaining pressures are now largely not from poorly treated waste water discharge, instead they are from storm overflows (where urban run-off overflows the sewage system during heavy rainfall) and run-off from agricultural land where animal manure introduces microbial contaminants to the water. Both pressures can be addressed with additional investment. Further, it should be noted that investment to address pressures from storm overflows is necessary to deliver the objectives of the WFD.

### 3.2.3 Drinking Water Directive

The DWD sets standards for drinking water delivered to households. These standards include a range of microbial and chemical substances. Meeting these standards requires investment in water treatment to remove contaminants and in a distribution system that ensures treated water remains of good quality until it reaches the households.

The most recent implementation report from the Commission was published in 2016 and covered Member State reporting for the period of 2011-13. Across the EU compliance with the DWD is extremely high. It was 99% for microbiological and chemical parameters. The lowest compliance rate was for arsenic (98.83%) (driven by catchment geology in Hungary and Italy). However, overall this level of compliance is very high and represents huge progress since the first DWD was adopted and compliance rates were poor. This has been driven by major investment.

The implementation report examines the causes of non-compliance where it does occur. It places these into three categories: “catchment related”, “treatment related” and “distribution related” (public network and domestic network). Where such causes occur affects what action/investment is needed to address them. For some substances; biological parameters (coliform bacteria, colony count, E-coli, Enterococci, Clostridium) and iron - the cause is not known. For ammonium, manganese, pH, chloride, sulphate, arsenic and nitrite, the cause is catchment-related. For lead, the cause is distribution-system related. Member States are undertaking different remedial actions to address these problems. For coliform bacteria contamination, most remedial actions (67%) were related to the public distribution network or treatment infrastructure and operation (through better disinfection). Remedial actions concerning arsenic were mostly related to treatment (46 %) or catchment (29 %). For lead, 67 % of reported remedial actions consisted of the replacement or disconnection of lead pipes in the domestic distribution network.

Overall, therefore, compliance is very high, but where problems occur, further investment is needed to address this.

Finally, it should be noted that the DWD is being evaluated. Part of this evaluation will consider updating of standards in the light of recommendations from the World Health Organization. It is possible that new standards may require additional investment to meet them.

### 3.2.4 Water Framework Directive

The WFD establishes detailed objectives for the achievement of good status in surface and groundwaters across the EU to be met variously to 2027. Currently the Commission is undertaking a series of analyses of implementation of the 2nd RBMPs, reported in 2016, so that assessing compliance at this point is difficult.

The 2012 Commission Blueprint to safeguard Europe’s Water Resources found that about half of EU surface waters were unlikely to reach a good status in 2015, thus demonstrating the compliance challenge. The two most important pressures prevent achievement of good status were hydromorphological pressures and diffuse pollution from agriculture. However, other pressures (e.g. waste water) are also important. Further, the Commission concluded that “exemptions are too widely applied and without adequate justification”. The Commission found that many Member States “have planned their measures based on ‘what is in place and/or in the pipeline already’ and ‘what is feasible’, without considering the current status of water bodies and the pressures identified in the RBMPs as preventing the achievement of ‘good status’. Instead of designing the most appropriate and cost-effective measures to ensure that their water achieves ‘good status’, thus tackling the persisting performance gap, many Member States have often only estimated how far existing measures will contribute to the achievement of the WFD’s environmental objectives.”
The Commission (COM(2015)120) has highlighted the following areas where investment is needed to address pressures preventing achievement of WFD objectives:

- **Sewer overflows** are a major source of urban pollution and investment is needed for stormwater management through sustainable urban drainage systems, modular building of WWTPs and sewerage systems, provisions for asset replacement and upgrade, asset management, etc.

- **Industrial activities** are a threat and treatment of industrial waste water needs to be improved to address some priority substances

- **Over-abstraction** requires investment in improved irrigation

- **Improved flow regulation** requires extensive investment to address hydromorphological barriers and pressures.

With regard to investment, the Commission concluded that Member States “have not exploited to the full extent EU funding possibilities to support objectives under the WFD notwithstanding some good examples” (e.g. using rural development funds for rural waste water infrastructure). Further, the lack of cost recovery as required by the WFD, including for environmental, resource and infrastructure costs, not only disincentivises investment, but could lead to increased future costs.

The issue of investment needs was further explored in SWD(2017)153, although this focused on the agriculture sector. This highlighted the importance of using measures under the CAP and Regional Funds to target needs in rural areas and support on-farm investment to tackle the problems facing WFD compliance. Thus, financial support to farmers for implementing water and nutrient management measures is often perceived as insufficient. Examples of projects could be: smart irrigation, water reuse and smart land management measures. In addition, existing CAP measures could be supported by additional investment, e.g. greening, agri-environment schemes, manure storage, etc. Delivering the necessary investment is thought to be hampered by governance problems - from adequate communication between authorities within Member States, to problems of timing between adoption of RBMPs and RDPs.

The SWD also noted that the Commission, together with the European Investment Bank, is reviewing possibilities for supporting investment to help achieve good water status. The European Fund for Strategic Investments may also provide another funding stream.

### 3.2.5 Proposal for minimum requirements for water reuse

This proposal, made in May 2018 (SWD(2018)250), is intended to extend the measures set to address water reuse in both the Water Framework Directive and the Urban Waste Water Treatment Directive. It seeks to address the fact that only six Member States (Cyprus, Greece, Spain, France, Italy and Portugal) have explicit requirements on water reuse. The proposed regulation defines minimum standards for reclaimed water to be used for agricultural purposes.

In terms of investment, the associated impact assessment associated with the preferred approach (a legal instrument with a “fit-for-purpose” approach) is expected to require investment in treatment facilities of €38/m³/day.

### 3.3 Sources of finance and financial instruments in theory

Water utility infrastructure is financed in a variety of ways, including private, public, national and international sources of finance – often in combination – and via a range of different financial instruments. Generally, recurrent financing is based on a combination of tariffs, taxes and transfers (the ‘3Ts’). This revenue stream is also used as basis for attracting and repaying private finance, including loans, bonds (debt) and/or equity, which is used to address financing gaps or meet short-term budgetary needs (Lago et al, 2011; OECD, 2009). A different mix of financial instruments might be necessary to fund each phase in an infrastructure project, as each phase has different risk and return characteristics (Ehlers, 2014).

While the 3Ts concept is still useful for understanding financial flows in the European water sector, in 2015, the World Water Council (WWC) and OECD suggested an updated categorisation to reflect new financial sources and instruments made available for water infrastructure in recent years (see Table 3.1). The view that additional financial sources for investment is necessary is shared with important stakeholders, who emphasise that a balance must be struck to ensure financial stability.
Table 3.1 Water infrastructure instruments (OECD)

<table>
<thead>
<tr>
<th>3Ts and other contributions to recurrent finance</th>
<th>Loan and bond finance</th>
<th>Equity finance</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tariffs and user charges</td>
<td>Public development banks</td>
<td>Institutional investors</td>
</tr>
<tr>
<td>Taxes (national budgets)</td>
<td>Commercial banks (incl. project finance)</td>
<td>Sovereign Wealth Funds</td>
</tr>
<tr>
<td>ODA</td>
<td>Institutional investors</td>
<td>Specialised water funds</td>
</tr>
<tr>
<td>Philanthropic funds</td>
<td>Sovereign Wealth Funds</td>
<td>International Financial Institutions</td>
</tr>
<tr>
<td>Property taxes and other levies and contributions</td>
<td>Public bond issue</td>
<td>Private equity funds</td>
</tr>
<tr>
<td>Self finance by users</td>
<td>International Financial Institutions</td>
<td>Venture capital</td>
</tr>
</tbody>
</table>
<pre><code>                                                             | Private equity funds |
                                                             | Public-Private Partnerships |
                                                             | Individual shareholders |
                                                             | Institutional investors |
</code></pre>

Source: Developed from World Water Council and OECD, 2015
Note(s): This table represents the global water sector. Some financial sources might not be relevant to Europe.
3.4 Financing available to the EU water sector in practice

3.4.1 National sources and instruments

Water sector infrastructure investment in the EU has primarily been financed by national governments and local authorities through public budgets. It has traditionally been common for large European cities to finance urban water services by issuing municipal bonds as debt security (World Water Council and OECD, 2015). There are also various schemes set up by individual EU Member States. The Netherlands has for example established a public bank – the Netherlands Water Bank (NWB Bank) – which arranges short- and long-term loans for water authorities, municipalities, provinces, social housing, healthcare, educational institutions, public-private partnerships (“PPP”) and activities in the field of water supply and the environment. In 2016, NWB Bank issued new long-term loans totalling over €7bn (NWB Bank, 2017).

EU funding sources and instruments

The EU water sector is eligible for European cohesion policy funding from the European Regional Development Fund (ERDF) and, depending on region, the Cohesion Fund (European Commission, 2015). In the 2007-2013 programming period, around €17.8bn was allocated from the Cohesion Policy funds to 22 Member States to carry out investment to meet the provisions of the Council Directive 91/271/EEC concerning urban waste water treatment (COM(2016) 105 final). So far in the 2014-2020 period, funding via all the European Structural and Investment Funds has contributed to improved water supply for over 12 million people (including planned investment) and improved wastewater treatment for almost 17 million people (including planned investment) (European Commission, 2017).

European Investment Bank

The EIB has been the largest source of loan finance to the water sector to date – both in Europe and globally – compared with other international financial institutions, providing €64bn for 1,400 water sector projects worldwide (European Investment Bank, 2017, n.d.). This equates to an average of €4bn annually (T. van Gilst, EIB).

The EIB lends to public and private utility companies, national and local authorities, or directly to individual projects. It offers project loans, intermediated loans, Natural Capital Financing Facility (NCFF) and Joint Assistance to Support Projects in European Regions (JASPERS). It lends on average 30% of project investment costs but can lend up to 50%. Lending focuses on the “modernisation and extension of existing distribution, collection and treatment networks as part of large-scale national/ regional/municipal capital expenditure programmes”. Investments have been made in all Member States.

Between 2001 and 2007 the EIB lent €9.1bn to the water sector in 19 of the EU-27 countries. Over this period, 37% of the loans went to the UK and Germany (EIB, 2008). In the period 2003-2007 the EIB increased its lending to the water sector to an annual lending of around €2.1 bn. Tightening of water legislation has been a major drive of investment in the EU water sector (EIB, 2008). Between 2008 and 2012, the EIB lent about €17bn to water-related projects, including irrigation and sewerage, 89% of which was for schemes in EU Member States (European Investment Bank, 2013). In 2016, the EIB lent €37m to EU water and wastewater projects related to climate action (European Investment Bank, 2017). In the period 2016 to 2017 the EIB lent a total of €4.4bn to EU member states for projects relevant to water and waste water management (EIB, 2017).

European Fund for Strategic Investments

In June 2015, EIB and the European Commission launched the European Fund for Strategic Investments (EFSI), intended to mobilise private financing for strategic investment. EFSI has capital of €33.5bn (€26.0bn from the European Commission and €7.5bn of EIB’s own capital) and a ring-fenced budget for infrastructure and innovation, including water infrastructure (Mestres Domènech, 2017). The target is to mobilise a total of €315bn over three years, and in 2016 alone, EFSI-related total investment reached €163.9bn. Since the 2015 launch to the end of 2016, EFSI investment contributed to the construction or upgrade of almost 120km of water mains or distribution pipes with over 2 million people benefiting from safe drinking water (European Investment Bank, 2017).

At the EU level, for the period 2014-2020, the ESIF has planned investment in water supply, wastewater treatment, and flood protection benefiting 12.4, 16.9, and 13.2 million people respectively. By the end of
this programming period, the EU will have reduced the European population that is currently without access to public water supply by 56% (Investing in Jobs and Growth). The financing of these planned projects will be realized by the ERDF and the CF, with different levels of participation in each case. The ERDF will provide 36% of the funds needed to improve water supply, 51% of the investment in water treatment and 57% in the flood protection schemes, while the remaining capital needs will be covered by the Cohesion Fund.

EU-wide financial programmes
Water sector investment in the EU is supported by a number of programmes which provide funding, such as LIFE and the Danube Transnational Programme; or those which facilitate networking and advice in relation to accessing funding for water-related investment, such as ACQUEAU Open Calls, INNEON and RIS3 Regions (EIP Water, n.d.).

Other International Financial Institutions and development banks
Some EU countries are eligible for borrowing through the World Bank. In 2016, the World Bank lent almost €370m for Water, Sanitation and Flood Protection in Europe and Central Asia (World Bank, 2016). The improvement of water and wastewater systems is also a mandate of the European Bank for Reconstruction and Development (EBRD). By the end of 2013, EBRD had financed 153 water and wastewater projects to a value of €2.18bn, focusing on the efficient provision of drinking water in larger towns and cities in developed countries (European Bank for Reconstruction and Development, 2014). Further, multi-lateral development banks, such as the Council of Europe Development Bank (CEB) offers finance to infrastructure projects in the areas of “urban renewal and rural modernisation”, where these are part of national, regional or municipal budgets (CEB, 2017). The CEB have financed a number of water relevant projects in EU Member States including:

- Large-scale irrigation projects in Spain;
- Wastewater treatment and irrigation in Cyprus

Over the period 1957-2016, the total volume of projects approved in this sector was €9bn, representing 15% of all loans approved.

Private financial sources and instruments
Private repayable financial instruments to fund water industry investment include debt (various loans, including bonds and export credits), with fixed (and often interest) payments to the provider, and to a lesser extent equity (the buying and selling of stock). One example of the former is green bonds, where water accounts for about 10% of the €55bn raised globally. “Sustainable water management (including clean and/or drinking water)” is one of the categories of projects deemed eligible according to the Green Bond Principles (GBPs) (Boccaletti, 2015). As one example from Europe, the NWB Bank launched a 5-year, €500m green bond in 2014 to lend to Dutch water authorities (World Water Council and OECD, 2015), and in August 2017, British Anglian Water issued a utility sector green bond of over €270m (WaterBriefing, 2017).

There is an increasing interest from non-traditional private financial sources to invest in European water infrastructure, including, for instance, various kinds of institutional investors such as pension funds, insurance companies, Sovereign Wealth Funds, specialised water funds and new international development banks (Linklaters LLP, 2014; World Water Council and OECD, 2015). Institutional investors often invest long-term and have large pools of capital (Collins, 2017). For example, the Thames Tideway Tunnel in London, UK, is partly funded by UK pension funds (Mooney, 2016). The UK defence group BAE Systems and the Swedish national pension fund AP1 have recently invested in a €250m water fund that will focus on industrial water projects in Europe and Asia (Flood, 2016).

Specialised water funds are a type of private equity funds, with the Swiss investment manager Pictet’s water fund being the world’s largest at €2.4bn. Between 2010 and 2011, global assets of funds focused on water and specialist water funds nearly doubled to just over €20bn in 2011 (de Sa’Pinto and Menon, 2012).

Further, with the significant and increasing need for new and updated water supply and waste water treatment technology, venture capitalists have recently turned their attention to the water industry (Gies, 2012). While venture capital is still significantly lower in Europe than, for instance, in North America
(European Investment Bank, 2016), the venture capital stakeholder that provided input to this report emphasised that the current inefficiencies create opportunities for both investment and innovation within the EU.

Table 3.2 presents a selection of European water sector projects and illustrates typical financing instruments.
<table>
<thead>
<tr>
<th>Country</th>
<th>Scope</th>
<th>Finance</th>
<th>Source</th>
</tr>
</thead>
</table>
| Italy   | Acqua di Arezzo PPP project. Upgrading water supply and treatment infrastructure in 37 municipalities in the region of Alto Val d’Arno, Tuscany | Total costs: €70m  
EIB financing: €44m (direct loan to Nuove Acque SpA)  
25 year concession agreement, mixed public private partnership. 54% owned by 37 municipalities 46% private consortium, including 23% owned by Suez-Environnement. | (EIB, 2008) |
| Italy   | Financing of several small water utilities to ensure compliance with EU and national environmental legislation. The investment will deliver substantial health advantages and environmental quality improvements. | Total cost of project: €400m.  
EIB finance: €200m  
Remaining finance from public sector. | (EIB, 2017) |
| UK      | Thames Tideway Tunnel, 2016-2023, construction of a new 25km sewer tunnel. Delivers environmental improvements to the River Thames and brings the UK into compliance with the European Directive. | Total cost of project: £4.2bn.  
EIB loan of £700m.  
Remaining finance from private UK water industry, but includes governmental protection against certain risks, including insurance, cost overruns and termination compensation. | (Tideway, 2017) |
| UK      | Flood defence investment in Calder Valley, York, Cumbria and Leeds. | Total cost of project £4bn, nearly £3bn until 2021, financed by the UK infrastructure investment pipeline (UK government). | (UK Government, 2016) |
Further PPP loans were obtained from Rabobank and Dexia Credit Local. | (EIB, 2008) |
| Netherlands | Investment in flood protection and water waste management infrastructure, implemented 2015-2020. Investment motivated by expected higher sea levels and more rainfall, which affects availability of sufficient drinking water. | Total cost of project: €400m.  
EIB finance: €100m  
Remaining finance covered by the Dutch Water | (EIB, 2015) |
<table>
<thead>
<tr>
<th>Country</th>
<th>Project Description</th>
<th>Cost and Financing Details</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Portugal</td>
<td>Investment in the water and wastewater sector, during 2014-2020, with multiple objectives including compliance with the Urban Wastewater Treatment and Drinking Directives, as well as improving the quality of water bodies.</td>
<td>Total cost of project: €836m. EIB finance: €418m. Remaining finance covered by the Portuguese Aguas de Portugal SGPS SA</td>
<td>(EIB, 2016)</td>
</tr>
<tr>
<td>Romania</td>
<td>Six counties received funding for water supply and wastewater infrastructure investment (Prahova, Bacau, Maramures, Dolj, Bihor, and Botosani)</td>
<td>EBRD loans of €9.2, €16.4, €10.3, €12.8, €5, and €7.4m, remaining public sector investment</td>
<td>(EBRD, 2014)</td>
</tr>
<tr>
<td>Croatia</td>
<td>Rijeka Water and Wastewater Investment Project.</td>
<td>EBRD loan of €13m, remaining public sector investment</td>
<td>(EBRD, 2014)</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>National Flood Prevention Strategy 2002-2012. Increase protection for 800,000 people</td>
<td>EIB finance: €360m Total costs: €750m</td>
<td>(EIB, 2011)</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>City of Plzeň to support its 5-year municipal investment programme for water and wastewater facilities</td>
<td>EIB finance: €15.3million</td>
<td>(EIB, 2008)</td>
</tr>
<tr>
<td>Poland</td>
<td>Aquanet’s investment programme to improve the urban environment of Poznan. Wastewater schemes aimed to reduce pollution load originating from some 750 000 residents in Poznań and nine neighbouring municipalities</td>
<td>EIB finance: €128m</td>
<td>(EIB, 2008)</td>
</tr>
<tr>
<td>Romania</td>
<td>Construction of the Bucharest Glina Wastewater Treatment Plant and infrastructure development programme for small and medium-sized towns (SAMTID) launched in 2005</td>
<td>EIB finance: €25m</td>
<td>(EIB, 2008)</td>
</tr>
<tr>
<td>Czech Republic</td>
<td>Partial funding of infrastructure investment, including local water treatment, network facilities, and waste treatment.</td>
<td>Total costs: €500m CEB lending: €250m (three traches 2011, 2014, and 2015) to Komerční banka.</td>
<td>(CEB, 2017)</td>
</tr>
</tbody>
</table>
4 Future Policy Priorities

4.1 Research questions

- What are the links between these sectors and the Sustainable Development Goals and the different targets?
- What is stopping the development of ‘water tech’ sector in Europe? Is it a lack of relevant R&D, or a lack of investment? If the latter, how much investment are we talking about?
- Are there some differences and good practices to be learnt from non-EU countries?
- What are the opportunities for these sectors (including jobs, growth and investment opportunities) provided by the environment and by environmental policy?
- What are the potential business evolutions (business model, product specificities, etc.) driven by environmental policies or voluntary initiatives influenced by the environment in a given sector?
- What does a more resource-efficient water system look like? Is it led by supply- or demand-side measures? What are the economic and resource implications of this?
- If we successfully developed a European ‘water tech’ sector, how would we do it (what policy levers?) and what would it look like? Does it create jobs/economic activity in Europe, and does it create the potential for exporting this globally?

4.2 Challenges to the financing of investment in water treatment and supply

There are several factors and sector-specific characteristics that contribute to the underinvestment in the water sector. These include the long-term nature of investment projects, poor management of existing stock, emerging challenges linked to climate change, the risk of low return on investment, and complex site specific and legislative requirements. Based on the literature reviewed, an analysis of some of the drivers of these challenges is outlined below:

4.2.1 Water infrastructure investments represent sunk costs

As water infrastructure is a long-term asset, its performance is dependent upon physical factors (such as geography, topography and climate) as well as socio-economic drivers (investment, maintenance, quality of execution). Generally, because investment in the water infrastructure tends to be based upon returns over a long period of time, existing infrastructure has the characteristic of a “sunk cost”. Where infrastructure is already in place, it is likely that policies will focus on maintenance, repair and upgrades. Where new infrastructure is developed there is likely to be a greater emphasis on cost recovery (EEA, 2013; EEA, 2014; ACTeon et al 2015).

4.2.2 Water infrastructure in Europe is aging

Like other sectors in Europe, water infrastructure for supply and treatment is aging. Consequently, investment to cover maintenance will become insufficient in some areas (EIB, 2016). The EEA also acknowledges the importance of investment in Europe’s aging infrastructure, including investment in new technologies, training of staff and public awareness raising (EEA, 2014).

In Eastern Europe, the rapid rate of urbanisation has resulted in pressures on fresh water supply in some places (CEB, 2017). In the EU-13 (the newest Member States, in central & eastern Europe) it is anticipated that there is a need for significant infrastructure investment. Capital expenditure is necessary to comply with legislation, and to cover a backlog of low investment in water infrastructure in these countries (EIB, 2008).

4.2.3 Flooding and droughts

Specific challenges to water are also the increased risk of water scarcity and flooding (EIB, 2016). There is a need for investment to reduce vulnerability to floods and droughts and to support natural water retention measures (e.g. green infrastructures and green Common Agricultural Policy) (EEA, 2014). Floods continue to...
be the largest course of GDP losses from natural disaster, costing €150 over 2002-13. The economic cost of droughts was €86bn across Europe over the period 1980-2010 (EEA, 2010).

4.2.4 Riskiness of investment

A common misconception within the water sector is that infrastructure, upkeep and maintenance costs are covered by the normal revenues of water utilities. This is often not the case, and shortfalls on repairs and maintenance lead to a need for higher investment (EIB, 2016). Due to lack of clarity on the economics of water resource planning, the sector has seen decades of underinvestment (OECD-WWC-Netherlands, 2017; OECD, 2011).

Competitiveness is largely determined by the risk-return profile of a project, which is influenced by expected revenue streams and the underlying risk associated with the investment (OECD-WWC-Netherlands, 2017). To provide an adequate risk-return profile, institutions need to identify the drivers of water related risks, such as policies of urban development, measure and monitor water sector costs, and identify the benefits of water sector investment and communicate where they occur (OECD, 2016). Due to limited data, estimation of benefits is complex, and such benefits are often hard to monetise. But several stakeholders encourage extensive cost-benefit analysis as a tool for investors to evaluate projects (OECD-WWC-Netherlands, 2017).

4.2.5 Prioritising member state investment needs

In general, national governments agree on the importance of water sector investment, but their spending performance does not support this. The sector is given a disproportionally small share of the public budget, and no current policy is attracting private investor to pick up the slack (OECD-WWC-Netherlands, 2017; Winpenny, 2003). Based on the evidence presented in the previous section, there are several sector-specific characteristics to consider for successful policy formulation. Water sector investment projects are competing with other infrastructure projects for financier's attention.

4.2.6 Infrastructure needs for metering and leakages

The EEA note that leakages are a common infrastructure issue throughout Europe. As well as water losses, leakages also risk reductions in quality as pressure in distribution becomes lower (ACTeon et al, 2015).

A further infrastructural challenge for developing Europe’s water sector is the lack of metering infrastructure. This means that there is a lack of incentive to use water efficiently (ACTeon et al, 2015).

4.2.7 Heterogeneous legislative implementation

Water is a regulated sector in the EU, with the Water Framework Directive as the key regulatory instrument (EIB, 2013). The regulation does not require a specific sector organisation, and Member States have different opinions on how best to organise it. The difficulties of determining an optimal sector structure depend on a set of characteristics, listed below (Gee, 2004):

- Water distribution and wastewater collection, i.e. local transport of water to the final consumer and local collection of wastewater, are normally natural monopolies;
- Entry is associated with large sunk costs; fixed costs linked to water distribution represent up to 70% of the supply costs (Gee, 2004);
- Water is difficult and expensive to transport; transport costs per 100km represents approximately 50% of the water wholesale cost (compared to 5% for electricity);
- Water infrastructure is typically unusable for any other purpose and cannot be removed;
- There is a lack of transparency in the sector;
- National, regional and local authorities have traditionally imposed public service obligations on water sector operators due to requirements of health and environment, which is associated with exclusive rights.

The characteristics also explains why liberalisation of the water sector has not brought the same benefits of private sector investment as liberalisation of other network industries.
4.2.8 Stakeholder views of barriers to investment

Three principle barriers emerged from the stakeholder views:

- Regional differences — there are substantial regional disparities across the EU in terms of investment. While in some countries, utilities and institutions have adequate financial capacities, some Member States rely exclusively on EU funds, which disrupts the balance of investment sources through a strong dependence on transfers. In addition, justifying how much money the sector in a Member State needs is quite a lengthy exercise (T. van Gilst, EIB).

- Inefficient pricing — there is a big variation across Member States in how water tariffs are set up. In some cases, tariffs are not a suitable device for revenue raising, as they increase the price and reduce the affordability of water services. However, public authorities and utilities are almost wholly reliant on tariffs as a revenue stream, and a systemic revenue shortfall can also negatively impact their ability to borrow from banks (e.g. the EIB) due to a perceived reliance on transfers. Inefficient pricing remains a considerable constraint to investment and in the long-term leads to systematic underinvestment.

- Political considerations — the water sector is strongly politicised in a number of countries, where strong political and lobbying influences exist, and local utilities can operate as de-facto monopolies. Political interests in the water sector are a major barrier for reform and serve as a barrier to investment as they add considerable uncertainty in the market.

Some additional barriers that have been discussed are:

- A slow-moving market — although the water sector is a large market, is rather slow and very conservative in regard to governance, which presents a considerable barrier, particularly for transformative and innovative technological investment (A. Figeac, INNEON).

- Administrative barriers — for newer Member States such as Bulgaria, which requires large capital investment to construct infrastructure to ensure compliance with the water acquis, there is a lack of administrative capacity for handling the often large and complex tendering procedures that characterise the water sector (D. Simidchiev, Hydrolia).

4.3 Opportunities and instruments

Opportunities exist to develop new instruments and policies to leverage investment in Europe’s water sector. These can be summarised as;

- Extending blended finance
- Offering standardised green bonds
- Policy to encourage downstream measures in water-dependent industries
- Investment in water tech.

Each is discussed in more detail below.

4.3.1 Blended finance

Increasing the extent of blended finance may be a promising way to further leverage contributions from the financial markets towards projects of a more social and environmental profile. Governments may provide risk mitigation to long-term investment projects, where it would result in more appropriate allocation of risks and their associated returns (OECD, 2016; OECD-WWC-Netherlands, 2017).

Globally, PPPs have been particularly common for “Greenfield” projects, such as water and wastewater treatment, desalination and services to industry (World Water Council and OECD, 2015). Increasing private sector participation is also seen by stakeholders as a promising measure to make sector management more efficient.

It may be beneficial for the EU to develop a typology of water sector infrastructure projects based upon their risk and return attributes, to determine the bankability of projects. Economic analysis of the allocation of public finance, including concessional finance, is necessary to improve the efficiency of public spending, and an evaluation is required of where concessional finance might be crowding out private finance instead of catalysing. With analysis that goes beyond cost-benefit analysis of stand-alone projects, the EU would...
design investment pathways that maximise water security over the long term (OECD-WWC-Netherlands, 2017).

4.3.2 Green bonds

The use of green bonds as an investment vehicle has grown rapidly over recent years, and is a viable option for expanding investment in the EU. So far, green bonds have generally been steered toward low carbon investment projects (OECD-WWC-Netherlands, 2017). In the US a standard for water climate bonds, specific to the water sector, has been developed. The standard is intended to provide investors with verifiable science-based criteria for evaluating water-related bonds (OECD, 2016). The EU could develop a similar standard.

4.3.3 Water dependent industries

Water dependent sectors are increasingly aware of the water related risks throughout the value chain. For some sectors these risks have the potential to halt operations. Water security is a topic of increasing importance for large companies. Reporting by the CDP provides insights on the level of exposure to water related risks of companies across the globe. Of the 48 European companies who responded to their information request, 73% reported being exposed to risks in either their direct operations or supply chains.

Sectors at risk of water scarcity include agriculture and power generation. Agriculture accounts for 24% of water abstraction across Europe, but this value can be as high as 80% in some southern Member States. Energy provider E.ON experienced power generation losses of 9% across its hydro-electrical power generators in 2011 following water shortages (CDP, 2014). Overall, 44% of water abstraction is used for energy production in Europe (mainly for cooling coal and nuclear power plants, as well as hydropower).

Firms have incentives to engage in investment that reduce water risks for their own production, but also to reach higher environmental, social, and governance (ESG) factor scores (OECD-WWC-Netherlands, 2017). For large scale implementation, it is important to ensure synergies and complementarities with investment in other sectors (EIB, 2016). A policy to foster sustainable agricultural practises has the potential to bring substantial benefits to the sector in terms of avoided costs of expanding or improving water treatment plants. Such schemes may have modest funding requirements, but can be difficult to fund under a constrained regulatory framework, where such payments may be seen as a transfer of public funds to private farmers (OECD-WWC-Netherlands, 2017). The nexus between water and energy is another area to explore for policy makers, where industry may be interested in investing in more efficient infrastructure, technology, or procedures in the right policy environment (2030 Water Resources Group, 2009).

4.3.4 Reflecting wider water sector needs

Discussions of sector finance tend to focus on upgrading infrastructure, but other areas of resource management, including data collection, weather forecasting, afforestation, land use regulation, conjunctive use of surface and ground water, conservation measures, ecosystem management and pollution control are generally less risky and equally important to emphasise in policy forums (EIB, 2016). Furthermore, EU countries’ urban design increasingly relies on computer tools, inspection robots, and geographical information systems to gain precise knowledge of the state and performance of assets. This information allows better planning of investment in maintenance, and is an area where innovation could attract venture capital if accurately communicated (OECD, 2016). Finally, Europe is a world leader in water technology, including smart and low-cost tech (accounting for over 40% of worldwide patents). Current annual investment in R&D for water technology is €4bn (EIB, 2016).

4.3.5 Lessons from other sectors

There are several lessons from analysis of other infrastructure investment that may provide important insight in the context of water sector investment. One such parallel is the renewable energy sector, where analysis emphasises the need for strong and coherent climate mitigation policies and the facilitation of a conductive investment environment. However, compared to renewable energy, water has a poor record of cost recovery. While energy pricing, in many cases, may be set to inefficient levels, water pricing and adjustment is sometimes completely absent (OECD-WWC-Netherlands, 2017). Water demand is relatively price inelastic, and more effort is required to design pricing instruments for water management (OECD,
For a stronger mobilisation of private sector finance toward water sector investment, a policy to promote efficiency gains, cost reductions, and cost recovery needs to be implemented, as well as improving the balance of tariffs and taxes as source of finance (OECD-WWC-Netherlands, 2017).

4.4.6 Stakeholder views of possible support and solutions

Four principle suggestions for addressing of barriers emerged from the stakeholder consultations:

- **More clarity on requirements** - While EU policy is recognised as a principal and necessary driver for investment, there is room for improvement in the clarity of legal requirements. This echoes back to the complexity of water sector investment and the lack of administrative capacity in multiple Member States, particularly at the local or municipal level.

- **More transparency in the sector** – the water sector, due to its broad scope, complexity, large volume of capital assets and relatively low share of private investment, is often characterised by opaque decision-making and vested interests. This adds uncertainty in the larger business and investment environment, and should be remedied in order to allow for a more efficient sector.

- **More efficient pricing** - the sector can benefit from increased innovation, both in terms of technologies and business models - e.g. applying smart water meter approaches and intelligent pricing mechanisms (A. Figeac, INNEON). By increasing the stability of revenue streams, increased adoption of flat-rate components in tariffs can allow for improved meeting of fixed costs (D. Simidchiev, Hydrolia). In addition, water pricing should in fact reflect the full cost of water - i.e. a fuller application of the polluter/consumer pays principle. Likewise, new models are also needed in the context of circular economy, in order to reduce consumption and increase re-use rates. A potential measure here could be to increase the recycling rates of water for manufactured companies at the margin (A. Figeac, INNEON).

Additional suggestions that have been proposed are:

- **Strengthening control at all levels** - this measure is crucial for achieving compliance with EU policy and for overcoming pressing constrains for the sector such as local monopolies and lobbying and consequently for increasing efficiency in the sector (D. Simidchiev, Hydrolia).

- **Wide involvement of stakeholders** – toward developing comprehensive solutions and incentives for all market players (A. Figeac, INNEON). Another measure related to this would be the increased involvement at the national level of stakeholders such as water associations, and local players such as utilities and municipalities, in order to more comprehensively evaluate local needs in order to prioritise investment toward more efficient outcomes (T. van Gilst, EIB).

4.4 The socioeconomic impact of meeting Europe’s water investment needs

Improving access to, and the quality of, water supply has a broad range of socioeconomic benefits; the stimulus of the investment creates jobs and economic opportunities, while the provision of water improves human health, increases productivity and facilitates levels and types of socioeconomic activity that would not be possible without such a supply.

The particular focus of our analysis in the water treatment and supply sector has been to better understand the first of these effects; the socioeconomic impact of the initial investment.

E3ME has been used to model the macroeconomic impacts of a scenario in which Europe’s water sector investment needs are addressed fully. The scenario developed uses an EIB (2016) estimate of the investment gap in the EU: namely, additional annual investment of 2016EUR 58bn, compared to recent historical expenditure, over the period 2019 to 2025. Investment is financed through borrowing at the national and EU level, which is repaid by Member States in the following 20 years.

This exercise focuses on assessing the benefits and costs of increased investment in water infrastructure. The principal effect in the short term is a significant investment stimulus across the Union. Investment in 2025 is 2018EUR 77.7bn higher than baseline. The investment stimulus results in a positive GDP difference of 0.48% to baseline, and 0.45m increase in EU wide employment in 2025. The increase in economic activity from 2019 to 2025 is greater than the investment stimulus itself, given indirect and induced effects of the water sector investment. The impact of these multiplier effects is shown in Table 3: both wider investment
in the economy and consumer expenditure are greater than baseline values. There is some leakage of the effect, shown in negative changes to net exports.

After the investment period, the cost of the investment reduces growth in economic activity, compared to the baseline. Figure 4.1 shows the negative GDP results, compared to the baseline, throughout the period of repayment. Increased direct taxation in the scenario reduces personal disposable income, reducing consumer expenditure, see Table 4.1. Following full repayment of the debt, EU wide economic growth recovers. From 2046 onwards, EU GDP is at a level very similar to baseline.

Given that EU-level resources are focused on lower-income Member States, the water investment has redistributive effects, reducing economic disparities. The lower-income Member States benefit from significant investment stimuli, but pay only part of the subsequent cost.

Table 4.1 Water investment scenario: EU28 disaggregate GDP results differences to baseline (2018EUR bn)

<table>
<thead>
<tr>
<th></th>
<th>2020</th>
<th>2025</th>
<th>2030</th>
<th>2035</th>
<th>2040</th>
<th>2045</th>
<th>2050</th>
</tr>
</thead>
<tbody>
<tr>
<td>GDP</td>
<td>87.5</td>
<td>81.8</td>
<td>-28.1</td>
<td>-25.5</td>
<td>-19.0</td>
<td>-26.4</td>
<td>-7.0</td>
</tr>
<tr>
<td>Investment</td>
<td>82.0</td>
<td>77.7</td>
<td>-9.7</td>
<td>-3.9</td>
<td>-2.9</td>
<td>-7.6</td>
<td>1.3</td>
</tr>
<tr>
<td>Consumption</td>
<td>15.9</td>
<td>14.6</td>
<td>-20.4</td>
<td>-22.9</td>
<td>-17.7</td>
<td>-17.3</td>
<td>-7.5</td>
</tr>
<tr>
<td>Expenditure</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Net Exports</td>
<td>-10.4</td>
<td>-10.5</td>
<td>2.0</td>
<td>1.3</td>
<td>1.6</td>
<td>-1.5</td>
<td>-0.8</td>
</tr>
</tbody>
</table>

Figure 4.1 Water investment scenario: EU28 GDP and employment differences to baseline

It should be noted that this exercise does not capture some key effects of closing the investment gap, namely water security, resilience, and productivity effects. The investment will improve water security, avoiding disruptions to industry and to power generation. Investment will increase resilience to flooding,
reducing economic and human costs of natural disasters. The EIB’s analysis of the ‘Investment Plan for Europe’ shows a direct investment impact, as does this analysis. The ‘structural impact’ of water investment cannot be fully captured, however: this would require an estimate of the productivity and disaster costs of inadequate water infrastructure which are implicit in the baseline projections and are avoided through investment modelled.

In addition, there are substantial environmental and other benefits to improving access to, and quality of, clean water for use both in the agricultural sector and for human use/consumption. The extent (and specific impacts) of these depend upon the aims of individual policies; for example, the Drinking Water Directive primarily aims at improving the quality of water available to households, so impacts in terms of reduced rates of eutrophication, for example, are small and only arise indirectly as a result of the policy. However, improvements in human health, from improved access to high quality drinking water, can be substantial. There are also indirect environmental benefits from policy; again taking the Drinking Water Directive as an example, the Impact Assessment states that the directive will also reduce demand for bottled water, which can serve to reduce plastic production and all of the associated environmental damages.

4.4.1 The role for resource efficiency

The analysis above assumes that the water investment needs gap requires complete bridging. An alternative for policy would be to improve water efficiency levels (both in households and industry), with the aim of reducing future levels of water demand, and thereby reducing the required investment.

Such policies (which would be enacted at the sector-level, and so are beyond the scope of this analysis) would reduce the socioeconomic impact seen above, although it should be noted that they would do so without any impact on the wider benefits of improved water supply.

Previous analysis has highlighted the potential positive socioeconomic impact of improved resource efficiency (Cambridge Econometrics, 2014). By reducing costs faced by consumers and industry, it is possible to generate additional economic activity elsewhere in the economy (for example in consumer services) which is greater than the activity lost in the resource-intensive sector (water treatment and supply, in this case).

4.4.2 Developing a water technology industry

As outlined at the start of this study, an analysis of water supply typically focuses on the wider benefits that water brings, rather than the direct socio-economic benefits, in terms of jobs and output, associated with the extraction, reuse and supply of water. The industry is a relatively small part of the European economy (water supply contributed 0.26% of total EU28-wide value added in 2016) and is almost exclusively domestically focussed. However, it is clear that efficiency of water use is going to become an increasingly major issue globally, as climate change leads to increased scarcity. This presents an opportunity to Europe; if Europe can develop a competitive industry in water technology, improving the efficiency of water supply operations and ensuring the efficient use of water, then this can present an export opportunity to improve the trade balance of Europe vis-à-vis the rest of the world.

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138 EIB (2016) cites possible damages of EUR 5.5bn and EUR 23bn by 2050, depending on climate and economic changes. It is not clear that the EIB finance requirement for risk management and resilience would be sufficient to prevent the full effects of these potential damages.

139 http://www.eib.org/en/about/key_figures/eib-impacts/index.htm

Appendix A: E3ME macroeconomic methodology

A.1 Specification of E3ME scenario

The scenario constructed in E3ME is formulated to model the effect of the EU28 fully addressing its short-medium term water investment gap, over the period 2019 to 2025. The investment gap value is taken from EIB (2016): summing the EUR 13bn investment gap in ‘water security, including flood risk management’ and EUR 45bn ‘compliance and rehabilitation of Europe’s water infrastructure’. The EUR 13bn is allocated to Member States on a per capita basis, in the absence of any better data. The EUR 45bn is allocated using EC data on per capita investment in urban waste water collecting system and treatment plants. EIB (2016) calculates an investment requirement across the EU’s water infrastructure network: investment requirements across both water treatment and supply, and sewerage. Given no disaggregation by NACE sector in the EIB (2016) report, investment is allocated proportional to baseline forecast investment in the respective sectors.

Source of funding is a key question in realising the closure of the investment gap. In this modelling exercise, investments are funded through national and EU-level resources. Finance of these resources is modelled in a stylized manner. National and EU debt accumulates from 2019 to 2025, during the investment phase, and is repaid in the following 20 years. EU financing is funded by Member States, payments made proportional to contributions to the EU budget. To ensure policy fiscal neutrality with the baseline, Member States increase taxation to fund their national investments and contributions to EU resources.

In this scenario, 25% of investment is met by EU level resources. This figure is chosen as a representative value of EU co-funding in water projects; planned EU related co-funding represents 25% of the total investment needs to meet the Urban Waste Water Treatment Directive (EC 2016). Results of modelling are presented at the EU28 level, which minimises the impact of EU vs national level funding. EU level resources are a stylized version of the ERDF and Cohesion Fund (CF). Distribution of ERDF and CF finance is allocated across Member States using EC data of planned European Structural and Investment Funds from the ERDF and CF; this data reflects Member States access to EU level financing.

In this formulation, the cost of water investment is borne across the Union, richer Member States generally contributing a surplus to EU level finance. All Member States currently eligible for the CF receive a net subsidy from EU level finance. The only Member State receiving a surplus, which is not eligible for the CF, is Spain.

A.2 Assumptions

The key modelling assumption in this scenario is crowding out in capital markets. The scenario construction does not model any crowding out of government or private investment at the whole economy level, following standard assumptions in E3ME modelling. Under crowding out, expansion of economic activity during the investment phase would be substantially lower; the magnitude being dependent on the size of the crowding out. It should be noted, however, that E3ME does model capacity constraints in the labour market and product markets. This issue is explored in more detail in another of the Annex reports to this study, Analysis of environmental policy in the 7th EAP and potential challenges to the successful implementation of environmental policy in the future.

As stated above, modelling imposes policy fiscal neutrality with the baseline; that is, the cost of the water infrastructure is fully paid over the forecast period, without increasing government debt. Revenue is raised through an increase in income taxation, which is chosen given its broad tax base, in order to minimise distortions. Current government expenditure is fixed across scenarios, as seen in Table 3.

---

142 AMECO nominal long term interest rates are used for cost of borrowing.
143 http://ec.europa.eu/budget/figures/interactive/index_en.cfm
145 Cohesion Fund eligible Member States include: EU13, Greece, and Portugal.
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Mooney, A (2016) Pension funds crave more infrastructure projects https://www.ft.com/content/a05fe960-95ec-11e6-a1dc-bdf38d484582 Accessed 12 September, 2017


Links between production, the environment and environmental policy

Sewerage and waste management – sector study
This report is an Annex to the Final Report *Links between production, the environment and environmental policy*, ordered and paid for by the European Commission, Directorate-General for Environment, Contract ENV.F.1/FRA/2014/0063. The information and views set out in this study are those of the authors and do not necessarily reflect the official opinion of the Commission. The Commission does not guarantee the accuracy of the data included in this study. Neither the Commission nor any person acting on the Commission’s behalf may be held responsible for the use which may be made of the information contained therein.

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1 Introduction

The sewerage & waste management sector is an environmental services sector with a heavy dependence on ecosystem services. The sector has strong links to other sectors, which are dependent on water as a resource. There are significant environmental policy challenges relating to wastes and emissions.

The waste management sector has clear links to several of the Sustainable Development Goals and targets contained within them. Regarding Goal 6: clean water and sanitation, the sector has a role to play in helping to improve water quality by reducing pollution, eliminating dumping, minimising release of hazardous chemicals and materials, and substantially increasing recycling. Regarding Goal 11: sustainable cities & communities, the waste management sector is crucial to the target of reducing the adverse per capita environmental impact of cities, including addressing municipal and other waste management. Towards Goal 12: responsible consumption and production, the sector’s role includes contributing to the sustainable management and efficient use of natural resources, significantly reducing food waste, achieving the environmentally sound management of chemicals and wastes and significantly reducing their release to air, water and soil, and substantially reducing waste generation. Regarding Goal 14: life below water, the waste management sector will contribute to preventing and significantly reducing marine pollution from land-based activities, including marine litter. The sector is crucially relevant to the 7th EAP regarding waste management to increase resource efficiency\textsuperscript{146} and regarding water management\textsuperscript{147}.

Circular economy policy opportunities exist in this sector in the recovery of materials and biomass; again, using waste as a resource. The recovery of material is related to the three manufacturing sectors under analysis in the project, given that all produce waste in production processes and in the consumption of their products.

- **Waste collection:**
  3. Extend collection of organic waste from municipal solid waste stream to contribute to maximising recovery of biomass
  4. Extend capture of organic waste from waste water treatment plants to contribute to maximising recovery of biomass
  5. Adapt waste water treatment plants to capture (more) microplastics to reduce this type of marine litter
- **Regulation:**
  6. Increase recycling targets to drive separate collection to aid material recovery
  7. Introduce additional bans on landfilling and incinerating certain types of waste
  8. Improve minimum requirements for certain waste streams to facilitate recycling and reuse
- **Processing:**
  9. Increase use of optical sorting machines to optimise sorting processes
  10. Develop measures to support the interface between chemical, product and waste legislation (i.e. innovative tracing technologies, to improve information flows)

\textsuperscript{146} 39, 40, 43(d), & 43(viii) of the Annex of the 7th EAP.

\textsuperscript{147} See points 8, 17, 28(vii), 41, 43(ix) of the Annex of the 7th EAP.
2 The environmental footprint of the sector

2.1 Research questions

- How do these sectors benefit from the environment?
- How do they affect the environment?
- What is their resource use?

2.2 Waste Management

Each year, 2.7 billion tonnes of waste are produced in the EU, of which 98 million tonnes (4%) are hazardous. In 2011, per capita municipal waste generation averaged 503kg throughout the EU, but ranges between 298 and 718kg across individual Member States.

The collection and transportation of waste from the point of generation to the point of reuse or treatment may generate significant greenhouse gas and NOx emissions, as well as resulting in significant fossil resource depletion and traffic/congestion (EC, 2016a).

On average, only 40% of solid waste is prepared for re-use or recycled, although some individual Member States achieve a rate of 70%, demonstrating how waste could be used as one of the EU’s key resources. However, many Member States still landfill over 75% of their municipal waste (EC, 2013).

Figure 2.1 EU municipal waste management

![Graph showing EU municipal waste management]

Source: EC, 2018a

2.2.1 Recycling

Environmental impacts associated with the reprocessing operation and transportation of materials to the reprocessing facility. GHG emissions come from the carbon dioxide associated with electricity consumption for the operation of material recovery and sorting facilities (see e.g. EC, 2016a). The key environmental advantages of recycling and recovery are reduced quantities of virgin material use and disposed waste, and the return of materials to the economy. For example, it takes 5 tonnes of bauxite ore and 32 barrels of oil to make a tonne of aluminium (Upstream, 2015). Recycled aluminium generates energy and air pollution impacts 75-90% lower than virgin aluminium whilst also avoiding most of the resource depletion associated with aluminium ore extraction. Production of recycled glass uses around 20-30% less energy than virgin glass (EC, 2016a). Recycling of high quality paper products e.g. office paper can result in significant environmental benefits (WRAP, 2010), and recycling rates of over 80% are not uncommon in EU Member States with EPR schemes for graphic paper (e.g. in FI, NL and SE) (Bio by Deloitte, 2014). Recycling and incineration of paper with the latest energy recovery efficiencies are broadly comparable in terms of...
climate change impacts (WRAP, 2010), but the use of recycled rather than virgin paper pulp can help to prevent land use change and indirect emissions from the loss of carbon stored in forests (James, 2013). It takes one gallon of used industrial oil to produce 0.25 gallons of new high-quality lubricating oil (as opposed to 42 gallons of crude oil required to make the same quantity, albeit alongside a range of other products) (US EPA, 2015); again recycling rates of over 80% are achievable if EPR schemes are in place (e.g. in BE, FI, IT, DE and PT) (Bio by Deloitte, 2014). For plastics, mechanical recycling is the best waste management option in respect of reducing the depletion of natural resources (WRAP, 2010). Recycled rubber from end-of-life tyres can be processed into rubber shred and crumb of various particulate sizes which can then be incorporated into a wide range of products (e.g. paving, roof tiling, mulch, vehicle parts, new tyres, road surfaces, carpet underlay, footwear etc.) (WRAP, no date), reducing the need for various raw materials.

Table 2.1 GHG emissions avoided per tonne of different types of waste avoided or recycled

<table>
<thead>
<tr>
<th></th>
<th>Glass</th>
<th>Board</th>
<th>Wrapping paper</th>
<th>Dense plastic</th>
<th>Plastic film</th>
<th>Metals</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoided</td>
<td>Kg</td>
<td>CO₂e/t</td>
<td>Kg</td>
<td>Kg</td>
<td>Kg</td>
<td>Kg</td>
</tr>
<tr>
<td>920</td>
<td>1,600</td>
<td>1,510</td>
<td>3,320</td>
<td>2,630</td>
<td>12,000</td>
<td></td>
</tr>
<tr>
<td>Recycled</td>
<td>Kg</td>
<td>CO₂e/t</td>
<td>Kg</td>
<td>Kg</td>
<td>Kg</td>
<td>Kg</td>
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<tr>
<td>390</td>
<td>1,080</td>
<td>990</td>
<td>1,200</td>
<td>1,080</td>
<td>3,300</td>
<td></td>
</tr>
</tbody>
</table>


2.2.2 Reuse

Re-using products may result in an overall offset in environmental effects due to a lower requirement for manufacture of replacement products, thereby reducing raw material inputs, water and energy use associated with manufacture, and the transportation of new products to the point of sale. Design for reparability can facilitate product reuse. Reuse can potentially save almost 100% of materials, energy and water, whilst remanufacturing can save 85% and 80% for materials and energy respectively used in manufacturing (Steinhilper, 2006). However, product re-use is also associated with environmental impacts arising from transport and collection, as well as product cleaning operations. Some disassembly operations may require significant electricity demand, and such operations must also be carefully controlled to minimise leakage of hazardous substances such as refrigerants, oils or PCBs (EC, 2016a).

2.2.3 Landfill

The environmental impact of landfilling depends to a large extent on the design and management of the landfill and the type of waste landfilled; well-lined and capped landfills, with landfill gas captured and used for energy generation, or landfills for inert waste, typically have the lowest environmental impacts (EC, 2016a). Approximately 120 m³ of biogas (60% of which is methane, with a global warming potential 25 times that of CO₂) is produced per tonne of MSW (fresh weight) landfilled (Obersteiner et al., 2007), and emissions can be anywhere in the range of 158 to 1,285 kg CO₂e per tonne of MSW deposited, depending on the landfill’s design (EC, 2016a). If MSW is mechanically and biologically treated (MBT) before landfilling, gas production can be reduced by approximately 95 % (JRC, 2006). Whilst landfill leachate is typically collected, removed and treated, in lower-standard landfills it may leak into soils or water courses, causing contamination.

2.2.4 Incineration

Incineration, the combustion of waste at high temperatures for a sustained period, leads to a very substantial reduction in the volume of waste and effectively destroys pathogenic biological organisms. By-products of the combustion process principally comprise emissions to the atmosphere (which can, in high-standard incinerators, be largely captured e.g. by catalytic processes) and residual ash, which can

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148 Including: sulphur dioxide (SO₂), nitrogen oxide and Nitrogen Dioxide (NO and NO₂), hydrogen chloride (HCl), hydrogen fluoride (HF), gaseous and vaporous organic substances, as Total Organic Carbon (TOC), carbon monoxide (CO), dust, heavy metals, polychlorinated dibenzo-p-dioxins and -furans (PCDD/F) (EC, 2016).
account for something in the region of 20-30% of the weight, and 10% of the volume, of inputted MSW. Many incinerators are now classed as waste-to-energy facilities, since waste combustion can be used to generate heat and energy. The energy recovery efficiency of incineration plants can vary considerably, from anywhere between 14-90% thermal efficiency (with the higher rates achievable e.g. where generated heat is used directly for district heating) (EC, 2016a). It should however be noted that with regards to the waste hierarchy, waste prevention, reuse and material recycling are all deemed preferable to energy recovery.

2.2.5 Composting/anaerobic digestion
Like landfilling, composting of organic waste generates CH$_4$, NH$_3$ and N$_2$O emissions and nutrient leaching, although these may be partially compensated by the fertiliser replacement and soil improver (humus) properties of compost (EC, 2016). Anaerobic digestion (AD) can be an efficient option to recycle nutrients and recover energy from organic wastes, but overall environmental impacts also depend on emission rates of CH$_4$ and NH$_3$ and methods to store and apply digestate. Larger, centralized AD plants can be more efficient, whilst emissions may be high from small plants (EC, 2016a).

2.3 Sewerage
Sewage, or municipal wastewater, contains nitrogen and phosphorus from human waste, food and soaps and detergents. On a global level, nutrients recovered from these waste streams could contribute approximately 2.7 times the nutrients currently contained in chemical fertilizers. Increasing urbanization trends are leading to growing nutrient concentration in solid waste and sewerage as sewage sludge (EMF, 2017).

Sewage sludge, the by-product of waste water treatment processes, is mainly disposed of through incineration, landfilling and application to land. Sewage sludge is composed of six categories of components, some of which can lead to environmental impacts at the final disposal stage: non-toxic organic carbon compounds; nitrogen- and phosphorous-containing components; toxic inorganic and organic pollutants (e.g. dioxins, pesticides, polychlorinated biphenyls etc.); pathogens and other microbial pollutants; inorganic compounds (e.g. silicates, aluminates, calcium- and magnesium-containing compounds); water (Sathir et al., 2017).

In 2014, 8.7 million tonnes of dry solid matter of sludge were produced in the EU, representing approximately 17 kg per inhabitant (EC, 2017). Of the total sewage sludge produced, 58% was reused, mostly in agriculture. Some countries showed ratios below 10kg, suggesting low levels of collection and treatment – Italy, Cyprus, Portugal, Bulgaria and Romania.

In the EU, 70% of the phosphorous concentrated in sewage sludge and solid waste is not recovered, suggesting untapped potential for the recovery of nutrients which could be looped back into the soils rather than being discharged. A way to recover nutrients from wastewater is to produce concentrated NPK fertilisers. Estimates indicate that capturing all nutrients from excreted waste in household sewage could lead to the recovery of 30 million tonnes of nitrogen, 5 million tonnes of phosphorous and 12 million tonnes of potassium – a third of global fertilizer demand (EMF, 2017).

2.3.1 Recycling sewage sludge for agriculture
The practice of applying sewage sludge to agricultural land is common in Europe, as a way to utilize nutrients such as phosphorous and nitrogen and organic substances for soil improvement (Milieu et al). In addition, this would reduce the use of synthetic fertilizers which bring in nutrients from non-renewable sources (EMF, 2017). Despite the benefits that such practice can deliver for agriculture, the application of sewage sludge as a fertilizer on land can have negative implications for the environment: decrease in soil value, ecosystems degradation and decrease in groundwater quality. This is due to the volatilization of pollutants to air, the release of pollutants to surface water and to soil (Milieu et al, 2010).

In addition to the emission of pollutants, the use of sewage sludge as fertilizer for land represents an important source of primary microplastic pollution in soil. Sewage sludge is generally contaminated with micro fibers and microplastics, as these are not filtered out from waste water treatment plants. The application of sewage sludge to land is common in Europe and North America – approximately 50% of
sewage sludge processed for agricultural use. Annual additions of microplastics to land in Europe are estimated between 125 and 850 tonnes per million inhabitants (Nizzetto et al., 2017).

The reuse of wastewater is considered a reliable source of water supply, due to its independence from seasonal variabilities and to the over-abstraction of water as a major water stress. Wastewater reuse can bring a number of economic, environmental and social benefits, for instance in its application to farming (e.g. irrigation).

2.3.2 Wastewater reuse for irrigation
In addition to providing a reliable water resource, the reuse of wastewater reduces wastewater disposal, contributing to the preservation of water quality downstream. Thanks to the nutrients contained in wastewater, reusing it for irrigation reduces the need for fertilizers. However, irrigation through wastewater requires appropriate treatment. Potential environmental impacts include: groundwater contamination due to heavy metals, nitrate and organic matter; soil contamination due to salt accumulation and acidifications; crop contamination due to potential toxic substances in wastewater. The reuse of wastewater differs significantly across Europe as the volume of water for irrigation depends on climate, crop type, irrigation method and on the area to be irrigated (EC, 2013). Currently, annual reuse of treated urban wastewater is approximately 1 billion cubic meters, well below the EU potential, estimated at 6 billion cubic meters. Initiatives for the reuse of wastewater for irrigation are present in southern Member States such as Spain, Italy, Greece, Malta and Cyprus, as well as in some northern Member states such as Germany and Belgium (EC, 2018b).

Wastewater treatment follows a number of steps which require energy: solid waste removal, biological digestion, disinfection and discharge. Energy recovery from sewage can offset the energy required for its treatment. It is estimated that the energy embedded in wastewater is 14 times higher than what is needed for treatment. The European Commission estimated that if all organic waste was turned into energy, 2% of the EU’s renewable energy target could be met (EMF, 2017).

2.3.3 Sewage sludge to energy
The growing amount of sewage sludge produced has led to the introduction of technologies to produce energy from sewage sludge. Dried sewage sludge is an attractive source of energy. Anaerobic digestion (AD) is one of the most common methods of sludge treatment and energy recovery. AD generates biogas which can be used as fuel.

In 2013, Thames Water saved £15 million on its energy bills through the generation of 14% of energy demand from sewage sludge (EMF, 2017).

2.4 Environmental impacts of anaerobic digestion technologies for treating the organic fraction of municipal solid waste (OFMSW)
Anaerobic digestion (AD) of biomass has become a common technology used in many European countries. The main advantage of AD is it generates renewable energy, but it also manages and treats organic waste, and recycles key nutrients to soil, making it an attractive technology to public policy makers (Edwards, 2015).

Research has shown that capture of and energy recovery from bio-methane may considerably contribute to Greenhouse Gases (GHG) emission reductions. Anaerobic digestion (AD) is a process for decomposition of organic matter by anaerobic bacteria under lack of oxygen. There are a number of different techniques distinguished on the basis of the operating temperature (i.e. thermophilic or mesophilic plants) and the percentage of dry matter in the feedstock (i.e. dry systems with 30-40% dry matter and wet systems with 10-25% dry matter). AD encompasses a broad family of processes which could be classified according to their feedstock input mode into (i.e. batch or continuous processes) and the geometry of the main treatment unit (i.e. vertical or horizontal units). A study by Karagiannidis and Perkoulidis (2008) evaluated different anaerobic digestion technologies by applying a multi-criteria decision support method. The research compared the following commercial AD processes:
- **Waasa**: a vertical digester, internally separated for the predigestion of the input material. Used in order to digest waste with 10–15% volatile solids (VS) content and has been tested on a number of wastes as mechanically or source-separated MSW, sewage sludge, slaughterhouse waste, fish waste and animal manure.

- **Valorga**: this process uses a vertical digester with biogas recirculation (internally, within the digester) and typically operates with a 25–32% VS-content. It was initially designed to treat organic fraction of MSW and was later adapted to treating mixed MSW.

- **Dranco**: the Dry Anaerobic Composting process is a thermophilic, high-solids, single-stage technology with no biogas recirculation and a 15–40% VS-content. It is a pure dry-process for treatment of the OFMSW, which requires high total solids (TS) content in the reactor in order to have optimal performance.

- **Kompogas**: this is a high VS-content digester with no gas recirculation, operating in the thermophilic range. The reactor is a horizontal cylinder with a stirrer, which provides some mixing of the waste.

- **BTA**: this is a multi-stage, low-solids system for treating mixed MSW or source-separated OFMSW. BTA is a wet AD process, combining waste pre-treatment and separation stages in a fully enclosed and highly automated facility.

The comparison of the AD processes was performed based on the following criteria: GHG emissions, energy recovered, material recovered and operating cost, with performances of the above five AD processes calculated for a standard plant capacity of 20kt (input material)/year. The results of the comparison are presented below:

*Table 2.2 Performances of AD processes*

<table>
<thead>
<tr>
<th>Process</th>
<th>GHG emitted (kg CO₂-eq/t)</th>
<th>Recovered energy (kWh/t)</th>
<th>Recovered materials (kg/t)</th>
<th>Operating cost (€/t)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waasa</td>
<td>216</td>
<td>730</td>
<td>300</td>
<td>90</td>
</tr>
<tr>
<td>Valorga</td>
<td>228</td>
<td>700</td>
<td>320</td>
<td>68</td>
</tr>
<tr>
<td>Dranco</td>
<td>226</td>
<td>760</td>
<td>260</td>
<td>62</td>
</tr>
<tr>
<td>Kompogas</td>
<td>208</td>
<td>585</td>
<td>250</td>
<td>63</td>
</tr>
<tr>
<td>BTA</td>
<td>212</td>
<td>700</td>
<td>280</td>
<td>95</td>
</tr>
</tbody>
</table>

*Source: Karagiannidis and Perkoulidis (2008)*

The study applied Electre Multi-Criteria Decision-Making (MCDA) method in order to consider different preferences of various possible decision makers towards the criteria. The Dranco process was ranked in the best position at of all cases, while the methods most frequently ranked in the 2nd best position were Kompogas and Waasa. Dranco process combines the relative advantages of low-cost and high energy recovery, while BTA’s high cost seems to be critical in determining it’s position at the bottom of the list in this analysis.

The performance of the Dranco process was reviewed by Murphy and Power (2006) depending on the plant scale: 10 kt/a or about 50,000 persons (a small city); 30 kt/a or about 145,000 persons (a medium sized city); 50 kt/a or about 242,000 persons (a small waste strategy region); and 100 kt/a or about 485,000 persons (a medium sized waste strategy region).

*Table 2.3 Technical/economic data for the DRANCO process*

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>5kt/a</th>
<th>10kt/a</th>
<th>25kt/a</th>
<th>50kt/a</th>
<th>100kt/a</th>
</tr>
</thead>
<tbody>
<tr>
<td>Investment price</td>
<td>€ mil.</td>
<td>9</td>
<td>12</td>
<td>15</td>
<td>19</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>€/t/a</td>
<td>1,800</td>
<td>1,200</td>
<td>600</td>
<td>380</td>
<td>250</td>
</tr>
</tbody>
</table>
Despite the availability of published LCAs of AD treatment, it is difficult to draw conclusions on the comparison between different treatment technologies and scenarios because studies differ in the assumptions made on system boundaries and methodology for evaluating life cycle inventory burdens. In addition, most of the studies that tackle LCA of landfill treatments focus on general MSW or report insufficient details to enable a comparison with anaerobic digestion (Evangelisti et al., 2013). This study compared the environmental impacts of AD with energy and organic fertiliser production against two alternative approaches: incineration with energy production by CHP and landfill with electricity production. The AD plant in comparison was corresponding to a medium sized plant with capacity of 35 kt/year, with life cycle inventory data specific to the Greater London area, UK.

The results of the study show that AD is the most favourable alternative in terms of global warming potential, with a total of -2,300 tons of CO2 eq. per functional unit (35 kt/year), compared with an impact of +30,000 and -1,500 tons of CO2 eq. for the landfill and incineration scenarios, respectively. If calculated per ton of OFMSW, the GWP impacts would translate as -65, 843 and -42 kg CO2eq./ton for AD, landfill and incineration respectively. The negative values for AD and incineration scenarios indicate reduced emissions, arising from the avoided burdens of energy production and, for AD, inorganic fertiliser production. Looking at the acidification potential impact category, AD is again the most favourable option, even if the difference from the landfill scenario is smaller than for GWP (-0.6 compared with 3.7 tons of SO2 eq.). The comparison changes when the focus is on photochemical ozone creation potential (POCP) and nutrient enrichment (NE). For these two indicators, AD turns out to be the second option (3.63 tons of ethene eq. and 108 tons of NO3 eq.), while incineration appears to be the most environmentally friendly solution (0.25 tons of ethene eq. and 22.6 tons of NO3 eq.). It should be noted, however, that the results of the study could vary significantly with respect to some key parameters. Fugitive emissions of methane during the production of biogas are highly uncertain and they have a large impact on the environmental contribution of anaerobic digestion. This can influence the final ranking of the different treatment options, to the point of making the biological process appear worse than incineration in terms of GWP. In addition, different assumptions on the marginal technology for electricity production can change the ranking amongst the different process options in terms of AP, with the AD scenario shifting from being the best to the worst. The presented results are valid in case the average UK electricity mix is substituted by electricity from biogas.

![Parameter Table](image)

**Source:** J. Murphy, N. Power (2006)
As a conclusion, using life cycle thinking will enable decision makers to assess AD performance in an environmental and financial context. It could be suggested that in order to better disseminate the benefits of AD to policy makers, the use and refining of multi-criteria decision support tools based on lifecycle analysis and life cycle costing should be applied, considering the specific circumstances of a particular region or a country. (Edwards, 2015).

2.5 The socioeconomic footprint of the sector

According to the NACE rev. 2 statistical classification of economic activities, waste management includes waste collection, treatment and disposal activities as well as materials recovery. Overall, the waste management sector has witnessed substantial growth. Employment increased from 0.8 million (full-time equivalents) in 2000, to 1.1 million in 2014 (Weghmann, 2017; Eurostat\(^\text{149}\)). Among the activities in the sector, waste collection is associated with the highest labour intensity and, together with treatment activities, employment in these stages of waste management is dominated by bigger companies. On the contrary, material recovery remains in the hands of smaller companies (EC, 2016a).

The employment potential of the waste management sector is expected to grow further in the future. In particular, a stronger focus on re-use and recycling triggered by the adoption of circular economy principles could bring a significant benefit in terms of additional employment, due to the labour-intensity of re-use and recycling activities compared to disposal (EC, 2016a). Current legislation and stricter targets for recycling also contribute to increasing this potential. It is estimated that by 2020 an additional 400,000 jobs could be created in the waste management industry as a result of full compliance with EU waste policy and an additional annual turnover of €42 billion (EC, 2016b).

As far as recycling is concerned, the enforcement of higher recycling rates in EU Member States is expected to lead to the creation of 50,000 new direct jobs throughout the process’ value chain by 2020. Direct jobs are found in the sorting and separation of materials, collection and recycling. These mainly, but not exclusively, require low-skilled workers, therefore presenting the potential of contributing to social inclusion and poverty alleviation. In addition, an estimated 75,000 new indirect jobs could be created in recycling activities linked to construction and maintenance of facilities, research and innovation, administration and management (PRE, 2016; EEA, 2011). These number are expected to increase further by 2025, reaching 80,000 and 120,000 new direct and indirect jobs respectively (PRE, 2016).

### 2.6 Trends in environmental pollutant releases

Many sectors of the economy use and/or create pollutants during production processes, with the potential to damage human health and the wider natural environment. Data on pollutant releases is published in the European Pollutant Release and Transfer Register (E-PRTR\(^{151}\)). Data is classified by NACE code of the emitter, pollutant, location (including country), year, release medium and volume, and as such it is possible to track its evolution over time.

In the analysis below, we present data in volume terms, but also environmental impacts. These are calculated by applying coefficients reflecting the toxicity of different pollutants, taken from ReCiPe2016 LCIA, according to whether they were released via air, water or land. This allows the summation of different

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\(^{150}\) [http://ec.europa.eu/eurostat/web/products-datasets/-/sbs_sc_ind_r2](http://ec.europa.eu/eurostat/web/products-datasets/-/sbs_sc_ind_r2)

\(^{151}\) [http://prtr.eea.europa.eu/#/home](http://prtr.eea.europa.eu/#/home)
pollutant based upon the impact that they have on human health (measured in disability-adjusted life years, DALYs) and ecosystem health (measured as disappeared species per year, species.years).

*Figure 2.3 The health impacts of pollutant releases from the waste management & sewerage sector*

Source: Author calculation, using data from E-PRTR and ReCiPe2016 LCIA.

In the sewerage and waste management sector, 25% of all recorded pollutant releases included all data required to estimate environmental impacts; this proportion was consistent over time. The environmental impacts are also relatively steady over time, with the exception of a substantial spike in 2012, when environmental impacts more than doubled (see Figure 2.3). This reflects a large release of CO2 from the UK in this year (see ), as well as a (substantially smaller) accompanying increase in releases of carbon monoxide. Both releases were associated with energy use by the sector.

The impacts upon human and ecosystem health from sewerage and waste management pollutant releases are an order of magnitude greater than in any other sector studied within this analysis. Human health impacts spiked in 2012, at 182,734 DALYs; by comparison, the highest human health impact seen in other sectors is in the food, drink & tobacco sector in the same year, and reaches only 10,146 DALYs.
Figure 2.4 CO2 releases from the sewerage and waste management sector
3  Current trajectory – direction of travel

3.1  Research questions

- How does environmental policy affect the links to the environment (i.e. what has been the impact of policy so far?)
- How does the environment and environmental policy affect the links between these sectors and growth, jobs and investment?
- The links between production in these sectors and consumption including consumption orientated policy tools? (i.e. to what extent does demand-side policy affect the sector)
- The evolution in consumers and investors demand for increased transparency on environmental performance? (are consumers demanding water with less environmental impact?)
- Is there a clear picture of what ‘sustainability’ would mean for these sectors, how it has evolved and whether the sectors are actually moving towards it?
- What has driven changes over time (post-2000), and what changes are expected to occur in the future?
- How have the answers to all of these questions changed over time, and how are they forecast (modelled) to change in the future?
- How do SMEs differ from other firms in their answers to all of these questions?
- Are there examples of striking differences to any of these questions between Member States (and if so, why)?

3.2  The impact of policy

Over the last 20 years, a large variety of policies have been introduced in the EU with the aim of increasing recycling. These include targets for the waste management of a variety of materials and products: electrical and electronic equipment, end of life vehicles, packaging, batteries, household waste, and construction and demolition waste (EEA, 2011).

More recently, in 2015, the European Commission adopted the Circular Economy Action Plan (COM/2015/0614 final), including a proposed revision of legislation on waste to support the transition towards a circular economy. These include a range of revised waste management targets put forward in the Waste Framework Directive (2008/98/EC), the Landfill Directive (1999/31/EC) and the Packaging and Packaging Waste Directive (94/62/EC), summarised in the table below.

<table>
<thead>
<tr>
<th>Targets</th>
<th>Existing</th>
<th>2025</th>
<th>2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Share of municipal waste prepared for reuse and recycling</td>
<td>50% (2020)</td>
<td>60%*</td>
<td>65%*</td>
</tr>
<tr>
<td>Share of municipal waste landfilled</td>
<td>Reduce landfilled bio-waste to 35% of 1995 production levels (2016)</td>
<td>/</td>
<td>10%^</td>
</tr>
<tr>
<td>Share of packaging waste prepared for reuse and recycling</td>
<td>55-80% (2008)</td>
<td>65%</td>
<td>75%</td>
</tr>
<tr>
<td>Share of plastic packaging prepared for reuse and recycling</td>
<td>22.5% (2008)</td>
<td>55%</td>
<td>/</td>
</tr>
<tr>
<td>Share of wood packaging prepared for reuse and recycling</td>
<td>15% (2008)</td>
<td>60%</td>
<td>75%</td>
</tr>
<tr>
<td>Share of ferrous metal packaging waste prepared for reuse and recycling</td>
<td>50% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
<tr>
<td>Share of aluminum packaging waste prepared for reuse and recycling</td>
<td>50% (2008)</td>
<td>75%</td>
<td>85%</td>
</tr>
</tbody>
</table>
A relevant policy which has had implications on waste treatment and disposal is the ban on landfill adopted by several member States following the Landfill Directive. While in some Member States waste is still mostly landfilled, the ban has led to significant reductions in landfilled waste and consequently to increases in alternative options, such as recycling or recovery. As regards plastic waste, the ban has shifted waste treatment primarily towards incineration (EP, 2017). In addition, stricter environmental requirements on incineration plants and landfill sites have also contributed to increasing the competitiveness of recycling (EEA, 2011).

From the four legislative proposals on waste policy, a list of main measures which are relevant to the waste management sector, in addition to the aforementioned revised targets, can be identified (EP, 2016):

- Introducing an early warning system for monitoring compliance with targets
- Setting minimum requirements for extended producer responsibility schemes and differentiating the contribution paid by producers on the basis of the costs necessary to treat their products at the end of their life
- Promoting prevention (including for food waste) and reuse
- Streamlining provisions on by-products and end-of-waste status (the stage at the end of the waste treatment process when materials are no longer considered waste, provided they meet certain conditions)
- Aligning definitions, calculation methods for targets, reporting obligations and provisions on delegated and implementing acts.

**EPR schemes**

EPR can help to reduce both raw material consumption and waste generation by increasing the collection and recycling rates of the waste stream addressed. EPR schemes may also result in changes in product design to facilitate dismantling, reduce the level of hazardous substances used and increase the amount of recycled materials used (Ecologic & IEEP, 2009). EPR can also encourage producers to design products to reduce material input requirements and make greater use of recycled materials, for example through the eco-modulation of fees. EPR can lead to reduced raw material extraction:

- Full implementation of the WEEE Directive has an estimated potential of reducing lead (Pb) in the EU by 131-340 kilotons per year in the EU (Arcadis et al., 2008);
- In 2007, 28 tonnes of platinum and 31 tonnes of palladium were recovered worldwide from automotive catalysts, representing almost 15% of the global mining production (UNEP, 2009);
- ELV Directive implementation led to a reduction of over 50,000 tonnes of waste oils and other fluids in the EU per year (GHK, 2006).
- Between 2005-2013, the European Recycling Platform (ERP) collected 2 million tonnes of WEEE, equating to the recovery of 16 tonnes of gold, 130 tonnes of silver and 160,000 tonnes of copper, prevented emissions of around 3,000 tonnes of ozone-depleting substances (i.e. CFCs), and a saving of around 2.7 million tonnes of CO₂ emissions from the energy required to extract virgin metals (INSEAD, 2014).

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| Share of glass packaging waste prepared for reuse and recycling | 60% (2008) | 75% | 85% |
| Share of paper and cardboard packaging waste prepared for reuse and recycling | 60% (2008) | 75% | 85% |

Source: European Commission\(^\text{152}\), 2015

3.2.1 Regulations on sewage sludge application to land
Farmers’ usage of sewage sludge is regulated by the EU Directive (EU 86/278/EEC) on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. Limits are set on the concentration allowances for 7 heavy metals. Microplastic concentrations are not mentioned (EC, 2015, IEEP, 2018).

3.2.2 EU regulation on fertilisers
EU Regulation (EC) No 2003/2003 on fertilisers determines the manufacturing and application of fertilisers in the EU. While ensuring a common market for mineral fertilisers, it does not address waste-based fertilisers. Legislation at national level on fertiliser quality present some restrictions in relation to contaminants. For instance, Germany allows a maximum of 0.1% weight of foreign matter (e.g. plastics); however; particles smaller than 2mm are not taken into account (Withmann et al., 2018).

The proposed revision of the EU Fertiliser Regulation (COM(2016) 157 final) aims at promoting and facilitating the use of organic fertilisers in the context of a circular economy. By introducing harmonized requirements for all CE marked fertilizing products on quality, safety and labelling, the proposed revision of the regulation aims to address fertilisers contamination of soil, inland waters, sea waters and food (European Commission, 2016c).

3.2.3 New proposed regulation on wastewater reuse standards
Current rates of wastewater reuse are well below EU potential, due to the environmental and human health risks associated with improperly treated water. With the aim of ensuring the safety of water reuse, the European Commission has recently proposed new measures to encourage the safe practice of water reuse for agricultural irrigation (EC, 2018a). These include: harmonized minimum requirements for the reuse of treated wastewater from urban wastewater treatment plants; risk management plans for additional water reuse related hazards; and increased transparency on water reuse practices across the EU.

In addition, by improving information and awareness, the regulation can incentivise the adoption of water reuse practices and contribute to water stress alleviation. The introduction of harmonized standards for water reuse can prevent existing obstacles to the free movement of agricultural products which have been irrigated with reused water, increasing confidence in the practice (EC, 2018b). Existing standards are not harmonised at EU level and divergences across Member States can lead to insufficient levels of safety once the agricultural goods have reached the common market (Alcalde-Sanz and Gawlik, 2017). The implementation of the EU legal framework is estimated to increase water reuse in agricultural irrigation from 1.7 billion m$^3$ per year to 6.6 billion m$^3$. The proposed regulation builds on two previous EU legal instruments which already included water reuse as a practice to embrace the circular economy, but lacked specified conditions - the Urban Wastewater Framework Directive and the Water Framework Directive (EC, 2018b).

3.2.4 Common Agricultural Policy (CAP)
Support for wastewater reuse is provided through funding from the Pillar 2 of the CAP, the EU’s rural development policy, designed to support the Union’s rural areas in overcoming environmental, economic and societal challenges. This is offered on a voluntary basis by Member States through their 7-year Rural Development Programmes based on a European ‘menu of measures’. For water reuse in irrigation, measure 4 ‘Physical investment (processing of farm products, infrastructure, improving the performance and sustainability of farms, etc.)’ is of particular relevance (EP153).

Wastewater reuse is a common practice in Cyprus, with over 20 million m$^3$ re-used per year, primarily for irrigation but rules currently in place set strict standards on the uses of recycled wastewater. The EAFRD has been co-financing, since 2007, the installation of a smart irrigation system in a seedling nursery in the country. The new irrigation systems includes specific components: automated desalination of groundwater, rainwater and wastewater collection as well as treatment of the latter. The support from the EAFRD for the

smart irrigation system has decreased the costs of water use and improved product quality (ENRD, 2018) (ENRD\textsuperscript{154}).

<table>
<thead>
<tr>
<th>ReQpro project – Wastewater reuse for irrigation in Emilia-Romagna, Italy</th>
</tr>
</thead>
<tbody>
<tr>
<td>The Life project ReQpro aims at promoting treated wastewater as a substitute to surface and groundwater resources for irrigation, contributing to the protection of fresh water resources. The project, started in 2012, is applied in the Italian region of Emilia-Romagna, where water consumption for irrigation is 66%. The introduction of wastewater reuse represents a reliable source of water for irrigation. A concrete example of this is given by the Mancasale wastewater treatment plant, the largest one in Emilia-Romagna. The treated wastewater is used to irrigate an agricultural area of 2000 ha. The benefits achieved from the project included, in addition to preserved water resources, an annual energy savings between €2.4 and €4.8 million (ReQpro\textsuperscript{155}) (EC, 2013) (ENRD, 2018).</td>
</tr>
</tbody>
</table>

### 3.3 The impact of changing consumer preferences

The design of waste management services can affect their consumption. In particular, differences in payment systems, geographical location, collection strategies and public acceptability and engagement are among the main factors affecting the use and success of waste management services.

The implementation of Pay-as-you-throw systems can lead to a range of benefits in the waste management sector. Given proper infrastructure and collection, a system of this kind can increase the amount of waste being recycled and reduce residual waste (EC, 2016a).

The accessibility of waste collection centres, or the proper implementation of door-to-door collection, also influences the efficiency of waste management services and largely depends on the geographical location – densely populated urban areas or sparsely populated rural areas (EC, 2016a).

The transition to a circular economy requires substantial changes to happen at all stage of products value chain. On the consumer side, growing trends towards circular consumption approaches are leading to an increase in re-use and recycling. Changes in cultural norms have been achieved as a result of policy interventions, such as the integration of taxes/charges and deposit refund schemes. Changes of this kind have implications in the management of waste. While the impact of consumers’ behaviour strongly depends on the different materials and products, an increase in consumers’ awareness on the impacts of an improper management of waste and better knowledge of recycling processes can lead to greater engagement in waste sorting and recycling activities.

Consumers’ preferences represent a significant factor influencing the application of sewage sludge to land for agriculture. Perceptions of the general public towards the practice are generally negative but vary significantly across Member States and have evolved over time. The introduction of the Sewage Sludge Directive has gradually succeeded in encouraging the use of sewage sludge in agriculture and stricter limits have contributed to a higher level of acceptance of the practice. National legislation on sewage sludge have been strengthen in Member States over time, introducing stricter limits on the contaminants already regulated and adding requirements for others. Nevertheless, the sewage sludge applications remain limited in some Member States (Bio et al., 2014).

### 3.4 Dependence on investment and future risks

Investment in waste management is typically intended to bring about improved waste treatment e.g. developing more sanitary landfills, avoiding waste landfilling and increasing material recovery and recycling. Investment may come from the national level (motivated e.g. by the need to achieve EU waste management targets), the regional/local level (since in most cases local authorities are responsible for waste collection), and the private sector (including waste management companies and producers,

\textsuperscript{155} http://reqpro.crpa.it/nqcontent.cfm?a_id=11975&tt=t_law_market_www
particular those whose products are subject to EPR schemes). This therefore makes for a fairly complex picture of investment.

Patterns of investment in the waste management sector can sometimes lead to lock-in to specific methods of waste management, in particular since the development of waste management infrastructure typically involves significant up-front costs and long lead-in and depreciation times. This is particularly the case in countries where significant investment has been made in incineration/waste-to-energy capacity, since a constant stream of waste is required for such plants to function efficiently. This has led to overcapacity in some cases, increasing imports and creating a barrier for recycling by driving waste towards incineration/waste-to-energy (the so-called “vacuum cleaner effect”), in particular for commercial waste (Wilts and von Gries, 2014).

3.4.1 Wastewater reuse

Sectors at risk of water scarcity, such as agriculture and power generation, have incentives to invest in water reuse infrastructure to mitigate risks. 24% of water is abstracted in Europe for agriculture, reaching 80% in some southern Member states (CDP, 2014).

In 2013, the environmental investment in wastewater management was €5.4bn by governments and €13.8bn by private and public companies specialized in sewerage, waste collection, treatment and disposal, and remediation activities.

The factors influencing the investments in wastewater reuse can be categorized under: physical/environmental factors, economic/financial factors, and institutional/legislative factors.

Physical/environmental
Due to climate change, the frequency and intensity of droughts, as well as their related environmental and economic damages, have increased. The associated environmental and economic damages, such as damage to agriculture and infrastructure, call for a more efficient use of water resources in Europe, in particular in the southern Member States. Increased water stress as a result of both an inefficient use of water resources and seasonal, geographic and climatic variabilities can impact the supply and demand for water (EC, 2018b).

In agriculture, this has repercussions on the investments in irrigation infrastructure. In a context where water stress is a significant pressure, wastewater reuse for irrigation presents an alternative and a reliable water resource. In order to achieve acceptable risk mitigation for human health and the environment, further investments are needed for the treatment of wastewater for agricultural purposes (World Bank, 2006).

Economic/financial
Wastewater reuse should be seen as an alternative rather than an additional option to freshwater resources. This way, investing in wastewater reuse for agriculture can result in reduced costs as investments to develop new resources for agriculture are avoided. Wastewater reuse investments are therefore prioritised where the costs of developing new freshwater resources are high. In the agricultural context, wastewater reuse investments are more feasible compared to potable water uses, due to considerations of water quality and risks (World Bank, 2006). The economic factors influencing investments in wastewater reuse also depends on the infrastructure costs to be borne.

Institutional/legislative
The adoption of wastewater reuse practices for irrigation differ considerably across countries in the EU. This is linked to geographical and climatic conditions as well as to public acceptance and legislative frameworks. In particular, non-harmonised requirements for water reuse application, as currently is the case, can create obstacles to the free movement of agricultural products which have been irrigated with reused water, discouraging investment. On the contrary, new proposed EC regulations to implement EU-wide standards for wastewater reuse can positively impact on investments.

Economic and financial considerations of wastewater irrigation – The Llobregat Delta, Spain

FAO has carried out a cost-benefit analysis for the reuse of wastewater for irrigation purposes in the Llobregat Delta in Spain.
As regards the application of sewage sludge to agricultural land as fertiliser, usage is determined by a number of key stakeholders and factors:

**Farmers**

Sewage sludge provides an organic and low-cost alternative to synthetic fertilisers and therefore represents an opportunity for farmers to reduce costs and improve the image of their products. Nevertheless, they can encounter obstacles from food industries and retailers (i.e. the buyers of their products) due to their quality requirements. Reluctance to use sewage sludge as a fertiliser is based on these concerns which can lead to loss of consumers’ confidence.

**Food industry/retailers**

The general public plays an important role in determining the industry and food retail’s attitude towards sewage sludge applications to agriculture. The associated impacts on products quality, whether real or perceived, influences their image and marketing (EC, 2001).

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**Table: Cost and Benefit Analysis**

<table>
<thead>
<tr>
<th>Costs</th>
<th>Mill EUR/year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wastewater treatment</td>
<td>0.59</td>
</tr>
<tr>
<td>Wastewater conveyance</td>
<td>0.21</td>
</tr>
<tr>
<td>Freshwater conveyance</td>
<td>0.81</td>
</tr>
<tr>
<td>Total</td>
<td>1.61</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Benefits</th>
<th>Mill EUR/year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cost savings in water abstraction</td>
<td>0.06</td>
</tr>
<tr>
<td>Cost savings in fertilisers</td>
<td>0.01</td>
</tr>
<tr>
<td>Increase in yields</td>
<td>0.39</td>
</tr>
<tr>
<td>Value of released freshwater</td>
<td>8.13</td>
</tr>
</tbody>
</table>

*Source: FAO*  

[156](http://www.fao.org/fileadmin/templates/FCIT/PDF/WASTEWATER_ECONOMICS.pdf)
4 Future Policy Priorities

4.1 Research questions

- The links between production in these sectors and consumption including consumption orientated policy tools?
- What is stopping the development of ‘water tech’ sector in Europe? Is it a lack of relevant R&D, or a lack of investment? If the latter, how much investment are we talking about?
- What are the opportunities for these sectors (including jobs, growth and investment opportunities) provided by the environment and by environmental policy?
- What are the potential business evolutions (business model, product specificities, etc.) driven by environmental policies or voluntary initiatives influenced by the environment in a given sector?
- What does a more resource-efficient water system look like?
- If we successfully developed a European ‘water tech’ sector, how would we do it (what policy levers?) and what would it look like? Does it create jobs/economic activity in Europe, and does it create the potential for exporting this globally?

4.2 Links between the sector and the Sustainable Development Goals

The sewage and waste management sector have clear links to several of the Sustainable Development Goals and targets contained within them. Developments and improvements in the sector offer the potential to significantly progress on a number of goals. For Goal 6 (clean water and sanitation), the sector has a role to play in helping to improve water quality by reducing pollution, eliminating dumping, minimising release of hazardous chemicals and materials, and substantially increasing recycling. Regarding Goal 11 (sustainable cities & communities), the sector is crucial to the target of reducing the adverse per capita environmental impact of cities, including addressing municipal and other waste management. Regarding Goal 12 (responsible consumption and production), the sector’s development can contribute to the sustainable management and efficient use of natural resources. Regarding Goal 14(life below water), the waste management sector will contribute to preventing and significantly reducing marine pollution from land-based activities, including marine litter.
Figure 4.1 Illustration of Sustainable Development Goals and Targets relevant to the waste management and sewerage value chain

Mapping the SDGs against the waste management and sewerage value chain

6.1. Expand international cooperation and capacity-building support to developing countries in water and sanitation-related activities

6.1.1 Proportion of countries with sustainable (SCP) NAPs or SCP mainstreamed as a priority or a target into national policies

Goal 6: Ensure access to water and sanitation for all

12.1.1: Number of countries with sustainable (SCP) NAPs or SCP mainstreamed as a priority or target into national policies

New direct jobs linked to waste separation, sorting and collection

11.6.1: Proportion of urban solid waste regularly collected and with adequate final discharge out of total urban solid waste generated by cities

Introducing measures such as EPR schemes can increase collection and recycling rates

Proper wastewater treatment allows for safe secondary applications, such as on-farm agriculture

Revised EU waste management targets incentivise increases in recycling and reuse. The reuse of wastewater is increasing in the EU, providing a reliable source of water supply

Provision of comprehensive waste management strategies can reduce leakage of waste to the natural environment

System design

Waste production

Separation/transport

Collection/transport

Treatment

Secondary use

Disposal

-ve contribution

12.2. Achieve the sustainable management and efficient use of natural resources

Goal 12: Ensure sustainable consumption and production patterns

12.3.1 Proportion of land that is degraded over total land area

15.3.1: Proportion of land that is degraded over total land area

EU levels of waste generation are still high: each year, 2.7 billion tonnes of waste are produced in the EU

The collection and transportation of waste from the point of generation to the point of reuse or treatment may generate significant GHG and NOx emissions

The application of sewage sludge as a fertilizer on land is associated with emissions of pollutants and is an important source of microplastic pollution in soil

12.5. Reduce waste generation through prevention, reduction, recycling and reuse

14.1. Prevent and significantly reduce marine pollution of all kinds

14.2. Combat all forms of pollution, including through greening the BLUE economy

14.3. Promote integrated pollution prevention and control

15.3. Promote integrated water resource management and effective flood risk management

15.4. Promote the sustainable use of natural resources

15.5. Enforce严格 enforcement of national environmental laws

15.6. Strengthen the means of implementation for environmental protection and sustainable development

15.7. Improve environmental governance and raise awareness of environmental protection responsibility
4.3 Increased reuse

The revised waste management targets outlined above, together with the priorities set out through the waste hierarchy, emphasise the importance of preparing the different waste streams for reuse. Proper waste sorting and collection systems support improvements in the proportion that is reused. Increasing reuse can reduce demand for manufacture, therefore reducing raw material inputs, water, energy and transportation, reducing environmental impacts. As the potential for product reuse is determined at the design stage, in addition to the availability of proper sorting and collection systems, several market-based instruments have been identified to support ‘design for reuse’. Better harmonisation of extended producer responsibility (EPR) schemes and the introduction of eco-modulation of fees based on reusability, reparability and recyclability criteria can further support the uptake of reuse and recycling, while increasing collection rates.

4.4 Increased use of wastewater

The sectors offering the greatest potential to increase the use of wastewater and which have incentives to invest in water reuse infrastructure are those most at risk of water scarcity – power generation and agriculture (CDB, 2014).

4.5 Sewage-to-energy

Sewage sludge treatment represents a relatively minor fraction of total processed wastewater volume, yet its processing costs can account for up to 50% of wastewater treatment plant operating expenses (e.g. US EPA, 2008). What is more, older sewage sludge processing operations could also involve high environmental costs. As far as agricultural application is concerned, these costs are mainly associated with contents of heavy metals, organic compounds and pathogens as well as groundwater leaching of untreated sludge (Lamastra et al., 2018). Given that agricultural application of sewage sludge cannot handle the entire volume of EU production, use of sewage for energy production is of interest (Spinosa et al., 2011; Kacprzak et al., 2017). This option is all the more relevant in view of the EU’s renewable energy & climate goals.

Well established sewage-to-energy technologies, such as classic anaerobic digestion (AD), allow Waste Water Treatment Plants to offset some of their energy use while also mitigating sludge disposal. That being said, recent technological and sludge management advances call for a detailed research effort to establish how they perform in terms of environmental impacts.

4.6 Increased wastewater reuse for agriculture

As highlighted in the previous section, wastewater re-use is well below the EU potential and has large scope to be increased in many Member States. Where such water can be used as an alternative to freshwater, it has the double benefit of reducing investment costs for (e.g.) agricultural firms, who can access/process wastewater more cheaply than freshwater, while also limiting (or reducing) the environmental impact of the sector. Harmonised and stricter EU-wide standards for wastewater reuse can overcome the current divergences across Member States and further incentivise investments by both the agriculture sector and the water utilities. As the safety of the agricultural products irrigated with wastewater increase, so does the confidence in and the uptake of the practice (EC, 2018b).

4.7 Socioeconomic implications of sewerage and waste management investment

4.7.1 Sewerage

The primary question to be addressed in the impacts of future policy is the impact of investment, and what such investment might look like (in scale and distribution across the Member States). Modelling of the macroeconomic impacts of investment across the EU’s water infrastructure is presented in detail in in the sector study looking at Water Supply. Conclusions are set out briefly below.

Investment in water supply and/or sewerage has a small positive economic impact. The investment itself acts as a stimulus, leading to a short-term boost to GDP and employment, including the creation of local jobs in the construction and installation of infrastructure. However, in the longer term, the payback of such investment leads to slightly lower output in the economy than would otherwise be the case.

There are some mild redistributive impacts of investment in water, assuming that it is distributed according to need (and sourced from central pooled funds, rather than funding being structured at the national level).

In addition, improving the state of Europe’s water supply, treatment and sewerage infrastructure has additional benefits beyond these. It boosts water security (including flood risk management), reducing reliance upon rainwater and other fresh water sources, as well as improving resource efficiency and helping to reduce
the environmental footprint of water supply (directly) and all sectors of the economy which use water as an input.

4.7.2 Waste management

The waste management sector was analysed in detail in ‘Impacts of circular economy policies on the labour market’ (European Commission, 2018c). This study used E3ME to model the macroeconomic impacts of a more circular EU economy, across key sectors. The study used an ‘activities approach’, with scenarios defined by increased circular economy activity, rather than a ‘policy approach’ (for example, the plastics tax modelling in the Rubber & Plastics Annex report to this study used a ‘policy approach’).

The waste management sector is central to realising a more circular economy, ‘closing the loop’ from waste to resource for all sectors of the economy. European Commission (2018c) explains that most substantial impacts of the circular economy on the waste management sector ‘originate from circular economy activities in other sectors. Other sectors’ activities will lead to changes in demand for the waste management sector’.

The overall impacts of increased circular economy activity on both GDP and employment are positive. The ambitious scenario found GDP and employment increases, compared to baseline in 2030, of 0.5% and 0.3% respectively. Employment increase in waste management is over 50% compared to baseline. Positive economic impacts of circular economy improvements in the sector are driven by: 1) increased investment required in recycling facilities, a comparable stimulus mechanism to water infrastructure investment; and 2) higher labour intensity of recycling activities compared to traditional waste management, leading to higher aggregate employment and induced effects in consumer expenditure. Imports decrease by 0.8%, given reduction in demand for fossil fuels and raw materials. The analysis finds evidence of limited impacts on the competitiveness of EU industry; export changes are negligible.

The substantial expansion of the waste management sector in any scenario of increased circular economy activity raises the issue of capacity constraints in the sector, both in terms of capital and skills shortages.

Implementation of the policy priorities detailed earlier in this chapter require investment to be realised. Key macroeconomic impacts of increased use of wastewater and reuse for agriculture can be inferred from results for water infrastructure investment. A key difference is that cost of investment is likely to be borne by the private sector, rather than the public. In this case, redistribution effects across Member States would not arise. Where costs of investment exceed cost savings from reduced water usage, there would be a negative effect on the competitiveness EU industry.

Similarly, an expansion of sewage-to-energy capacity would provide an investment stimulus. The net effect on industry costs for Waste Water Treatment Plants would be a function of technology and investment costs and future energy prices. Future EU climate policy will have an impact on the relative cost competitiveness of using sewage for energy, through the price of carbon emissions.
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