DYNAMIX

Decoupling growth from resource use
and environmental impacts

Work package 5: Environmental Assessment

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1 Executive summary and overview

Sections 3 to 5 of this report provide a detailed assessment of the expected environmental impacts of each of the policies from the three mixes which we have selected for detailed review. The core of each assessment is a review of the available evidence on potential impact, and an analysis of the likely effects on the key criteria of raw material use, greenhouse gas emissions, land use, and water use; together with additional commentary on toxicity (where relevant), and (for land use policies), biodiversity. The assessments are summarised for each of the key criteria using a simple indicator of the direction and scale of impact:

Estimated magnitude of change (↗ = beneficial; ↘ = detrimental):

↗↗↗↗ or ↘↘↘↘ = High (above 100% deviation from BaU)

↗↗↗ or ↘↘↘ = Medium high (between 50-100% deviation from BaU)

↗↗ or ↘↘ = Medium low (between 10-50% deviation from BaU)

↗ or ↘ = Low (less than 10% deviation from BaU)

However, for reasons explained below, the different nature of the policies, in particular their different scopes, makes it unhelpful to make direct comparisons between these summary values.

1.1 Emerging issues

There are a range of common issues underlying the judgements set out in the assessments, which affect the overall assessment for each policy mix. Some of those issues overlap with the subject matter of the other assessments (social, economic, and in particular governance), and will require more detailed work in the light of the findings of the relevant tasks. Some issues which might usefully be addressed in further development of the policy mixes, however, can already be identified.

1.1.1 Uncertainty

There are a number of drivers of uncertainty in the impact of policies, and the assessments adopt slightly different approaches to its treatment. Some of the drivers of uncertainty are linked to issues pursued in more detail in the other assessments, and may need to be revisited following a synthesis of the findings. For example, governance challenges, both in terms of legal feasibility and in terms of public acceptability, are likely to have a significant impact on the implementation and effectiveness of the individual policies; and those governance challenges are themselves likely to depend in large part on likely (or perceived to be likely) economic and social impacts.

Other drivers of uncertainty are more intrinsic. Uncertainty over feasibility is a case in point: in advance of detailed work on the design of a policy, and indeed the testing of its implementation in practice, it is difficult to predict how feasible it will be to implement. The assessment of the enhanced producer responsibility policy in the overarching policy mix takes the approach of assuming that it will be possible to implement before assessing impacts, which is a valid simplification. However, it may be that for many of the policies which involve the extension of existing tools to new sectors, it will be found that the sectors which have already been targeted with those tools are those for which they are most adapted, and where implementation is easiest.

There are also in many cases uncertainties over the nature and extent of the impacts, beyond the immediate first order effects targeted by the policies. Reducing the environmental footprint of land use in the EU, particularly the agriculture sector, is one example of this problem. Many of the policies developed imply, or create risks of, reductions in production. Even in the event of a reduction in consumption (in response to the consumption policies also included in the mix), there would be an opportunity cost to that reduced production, and the impact on global supply and demand of agricultural products needs to be taken into consideration, with possible impacts through expanded land use in other economies. Similarly, reduction in EU demand for metals may lead to reduced commodity prices, with potential for increased consumption elsewhere. One possible lesson to draw from this is that – as with climate change - the full benefits of resource efficiency policies, as well as the full cost implications, are likely to depend on the extent to which similar policies are adopted in other economies in the world.

1.1.2 Flexibility of policy

One response to the problem of uncertainty identified above is to ensure that policy mixes designed to be flexible to a range of responses, recognising both the difficulty of predicting behavioural response and the reality of the democratic process. The policy mixes each have some elements which are designed to have an immediate impact, and others which are designed to ensure a gradual change in attitudes; this appears to be a pragmatic approach to the challenge of developing the range of policies needed to secure a sufficient shift in resource efficiency. Other policies are capable of being progressively made more ambitious in response to greater political feasibility.

One disadvantage of flexibility in approaches, however, is that it makes it more difficult for the private sector to read what is likely to be required in terms of product and process innovation. A possible avenue for further work is therefore to consider whether clarity of direction can be provided through over-arching policy statements, or through enshrining specific policy objectives for resource use in legislation.

1.1.3 Volume control

Perhaps the most significant challenge for addressing resource efficiency and the need to live within planetary limits through these policy mixes is the difficulty of developing mechanisms to manage the volume of resource consumption. Environmental policy is largely built around quality control mechanisms – either minimum requirements for the safety and environmental impact of products, or emissions controls over production facilities.

Volume control instruments have been developed in some areas – for example, the Emissions Trading System for greenhouse gas emissions; and the National Emissions Ceilings Directive. However, it has been possible so far only where it depends on the ability of

Governments to both monitor and exercise regulatory control over emissions; in effect, these are volume control instruments added to existing mechanisms for quality control over the emissions from production processes. Introducing constraints on the overall volume of raw materials and other resources used in the economy is much more challenging to design, since it implies a significantly greater degree of Government influence over the inputs to production processes.

Further work to examine options for volume control policies could be useful – for example, based on monitoring of resource use, with indicative trajectories consistent with resource efficiency identified in advance, or with triggers created in legislation for the introduction of more constraining policies/ higher taxes in the event of particular levels of resource intensity being crossed.

1.2 Summary of the assessments

Overall, the assessments largely – and unsurprisingly - identified positive impacts from the individual policies; although there are some (particularly in the land use mix) where potential for minor countervailing negative impacts has been identified. It has been challenging to ensure direct comparability of assessments, given the different scope of the instruments considered. We have taken the approach of assessing the impact in relation to the sectors and products covered by policy; but this has the effect of ascribing higher impacts to policies with relatively limited scope, which means that it would be potentially misleading to set the headline assessments alongside each other and invite comparisons. The alternative approach, of assessing all measures on their impacts at an economy-wide level, would have had the alternative drawback of preventing a nuanced summary of the impacts of measures which are only intended to affect a limited number of products or sectors. . However, in general the impacts of most individual policies is assessed as modest: there are few magic bullets, even given the likely optimism bias involved here.

It is difficult to draw conclusions at this stage on the interaction between policies, although there are some (for example, skills enhancement in the overarching mix, R&D in the metals mix) which are clearly aimed at enabling greater impacts from other policies.

1.2.1 Over-arching policy mix

Many of the policies in the over-arching mix are, unsurprisingly, generic approaches which can be applied to a range of individual products (for example, feebate systems, producer responsibility, restrictions on advertising). Impacts are therefore linked to the range of products to which, in practice, they can be applied, and are therefore likely to be highly dependent on feasibility, and on public acceptability.

1.2.2 Land use policy mix

Most of the production policies (in principle, those most directly affecting land use itself) are characterised by low, or in some cases ambiguous impacts on both land use and greenhouse gas emissions, in part because of the scale of the trends being addressed (the likely land use impacts tend to be marginal, even over longer periods, in comparison to the overall scale of agricultural land use, particularly given the global nature of trade in most agricultural products), and in part because of the feedback issues identified above in relation to reduced productivity and reduced production. The impacts from consumption-based policies need to be treated with caution, since they have been assessed largely on the basis of the impacts associated with the targeted products, rather than wider impacts from agricultural land use; however, they seem to provide initial signals that consumption policies are potentially the most beneficial (albeit likely to be those which face greatest levels of initial public acceptability challenges).

1.2.3 Metals policy mix

Similar issues to the over-arching policy mix apply: many of the instruments considered are generic approaches theoretically capable of being applied to a range of products and sectors. The implementation challenges of, for example, the full internalisation of environmental costs have not been addressed in detail, and can be assumed to add an element of uncertainty to the political acceptability issues already addressed in the policy mix. The metals mix relies more explicitly than do the other mixes on the synergies between policies.

1.3 Key issues for further work

Further work will be needed to ensure, where possible, a greater level of consistency between the approaches taken in the environmental assessments, in particular in clarifying how the loose concept of a baseline should be treated, and in resolving differences of approach in assessing the magnitude of impacts (for example, whether impacts should be assessed on the basis of the limited number of products affected by a policy, on a wider sectoral basis, or on the economy as a whole). However, the main value of the assessments lies in the detailed information they provide in relation to each policy, rather than as a crude scoring mechanism.

In the light of results from the other tasks in Work Package 5 (the social, economic, and governance assessments), further assessment should be made of issues affecting the conditions for implementation of the policy mixes. What guidance should we offer policymakers on the practical implications of our work? In particular, what can we say about:

- The extent to which implementation is dependent on securing a greater degree of public acceptability of the policy option;
- Which policies are either a necessary precursor to, or likely to contribute significantly to the success of, other policy options (in other words, how important is the mix, rather than the individual policies; and what sequencing is likely to be best in delivering the advantages of the mix);
- The potential for mitigating any negative economic, social, or public acceptability impacts identified in the other tasks.

It might also be valuable to consider whether responses are available to the general issues identified above (uncertainty; the need for flexibility; and the challenge of achieving volume control); and to consider whether significantly different outcomes are likely in the event of other economies either pursuing, or failing to pursue, similar approaches.

2 Methodology

A full description of the methodology is available in deliverable 5.1, the methodological framework for work package 5. All policy mixes were evaluated against the following objectives and indicators:

DYNAMIX environmental objective	Indicator
Extraction of raw materials: Reducing use of virgin metals by 80 % (base 2010)	Change in the extraction of raw materials
Greenhouse gas emissions (GHG): 2 t CO2-eq per capita per year	Change in the GHG emissions
Land use: Zero net demand of non-EU arable land	Change in the global land requirement required for EU consumption and production
Freshwater use: No region should experience water stress	Change in the water use

In addition, the land use policies were assessed against their impacts on biodiversity. Some of the metals policies have been assessed against impacts on toxicity.

Scoring system

Upwards arrows indicate a beneficial change with respect to the trend under a baseline scenario up to 2050, as described in the policy-mix descriptions, for each of the stated environmental objectives. Downward arrows indicate a detrimental change.

Estimated magnitude of change:

↗↗↗↗ or ↘↘↘↘ = High (above 100% deviation from BaU)

↗↗↗ or ↘↘↘ = Medium high (ie between 50-100% deviation from BaU)

↗↗ or ↘↘ = Medium low (ie between 10-50% deviation from BaU)

↗ or ↘ = Low (ie less than 10% deviation from BaU)

The assessment should be based primarily on literature review, to provide an evidence-based qualitative assessment of likely environmental impacts, both positive and negative. The assessment may also identify the conditions necessary to generate an environmental impact (e.g., public engagement/trust, other policies, funding/incentives etc.).

3 Overarching policy mix

3.1 Boosting extended producer responsibility (EPR)

3.1.1 Short description of the policy

This measure would revise the Waste Electrical and Electronic Equipment (WEEE), End-of-Life Vehicles (ELV), Packaging and Packaging Waste, and Battery and Accumulators Directives in an effort to further optimise extended producer responsibility (EPR). EPR would also be extended to the entire lifecycle of a selected range of products (therefore including their take-back, recycling or disposal) to decrease their total environmental impact. Potential candidates where EPR could be newly applied include: tyres, graphic paper, waste medicines, oils, agricultural films, children's toys and construction materials. Priority would be given to products that are not (yet) covered by EU Directives but for which EPR has been effectively introduced in some Member States. Appropriate standards would be developed and introduced to boost the implementation of the new EPR schemes. The schemes would apply both to EU-produced goods and imported goods. Under the new EPR schemes, manufacturers would have full responsibility (including costs) for the disposal of packaging and other materials associated with their products; this is expected to lead to the integration into the final market price of the products of the environmental costs associated with the goods throughout their lifecycle. This would therefore provide incentives to producers to designing more sustainable, less toxic and more recyclable products. The policy as described in the DYNAMIX fiche suggests that collective responsibility schemes may initially be allowed for a transition period, but that individual producer responsibility schemes would be preferred since they have proven to be more effective in delivering product design change (some comment is made on this in the following section). The proposed programmes would draw on lessons learned from existing EPR schemes, and would act in support of the implementation of the European waste hierarchy, to increase waste prevention, reuse and recycling. Given the European Commission's current drive on issues to promote the 'circular economy', it is anticipated that the work to increase the scope of ecodesign could be carried out over the next few years and find its way in EU legislation by 2018, and that the schemes for the additional product categories could be in place in the Member States by 2020.

3.1.2 Assumptions made, and identified conditions necessary to generate an environmental impact

This assessment assumes that the EPR schemes would be well-functioning and successful in achieving their aims (i.e. with a satisfactory level of participation from producers and consumers). The arrow scores awarded are based on the assumption that EPR schemes for all of the proposed additional waste streams are introduced in the majority of (if not all) EU Member States.

Although the DYNAMIX policy fiche suggests that individual producer responsibility schemes would be preferred over collective responsibility schemes, the choice between the two types of scheme may largely depend on the product group. For complex, expensive products for which the producer can be easily identified (e.g. EEE and cars), individual schemes can indeed be effective in driving product design change. For smaller and/or less easily identifiable products such as several of those suggested in the policy fiche as candidates for EPR (e.g. graphic paper, oils, agricultural films and construction materials) individual EPR may be difficult and costly to implement. In the case of long-lived products such as construction materials, it may also be the case that the producer no longer exists at the end of the product's life-span, meaning that measures would need to be included in EPR schemes to cover 'historical waste' (this is not insurmountable, but would require consideration). Careful consideration would therefore need to be given, likely on a product-by-product basis, over whether to implement individual or collective EPR.

In order to be successful, adequate collection and recycling infrastructure would need to be in place to ensure that the maximum possible amount of waste from the additional waste streams could be captured and processed. The impact of EPR for medicines is not specifically addressed in the assessment, since EPR in this area is assumed to be aimed more at ensuring the safe disposal of medicines rather than any form of recycling/recovery.

3.1.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗↗↗ Medium high (50-100% deviation from BaU)

This assessment proceeds from the assumption that extending the application of EPR to other waste streams (the waste streams suggested in the policy fiche are **tyres, graphic paper, medicines, oils** and **agricultural films**) would reduce both raw material consumption and waste generation. This is based on an assumption that EPR schemes will generally increase the collection and recycling rates of the waste stream addressed. 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).

Some of the proposed waste streams (tyres, graphic paper, oils) are likely to result in greater resource impacts than others (medicines).

Well-designed EPR schemes can help to better reflect the scarcity of natural resources in the price of raw materials/products (helping to tackle e.g. subsidies for resource extraction and the fact that the environmental externalities of resource extraction/processing are not always taken into account by markets). In this way, EPR can encourage producers to design products to reduce material input requirements and make greater use of recycled materials. EPR schemes that involve product take-back, producer-financing of waste management costs, and recycling targets, have the potential to bring about change in product design (i.e. greener, more recyclable products) and waste management (i.e. more recycling) (OECD, 2005).

Evidence exists that EPR can lead to reduced raw material extraction. The full implementation of the **WEEE** Directive is estimated to result in a 131-340 kilotons of lead (Pb) reduction per year in the EU (Arcadis et al., 2008), while in 2007, 28 tonnes of platinum and 31 tonnes of palladium were recovered worldwide from automotive catalysts, which represents almost 15% of the global mining production (UNEP, 2009). Implementation of the **ELV** Directive led to a reduction of over 50,000 tonnes of waste oils and other fluids in the EU per year (GHK, 2006). It takes 5 tonnes of bauxite ore and 32 barrels of oil to make a tonne of **aluminium**; recycled aluminium uses only 5% of the energy associated with using virgin ore (Upstream, 2015). After its first 8 years of operation (2005-2013), the European Recycling Platform (ERP) had collected 2 million tonnes of **WEEE**; this equated to the recovery of 16 tonnes of gold, 130 tonnes of silver and 160,000 tonnes of copper and the associated emissions (see section below on greenhouse gas emissions) (INSEAD, 2014).

With regards to **paper**, it is more beneficial from an environmental point of view to recycle high quality products such as office paper, rather than lower-quality materials (WRAP, 2010a). In EU Member States with EPR schemes in place for graphic paper, recycling rates of over 80% are not uncommon (e.g. in FI, NL and SE) (Bio by Deloitte, 2014).

Recycling rates of over 80% have been achieved for used **industrial oils** in EU Member States with EPR schemes in place (e.g. in BE, FI, IT, DE and PT) (Bio by Deloitte, 2014); collected oils are generally either re-refining (i.e. recycled to produce base oils for the production of new oils) or decontaminated for use as fuel (i.e. energy recovery). It only takes one gallon of used oil to produce 0.25 gallons of new high-quality lubricating oil (as opposed to the 42 gallons of crude oil which would be required to make the same quantity, albeit alongside a range of other products) (US EPA, 2015). For **plastics**, mechanical recycling is the best waste management option in respect of reducing the depletion of natural resources (WRAP, 2010a). 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 including virgin rubber. Technologies also exist to process **agricultural films**: once separated, shredded, washed and dried, they can be processed into pellets which can then be reused to produce film products from bin liners to new agricultural film. One plant in Belgium can recycle agricultural film into high-grade plastic repellets at a maximum rate of around 1,400 kg per hour, providing the capacity to process 20,000 tonnes per year (Wolters, no date), therefore offering the opportunity to reduce the need for that quantity of virgin plastic for the production of films.

3.1.4 Assessment of impact on greenhouse gas (ghg) emissions

Estimated magnitude of change: ↗↗ Medium low (10-50% deviation from BaU)

If EPR schemes function well and increase levels of recycling, it is likely they will result in a reduction in ghg emissions; this is also somewhat dependent on the resulting recycling being carried out to high environmental standards. The levels of reduction will also depend on aspects including the amount of energy needed to process the waste stream to produce useable recyclate.

Greenhouse gas emissions (GHGs) should be reduced through innovation and the application of eco-design during the entire lifecycle of a product (Ecologic & IEEP, 2009). After its first 8 years of operation (2005-2013), the European Recycling Platform (ERP) had collected 2 million tonnes of **WEEE**. This resulted in preventing the release of 3,000 tonnes of ozone-depleting substances (i.e. CFCs) into the atmosphere (representing a global warming potential of 19 million tonnes of CO_2 equivalents), and a saving of around 2.7 million tonnes of CO_2 emissions from the energy required to extract virgin metals (INSEAD, 2014).

With regards to **paper**, any waste management option that reduced landfill is beneficial from the perspective of climate change potential (since paper that degrades in landfill may release methane gas, which is 23 times more powerful than CO_2). Whilst landfill is the least preferred option, recycling and incineration with the latest energy recovery efficiencies are broadly comparable in terms of climate change impacts (WRAP, 2010a). A switch from recycled to virgin content for paper production is likely to incur land use change and additional indirect emissions from the loss of carbon stored in forests (James, 2013). It could therefore be assumed that the inverse is true, and that an EPR scheme that results in an increased use of recycled content to produce paper is likely to reduce the emissions associated with land use change and virgin material extraction. James (2013) also estimated that if the UK were to switch completely from using recycled fibre (which is equivalent to 16-21% of domestic wood demand), approximately 36 million tonnes CO2 eq could be lost from forests.

For **plastics**, mechanical recycling is the best waste management option in respect of energy demand impacts; incineration with energy recovery performs poorly with respect to climate change impact, but pyrolysis may be a beneficial option as the technology develops (WRAP, 2010a).

With regards to **oils**, less energy is required to produce a gallon of re-refined base oil than a base stock from crude oil (US EPA, 2015); this should result in a reduction in associated emissions.

3.1.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

One of the main impacts on land use could be a reduction in the amount of various types of waste going to landfill, as a result of more of the waste stream being captured through the application of EPR schemes which result in increased recycling/recovery rates. This may result in some reduction of the required volume for landfilling the types of waste addressed. The actual magnitude of the reduction and the degree of impact would of course depend on the quantity/volume of the waste stream under business-as-usual; it would be larger for bulkier wastes or those where more waste is produced.

James (2013) argues that the use of recyclate to displace virgin materials for producing **paper** may result in a greater area of planted (and in some cases primary) forest; i.e. in countries where recyclate replaces virgin materials, forest land coverage may increase. Canada, the USA, Norway, Spain and Sweden are cited as examples of countries where recyclate displacing virgin materials resulted in higher land coverage by plantations between 1990 and 2010; the USA also saw an increase in primary forest coverage. Whilst an increased use of virgin materials instead of recyclate would also increase the area of plantations (since they would be planted specifically for paper production), in some cases the area of primary forest would also decline; according to James (2013) this was specifically the case in Brazil. The SORT IT project argued that reduced demand for virgin pulp could reduce the pressure on forests to provide material for paper production (European Commission, 2014). It is, however, worth pointing out that the impact of paper recycling on European land use would be small, especially in northern Europe where forestry is driven more by demand for timber used in sawmills than for pulpwood.

3.1.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (ie less than 10% deviation from BaU)

There is unlikely to be a major impact on freshwater use in households as a result of the introduction of EPR schemes for the waste streams suggested in the policy fiche (tyres, graphic paper, medicines, oils and agricultural films), since they are wastes that do not usually need to be cleaned/rinsed before collection.

With regards to **paper**, recycling is the preferable waste management method in terms of water consumption (WRAP, 2010a).

3.1.7 References

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3.2 EU-wide introduction of feebate schemes for selected product categories

3.2.1 Short description of the policy

This policy measure involves the development at EU level of a common framework for the introduction of bonus-malus schemes across the EU, identifying specific products to be targeted (e.g. household appliances, batteries/accumulators, paints, detergents) and providing a methodology for setting both bonus and malus at the right level for the schemes to be cost neutral. Financial incentives (subsidies/bonuses) would be applied to lower-emission and less resource intensive goods, whilst fees (taxes/charges) would be applied to higher emitting and more resource intensive goods. The measure would aim to take into account a wider range of environmental impacts beyond GHG emissions (e.g. emissions of other harmful substances and small particles, noise). It would aim to change consumers' behaviour (encouraging the purchase of the more environmentally friendly goods) and to stimulate technological innovation amongst manufacturers. It is anticipated that such a feebate system could be introduced progressively for different product categories by 2020.

3.2.2 Assumptions made, and identified conditions necessary to generate an environmental impact

Eilert et al. (2010) refers to two studies that looked into the market impacts of feebates. They suggest that around 90% of the impacts of national level feebate policies would derive from the effect on manufacturers' reaction to the scheme (i.e. product re-design and technological innovation) rather than from a change in consumer behaviour (i.e. making purchasing decisions based on the feebate) (Davis et al, 1995 and Greene et al., 2005). If schemes are undertaken at a sub-national level, however, manufacturers may be more reluctant to respond to a feebate since it applies to a smaller market (Langer, 2005); the consumer response is therefore likely to make more of a contribution to the impact of regionally- or locally-applied feebates, and as such should be accompanied by clear communication of the scheme to consumers to maximise its impact (Eilert et al., 2010). This suggests that a careful analysis of the scope of a feebate scheme may help to determine how the most beneficial environmental impacts could be achieved.

Eilert et al. (2010) also make six suggestions for the design of successful feebate schemes (based on findings from vehicle feebates, but expected to be translatable to appliances and buildings): feebates should be visible to consumers; regional/local feebates should be designed to avoid them being vetoed and withdrawn by national level government; rebates and fees must be large enough to matter (compared with the purchase cost of the product concerned); stakeholders should have adequate opportunity to input to the development of feebates; there should be a clear timetable for the phasing-in/introduction of feebates (to allow manufacturers to react); and exclusions/exemptions and 'dead-band zones' (e.g. products within a 'mid-performance' emissions or energy efficiency band to which the feebate is not applied) should be minimised so that the scheme is not undermined.

3.2.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU)

Feebate/Bonus-Malus schemes may be able to contribute to reducing the toxicity of a product's components or the resource intensity of its production; if the right incentives are used to divert consumers' behaviour from environmentally damaging activities or products, such schemes could be effective in contributing to a reduction in overall resource consumption and associated environmental degradation (from policy mix fiche). However, since the main aim of feebates would be to tackle emissions, impacts on the extraction of raw materials is likely to be somewhat limited (except to the extent that manufacturers' efforts to develop products with reduced emissions results in them using less raw materials).

3.2.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ or ↗↗↗ = Medium low to medium high (ie between 10-50% or 50-100% deviation from BaU, depending on level of feebate and response of manufacturers/consumers)

Since feebate/bonus-malus schemes are particularly relevant for energy using household appliances, the introduction of this policy could reduce fossil fuel use, and therefore associated GHG emissions. A bonus-malus scheme on imported and domestically-produced appliances can be designed to reduce their energy consumption, as well as other environmental impacts (use of hazardous or rare virgin materials, etc.) (National Energy Policy Institute, 2013). If the fees and rebates in feebate schemes are proportional to the product's efficiency or GHG emissions level, products with the greatest GHG savings will receive a significantly large rebate, giving manufacturers a financial incentive to develop the most efficient technologies (Eilert et al, 2010).

The French model of bonus-malus for cars appeared to be efficient in reducing emissions of the car fleet and therefore contributing to fighting climate change; the CO_2 emissions of new cars decreased by 6% between 2007 and 2008 (ADEME, 2009). The bonus malus is estimated to have contributed to a change in the market share of the energy classes of cars, e.g. with the market share of class B cars (second most efficient) increasing from 14% in 2004 to 35% in 2008, and the market shares of less efficient cars decreasing over the same period (from 34% to 23% for class D, 20% to 11% for class E, and 6% to 2% for class F) (ADEME, 2009).

However, the emissions-related impact of feebates will only be positive (i.e. a reduction in emissions) when there is no or only a limited rebound effect (i.e. the products are not used more intensively because they have become more efficient, or the reduction in price does not lead to more products being bought). For example, the French bonus-malus system for cars targeted newly registered passenger cars, but attractive rebates had rebound effects in the form of increased sales of new cars and therefore of an increasing number of drivers and overall increases in environmental impacts (Withana et al, 2014) (although a car scrappage scheme that ran in parallel may also have contributed to a temporary boost in sales). Similarly, the 'Renove' programme for dishwashers in Spain (using labelling to set up a rebate scheme) found that a subsidy to support the purchase of efficient appliances (i.e. labelled as A, A+ or A++) could be expected to lead to an increase in energy bills since both the total number of appliances and the number of labelled dishwashers would increase (the former by 1.4-2% and the latter by 4.8-7.7%) (Galarraga et al, 2013).

One modelling exercise (Eilert et al, 2010) estimated the potential GHG savings that may result from application of a feebate to residential gas storage water heaters (in California). Typical 50-gallon heaters were divided into three efficiencies (an energy factor (EF) of either 0.60, 0.65 or 0.86, with the latter being highly efficient). The pivot point (the separation point between products subject to a fee and those subject to a rebate) was set at the medium efficiency level; high efficiency (and therefore low emission) heaters would receive a rebate of $234 (12% of the original installed price) and low efficiency (high emission) heaters would incur a fee of $80 (about 9% of the original installed price). It was estimated that lifetime CO_2 reductions per water heater resulting from the feebate scheme would be 1.3 tonnes when a consumer switched from a high to medium emission heater, and 5.2 tonnes when switching from a high to low emission heater. If the feebate influenced 25% of California customers to purchase a lower emission water heater, total lifetime emission reductions from three years of water heater sales would range from 0.6 to 2.4 $MTCO_2$, based on annual sales of 615,000.

Another modelling exercise (Galarraga and Abadie, 2014) used data from 11 retailers in 6 representative Spanish regions to estimate the results of feebate schemes for dishwashers, refrigerators and washing machines. The results depended on the goal of the feebate policy (i.e. whether the policy was designed to reduce emissions, to remain revenue neutral, to reduce the number of non-efficient appliances, or to increase the number of efficient appliances). When the purpose of the policy is to save energy (and therefore reduce emissions), the most effective option is to tax both labelled and non-labelled goods (with a lower tax for labelled goods). The analysis does however suggest that similar results in terms of reducing energy consumption can be achieved when a €0 subsidy/tax is applied to energy efficient appliances, and a tax is applied to those appliances that are less energy efficient.

3.2.5 Assessment of impact on land use

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU)

Since the main aim of feebates would be to tackle emissions, impacts on land use are likely to be rather limited, although any impacts there are should tend to be positive.

3.2.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU)

Since the main aim of feebates would be to tackle emissions, impacts on freshwater use are likely to be rather limited, although any impacts there are should tend to be positive.

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3.3 Local currencies for labour-based services

3.3.1 Short description of the policy

The policy would involve the introduction of alternative local currencies which could be spent only on services, and which would be facilitated by public support in the necessary technology, management of a central system for control and administration. Tax would not be levied on transactions in the local currencies. Individual localities would be able to vote to adopt a local currency, subject to a small increase in local taxation to reflect the expected drop in revenues from transactions in the currency.

3.3.2 Assumptions made, and identified conditions necessary to generate an environmental impact

There are two principal assumptions behind the measure. The first is that the availability of a currency which can only be spent on services locally, and which is effectively exempt of tax on transactions, would lead to a shift in consumption patterns towards services. The second is that such a shift in consumption towards services would reduce the environmental impacts and improve the resource efficiency of consumption.

The second assumption seems to be reasonably robust; although we have not been able to identify much published literature comparing the resource intensity of consumption of services as compared with consumption of goods, perhaps because the conclusion seems obvious. For example, Ayres, quoted in Jackson[1], comments that “a new growth engine is needed, based on non-polluting energy sources and selling nonmaterial services, not polluting products”. However, Jackson goes on to note that the services involved should not resemble:

> “anything that passes for service sector activity in modern economies at the present. When the impacts attributable to these are computed properly, most of them turn out to be at least as resource hungry as the manufacturing sectors. The recreation and leisure sector ought to be a prime candidate for de-materialisation in principle. In practice, it’s responsible for around 25% of all energy and carbon emissions attributable to UK consumers.”

It will therefore be important to ensure that the policy is implemented as intended, with a focus on services which avoid the energy costs of transport.

The policy fiche makes clear that the intention is to focus on the labour component of labour-based services; and that the parts of services requiring material inputs would be paid for in the usual currency. This adds an element of complexity to administration of the system (would there be a de minimis level for the “material inputs” element, to avoid catching e.g. hair products used by hairdressers, ingredients used by bakers, etc? how would calculations of the “material input” be verified?), however, which may significantly limit uptake by service providers, and add to the costs of the system.

There is also some potential for unintended consequences of the measure; for example, it would be possible, unless rules preventing it are included, for large service-based businesses to make use of the currencies in order to avoid tax. It may therefore be necessary to limit use

[1] See references section below.

of the currency to individuals, rather than businesses, which may (again) limit its potential impact, for example by making it difficult for more complex services with a variety of inputs to be included. That said, the principal objective of the measure is to shift patterns of consumption, and a focus on individual transactions would still be capable of achieving this.

While there has been some previous work carried out on the impacts of alternative and local currencies, some of which have referred to their imputed environmental impacts, most of them assume that the environmental benefits derive from the consumption of local products and services (thereby avoiding transport costs). It seems likely, however, that to a significant degree the environmental benefits of avoided transport costs are offset by environmental costs of sub-optimal allocation of production. A local currency based on the consumption of services only could – if the design issues identified above are successfully overcome - be more capable of delivering environmental benefits.

Finally, we assume that the success of the measure is unlikely to depend solely on a rational response to the price signals entailed in a lower exposure to tax of labour-based services. By creating a separate, privileged, mechanism for labour-based transactions, a shift in attitudes could be created, with those participating in the currencies reflecting more broadly on their consumption patterns, and on the non-monetary values they place on different types of transaction. There may therefore be a multiplier benefit of this policy in helping to induce the sorts of change in attitudes which would be necessary for higher levels of policy in a number of areas identified in our policy mixes to be introduced.

3.3.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗(?)

To the extent that the expected shift in consumption from products to services does indeed take place, it can be expected that there will be an impact through lower-than-trend consumption of products in comparison to services. There may be some rebound impacts, however, if prices for products are lower in response to reduced demand, leading to some additional consumption in areas which choose not to introduce a local currency; and it can be expected that the local currencies will appeal most to consumers who already have a bias towards consumption of services, and towards assessing the sustainability of their purchasing, which could reduce the overall impact.

3.3.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗ (?)

Again, it seems safe to assume that there would be an element of reduction in consumption of material goods as a result of the measure, with a consequent reduction in the emissions intensity of consumption. There may be some countervailing pressures, however, particularly if local labour-based services involve a more dispersed workforce, with greater demand on heating, lighting, transport, and other carbon-intensive services as a result of less efficient disaggregation of demand.

There may also be potential for using local currencies as a form of reward under local and regional energy demand management projects – although the approach is not mentioned in the policy fiche, it has been an element of some local currency systems or tokens as a reward mechanism for participation in low-carbon energy measures (see for example the description of the Torekes project in Gent (Belgium), described by Joachain and Klopfert (2012)). If such approaches catch the imagination of participants, there is some potential for them having a more significant impact, both in direct energy consumption terms, but also in terms of changing attitudes and preparing participants to welcome more ambitious policy measures.

3.3.5 Assessment of impact on land use:

Estimated magnitude of change: ↗↘ (?)

Some positive benefits should accrue from the reduced land footprint following from reduced consumption of products. However, these may be offset if the overall impact of increased consumption of local services is an increase in pressure on the built environment to house a greater number of micro-businesses.

3.3.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ (?)

As with other environmental impacts, a small positive impact can be expected from a shift in consumption from products to services, as a result of reduced water consumption in manufacture.

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3.4 Reduced VAT for the most environmentally advantageous products and services

3.4.1 Short description of the policy

This measure involves a reduced VAT rate of 6% for the most environmentally advantageous or least resource intensive products and services across a wider range of products and activities, which may include environmentally beneficial works done in the housing sector (renovation, retrofitting, maintenance), electricity from renewables, some local, labour intensive services and products, virtual books, organic products, hotels/tourist attractions etc., for which policy instruments in the form of EU environmental standards (Ecolabel, EU energy label) already exist or could be easily introduced. The aims are to: increase resource efficiency in the production of products and delivery of services, encourage more recyclable products, provide energy savings throughout the lifecycle of energy using products, and speed up the withdrawal from the market of less efficient products and services. The measures would be built on existing practices and instruments, e.g. by awarding the reduced VAT rate to products/services that have been given the European Ecolabel. Reduced VAT rates are already regularly used in MS all across Europe to pursue a wider range of objectives and this new environmental measure would be approved following the same procedures. It is anticipated that such a reform of the VAT system would not compromise revenues raised, but would rather shift the tax base towards more environmentally damaging products, according to the polluter pays principle. Given that this is a tax measure, it may be difficult to achieve consensus between Member States, but it is anticipated that this could be overcome in the context of a broader tax reform, and that the instrument could therefore be introduced as early as 2020 (through the open method of coordination (OMC) up to 2030, if the unanimity requirement is thought to be a barrier).

3.4.2 Assumptions made, and identified conditions necessary to generate an environmental impact

It is assumed that although some Member States may initially be reluctant to introduce such a measure, it would be possible to reach consensus between Member States to allow its introduction, e.g. in the context of broader tax reform which would create new sources of income for national exchequers.

It is assumed that the reduced VAT rates would encourage purchase of those greener products to which the reduced rate is applied, due to the comparable less green alternatives therefore being comparatively more expensive. Some studies suggest that permanently lowering the VAT rate on a particular good (or service) will lead to a reduction in the price of the good, and that as the price reduces, consumers' demand for the good or service will eventually expand (Copenhagen Economics, 2007).

One study from 2007 (Copenhagen Economics, 2007) argues that the rationale for extending reduced VAT rates to stimulate consumption of goods with positive externalities (e.g. energy saving appliances) is not clear. Although demand for any product can be boosted by lowering VAT rates, lower VAT on energy-saving appliances will have a limited effect on CO_2

emissions if they are already covered by other regulatory instruments (e.g. emission trading schemes), and the effect on total energy use may not be beneficial, since it may simultaneously switch consumption from less to more energy efficient appliances whilst increasing the number of such appliances sold (therefore cancelling out at least some of the energy gains). However, the study also concludes that there is an argument for selective cuts in VAT rates, e.g. for locally supplied services and parts of the hospitality sector.

3.4.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (0-10% deviation) to ↗↗ Medium low (between 10-50% deviation from BaU)

In principle this policy may affect a wide range of material flows which are linked to the production of products. Higher uptake of 'green products' (those that use resources more efficiently and cause less environmental damage along their life cycle than other similar products of the same category) can reduce environmental damage, create higher consumer satisfaction, and generate potential economic benefits for producers and consumers through more efficient natural resource use (European Commission, 2013a). The European Ecolabel, which is proposed as a criterion for the application of reduced VAT, tends to require a reduction in the use of the most harmful substances as product components. Hence, this measure may reduce the flow (extraction, use) of these types of product inputs in particular, whether they are imported or supplied domestically. Reduced VAT for environmentally-sound products and services could be expected to contribute to reducing the quantity of waste generated in an economy, since a whole range of ecolabelled products are made out of recycled materials or can be more easily recycled. If an ecolabel guaranteed access to a reduced VAT rate, it may also encourage manufacturers to create more ecolabelled products, resulting in a virtuous cycle with regards to resource use in products. By January 2012, there were approaching 18,000 ecolabelled products on the EU market, including cleaning products, paint, tissue paper, floor coverings, TVs, soaps and shampoos (European Commission, 2015a).

Some material-related impacts of ecolabelled products have been identified, e.g.: cardboard boxes for ecolabelled light bulbs use at least 80% recycled packaging; ecolabelled campsites and tourist accommodation are restricted on their use of pesticides and fertilisers; ecolabelled graphic paper is guaranteed to come from recycled fibres or sustainably managed forests and does not use chlorine gas as a bleaching agent; one ecological and ecolabelled hotel in Madeira reported an 11% decrease in energy consumption and a 26% decrease in gas consumption after its first year with the EU Ecolabel (2004); and an ecolabelled hotel in Paris saw a 16% decrease in gas consumption since being awarded the EU Ecolabel in 2010 (European Commission, 2015b). However, material use does not appear to be central to ecolabel criteria.

One study that attempted to estimate potential resource savings resulting from ecolabelled products[2] suggested that direct material savings (other than hazardous substances) could

[2] The study only took into account those product groups where ecolabels had already been awarded in 2004, i.e.: copying and graphic paper, tissue paper, cleaners for sanitary facilities, all-purpose cleaners, detergents for dishwashers, hand dishwashing detergents, laundry detergents, washing machines, dishwashers, refrigerators, televisions, personal computers, laptop computers, light bulbs,

amount to 530,700 tonnes per year in the EU25 if the market share of ecolabelled products were 5%, 2.1 million tonnes for a 20% market share, or 5.3 million tonnes for a 50% market share. Indirect material savings[3] could amount to an additional 1.5 million tonnes (AEA Technology, 2004).

3.4.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

Reduced VAT for environmentally-sound products and services could be expected to reduce GHG emissions, provided that the reduced VAT encourages use of products that produce lower emissions during their manufacture and/or use.

21 out of the 28 EU MS currently apply a VAT rate to public transport that is lower than the recommended 15% standard rate (European Commission, 2014). In those countries, amenities from using more public transport are obvious, and include environmental and economic efficiency, reduced congestion, less GHG emissions and lower local air pollution (Green Budget Europe, 2011).

Ecolabelled products typically have to use less energy, both in their production and use phases, than non-ecolabelled products. In this way they can contribute to reducing ghg emissions. Some climate change-related impacts of ecolabelled products have been identified, e.g.: EU ecolabelled wall paints use 10 times fewer VOCs (which can contribute to climate change) than conventional products; one ecological and ecolabelled hotel in Madeira reported an 11% decrease in energy consumption and a 26% decrease in gas consumption after its first year with the EU Ecolabel (2004); and an ecolabelled hotel in Paris saw a 16% decrease in gas consumption since being awarded the EU Ecolabel in 2010 (European Commission, 2015b).

One study that attempted to estimate potential resource savings resulting from ecolabelled products suggested that CO_2 produced from energy use could be reduced by 9.3 million tonnes per year in the EU25 if the market share of ecolabelled products increased to 5%, 37.3 million tonnes for a 20% market share, or 93.1 million tonnes for a 50% market share. Indirect CO_2 savings could amount to an additional 27 million tonnes (AEA Technology, 2004).

A review of the EU Ecodesign Directive in 2012 estimated that the first 12 Ecodesign Regulations will allow energy savings of 385 TWh per year by 2020, which is close to 14% of EU household electricity consumption in 2009 (European Commission, 2013b). This will result in associated savings in ghg emissions from energy generation.

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footwear, indoor paints and varnishes, hard floor coverings, mattresses, soil improvers, textiles, and vacuum cleaners.

[3] Indirect benefits were defined as use of Ecolabel criteria: by other ecolabel schemes; in public & private procurement; by companies to assess their environmental performance; to generate environmental product declarations/green claims; to generate wider minimum environmental requirements for products; to raise stakeholder awareness of products' environmental impact; and as a basis for establishing fiscal measures to promote green products.

3.4.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Imported or domestically used flows of goods benefiting from reduced VAT could be expected to have reduced impacts on land use compared to the standard product in the same product category.

3.4.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

Imported or domestically used flows of goods benefiting from reduced VAT could be expected to have reduced impacts on water use compared to the standard product in the same product category.

Some water-related impacts of ecolabelled products have been identified, e.g.: an ecological and ecolabelled hotel located in Madeira reported a 17% reduction in the volume (kg) of towels used per guest in the first year after the ecolabel was awarded, contributing to an 8% decrease in water consumption; and an ecolabelled hotel in Paris reported a 32% decrease in water consumed, whilst the number of guests increased (European Commission, 2015b).

One study that attempted to estimate potential resource savings resulting from ecolabelled products suggested that water use could be reduced by 12.3 million megalitres per year in the EU25 if the market share of ecolabelled products increased to 5%, 49.1 million megalitres for a 20% market share, or 122.8 million megalitres for a 50% market share. Indirect water savings could amount to an additional 35 Tera litres (AEA Technology, 2004).

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3.5 Step-by-step restriction of advertising and marketing

3.5.1 Short description of the policy

This policy would aim to use regulation to introduce step-by-step restrictions on advertising of certain products, to address the role of marketing and advertising in stimulating consumption levels and fostering values and norms of consumerism. The measure would be designed to address political opposition against marketing restrictions, for example by using voluntary codes of conduct, building on and extending existing regulation (e.g. on alcohol and cigarettes, marketing targeting children, and visual pollution in city centres), seeking synergies with other societal goals such as the improvement of public health or the preservation of historic monuments, using existing EU law on misrepresentative environmental claims, funding consumer/citizen organisations to bring legal action against misleading environmental claims, moving towards restrictions on advertising of luxury goods/conspicuous consumption, and catalysing municipal/national level action through an European Commission Communication suggesting possible actions and encouraging the diffusion of best practices. A step-by-step, bottom-up approach is suggested, to progress over a number of years and possibly lead to an EU regulation with minimum standards in 2030.

3.5.2 Assumptions made, and identified conditions necessary to generate an environmental impact

It is assumed that the restriction of marketing/advertising would be targeted on those goods and services that are deemed to be the most resource-intensive (the policy fiche suggests sweets, sugary drinks, fast food and luxury goods such as jewellery, sport cars or high end fashion). It is also assumed that such marketing/advertising restrictions would actually have a discernible impact on consumers' choices (i.e. discouraging the purchase of such resource-intensive goods and services and encouraging the purchase of more sustainable alternatives). It is not certain that this would be the case, since marketing and advertising is only one of many factors that influence purchasing decisions. Habitual behaviours, deep-rooted attitudes and likes/dislikes would also likely need to be changed amongst those consumers who buy significant amounts of resource-intensive and/or luxury goods and services. It is also assumed that the policy will be accepted by the producers of such goods and services, which again may not be the case; in addition, advertising is a significant industry which generates revenues (e.g. through sales of advertising space), so there may be considerable opposition to its restriction. The likely environmental impacts of this policy are therefore rather difficult to assess. Our analysis is based on the product categories likely to be affected, rather than on economy-wide impacts.

3.5.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

The overall aim of the proposed policy to restrict advertising and marketing of certain products is to encourage more sustainable consumption and production (SCP), which the UN defines as "the use of services and related products, which respond to basic needs and bring a better quality of life while minimizing the use of natural resources and toxic materials as well as the emissions of waste and pollutants over the life cycle of the service or product so as not to jeopardize the needs of further generations" (this is the widely-recognised definition of SCP agreed at the Oslo Symposium on Sustainable Consumption, 1994).

The products and services used and consumed by people have a significant impact on the environment, due to the resources used in their manufacture, the amount of energy they use to function etc. Defra (the UK environment ministry) has recognised that meeting targets to drastically reduce carbon emissions (by 80% by 2050) and to address water scarcity and biodiversity will require products and services that have significantly lower impacts on the environment. Defra also recognises that marketing and advertising about the environment has an important role to play, in terms of educating consumers about the environmental impacts of products and services, enabling better informed choices, and driving businesses to develop greener products. Defra guidance from 2011 suggested that it may be helpful to consumers if: information is provided to explain and raise awareness of less well-understood environmental problems; the prominence and visibility of products with good environmental qualities is enhanced, e.g. by promoting them in displays or websites, or using price discounts/loyalty rewards; and the prominence/availability of environmentally-damaging products was reduced (Defra, 2011).

In the UK, there are specific requirements that apply to advertising/marketing in certain sectors (e.g. food, beauty products, environmentally friendly products) (Gov.uk, 2015a). In addition, there are special rules for product labelling for precious metals and food and drink products (Gov.uk, 2015b). If such rules were adapted to restrict marketing/advertising of resource-intensive products in such categories, and to ensure that the environmental impacts of production were included in product labelling, this may lead to consumers limiting their purchases of such items.

The accumulation of material goods and expansion of services contribute to economic growth, and as such it is economically valuable for artificial needs to be created through advertising. However, goods produced and sold in this way are often unneeded, and therefore essentially waste; this is exacerbated by the waste of energy and materials in inefficient production processes used to make them (Henderson and Capra, 2009). If advertising/marketing of such products which promote 'bad growth' ('growth of production processes and services which externalise social and environmental costs, that are based on fossil fuels, involve toxic substances, deplete natural resources, and degrade the Earth's ecosystems' (Henderson and Capra, 2009)) were to be restricted, and consumers purchased fewer of these items as a result, this should contribute to a reduction in the levels of extraction of raw materials.

A study into the possible extension of the EU Eco-Design Directive to additional non-energy related product groups reviewed the levels of production of certain categories of interest to this policy: in 2010, 5 million tonnes of sausages and other prepared meat products, 3 million m3 of ice cream and frozen desserts and 66 billion litres of bottled and canned soft drinks were sold in the EU (CSES, 2011).

Reduction in the purchasing of goods such as sweets, fast food and luxury goods should also serve to reduce the production, use and disposal of packaging; such items are often sold in significant amounts of packaging, which would not need to be produced if demand for the items fell as a result of restricted advertising.

3.5.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

EU Directive 1999/94/EC on consumer information on fuel economy and CO2 emissions marketing of new cars requires that car manufacturers and dealers provide clear information to consumers regarding fuel consumption and CO2 emission characteristics of new cars at the point of sale. The UK's Passenger Car (Fuel Consumption and CO2 Emissions Information) Regulations transposing the legislation came into force in 2001, and also provide additional requirements about providing the same information in brochures and other promotional literature (Defra, 2011). Whilst increased consumer awareness of vehicles' CO_2 emissions may have contributed to the purchase of more vehicles with lower emissions, other factors also influence consumer choice, including rising fuel costs, CO_2-based vehicle tax regimes, environmental regulations/emissions standards, and technological advances (SMMT, 2014). It is therefore difficult to draw out how much of this impact is a result of advertising, and whether banning advertising for the highest-emission vehicles would have a significant impact. Consumers still choose vehicles that meet their own particular needs, e.g. related to space, utility and performance (SMMT, 2014).

The UK Advertising Standards Authority, which regulates media advertising, has ruled on misleading marketing/advertising information on vehicle emissions. For example, ads run by BMW and Renault in 2010 which claimed that electric vehicles had zero emissions were deemed to be misleading, since there would be emissions associated with the generation of the electricity used to charge the vehicle. The ads were therefore deemed to breach rules on environmental claims, and were not allowed to be run again in their current forms. In addition, BMW was instructed not to repeat claims that stated or implied that an electric vehicle would produce zero emissions in use (ASA, 2010a and ASA, 2010b).

In January 2013, the city of São Paulo (Brazil) enacted a ban on the majority of outdoor advertising (e.g. billboards, neon signs, buses and taxis). Mayor Gilberto Kassab argued that the so-called "Clean City Law" will help to address rising air, water, noise and visual pollution, and the Worldwatch Institute argues that such laws can help to combat global warming by tackling the 'consumer lifestyle that causes the greenhouse gases that cause climate change' (Worldwatch Institute, 2013).

The environmental impacts of fast food include emissions associated with food production, transportation, cooking and packaging. The production of processed meat in particular has a huge impact on the environment, including ghg emissions from farming animals, and in some cases the clearing of forests for more grazing and farm land, which releases extra CO2 emissions (Zollinger-Read, 2013). One study suggested that production of a single cheeseburger requires 7-20 mega joules of energy, the equivalent of 1-3.5 kg of CO2 emissions (Clark, 2013). Any reduction in production should therefore result in associated emissions reductions.

3.5.5 Assessment of impact on land use

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

A study published in 2009 estimated that reducing children's exposure to food advertising on TV to zero would result in a decrease of the average body mass index and a lower prevalence of obesity (e.g. in the USA it could result in between one in seven and one in three obese children not becoming obese) (Lennert Veerman et al, 2009). The study predicted that total consumption by children would decrease by 4.5% as a result of removing such advertising. Another study suggested that a ban on fast-food advertising targeting children in Quebec (Canada) resulted in a 13% reduction in the propensity to purchase fast food on a weekly basis (Dhar and Baylis, 2011). If overall consumption falls, it could be assumed that there may be an associated reduction of a similar magnitude in the amount of land required for the production of the sorts of foods usually subject to TV advertising (including fast foods and sugary foods). The production of meat for fast foods is sometimes associated with the clearing of forests for more grazing and farm land (Zollinger-Read, 2013) and is always associated with some degree of land use; a reduction in production as a result of advertising restrictions may therefore result in associated reduction in land take for these purposes (although the magnitude would of course depend on the amount of land used by the companies whose advertising is restricted).

3.5.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

Some studies (see section above on impact on land use) have suggested that reducing exposure to food advertising may result in a decrease in overall levels of consumption. As well as the potential impact of this on land use, it can be assumed that there may be an associated reduction in the amount of freshwater required for the production of the sorts of foods concerned by the advertising restriction.

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3.6 Enabling a shift from consumption to leisure

3.6.1 Introduction

This strategy measure aims at exploring policies to encourage reduced working hours (either in form of part - time or as sabbaticals). Among other measures it could include examination of longer statutory vacation times, dismantling of discrimination of part-time workers, the introduction of flexible wage records and reductions of the fixed cost of labour that currently disfavour part-time posts (e.g. in employee taxation and administration).

Policies enabling part time work exist in various forms in many EU Member States, particularly in France, Germany and the Netherlands. The EU could initiate a debate by comparing effectiveness and economic impacts of these policies and setting out options in a communication.

3.6.2 Assumptions made, and identified conditions necessary to generate an environmental impact

The instrument is a supporting instrument. It does not directly target physical flows. If however, the instrument does succeed in incentivising lower working hours and lower incomes compared to the reference case, it could potentially reduce resource flows associated with the purchase, use and disposal of consumer goods and the use of services. These reductions will only place take in the assumed amounts:

- if reductions in work time are shared more or less equally across income groups (Reisch and Bietz 2014, 19) as well as between men and women (Druckman et al. 2012, 158).
- if the evidence collected so far on the correlation between income levels and environmental footprint apply (on average) to all social groups,
- if the correlations holds in the future (Nässén and Larsson 2010, 11),
- if long-term effects of work time reduction, e.g. potential negative impacts on technology innovation that could e.g. lower energy efficiency improvements compared to BaU (Nässén and Larsson 2010, 11) do not cancel out positive environmental effects.

The extent of the environmental effects depends on the level of work time reduction achieved compared to the reference scenario which, in turn, depends on the acceptability of the measures. This assessment proposes a conservative assessment. Work time reduction compared to BaU is assumed to remain below 20 %. If however, work time reduction were to be pursued more vigourously, a more substantial contribution to resource use targets is possible.

In addition to affecting the total volume of goods and services consumed in a household, changes in working time might also alter the composition of consumption, but the effect is more difficult to track and evidence on the net effect with respect to environmental impacts is not conclusive so far (Pullinger 2014, 12). However studies that have assessed the ‘time effect’ show that its impact is significantly smaller than the income effect, i.e. the lower consumption level due to reduced income (Pullinger 2012; Nässén and Larsson 2010). Nonetheless supporting policy measures to prevent ‘time rebounds’ are highly recommended to avoid that people use their freed-up time in environmentally harmful ways. A key policy in this regard could be investment in sustainable transport infrastructure for organising both leisure and work activitites (Druckman et al. 2012, 158f.).

In the reference scenario description in Deliverable 4.1, no data on future work time assumptions are given, but only a reference to the Commission's Ageing Report. Our assessment has therefore been hampered by a lack of clarity on how patterns of work time are assumed to develop in the reference case.

3.6.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Several studies have statistically shown a significant correlation between working hours per employee in a country and the country's average ecological footprint which includes extraction of raw materials. In a simple regression analysis for 18 OECD countries, Schor showed that "hours per working-age person are […] a significant predictor of ecological footprint" (Schor 2005, 47). Based on a structural equation model that compares OECD and other nations' data Hayden and Shandra (2009) confirm the finding. Similarly, a more sophisticated analysis using 1970–2007 data from 29 high-income OECD countries also illustrated the correlation of working with higher ecological footprints. In this case, the authors controlled for other factors such as differences in average temperature (Knight, Rosa, and Schor 2013). In all cases the lower environmental impact is explained by the income effect due to the lower affluence resulting from lower working hours. With respect to changes in the composition of consumption or technology used, both negative environmental impacts, e.g. more free time leading to more travel, and positive ones, e.g. resulting from lower use of convenience food, are discussed, but authors agree that the income effects outweigh the combined effects of subsitutions.

Even though future trends cannot be simply extrapolated from correlations observed in the past, the basic correlation between incomes and an economy's material throughput has been sufficiently robust between and across countries to allow the assumption that significant working time reductions, shared more or less equally across society will on averaged dampen extraction of raw materials compared to BaU. Based on the most conservative scenario used by Rosnick and Weisbrot (2006) for energy consumption, one could for example use the assumption that every 1 % of working time reduction per worker results in a 0.32 % reduction of the extraction of virgin materials and GHG emissions.

3.6.4 Assessment of the impact on greenhouse gas emissions

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Of all environmental impacts, the effect of work time reduction on energy use and associated GHG emissions have been studied most to date – both through macro and micro level studies which have yielded comparable results.

On the macro level Rosnick and Weisbrot (2006) have shown in a comparison of Europe (EU-15), Australia, the U.S. and Canada that total hours worked in an economy correlate with overall energy consumption (controlling for population growth and temperature differences). Based on this simplified assessment the authors conclude that the U.S. would consume 20 % less energy if it were to reduce its working hours to EU-15 levels (Rosnick and Weisbrot 2006, 7).

Micro level studies use household-level time use or expenditure data or a combination of both to assess the effect of working time reduction on GHG emissions. Combining time use data for the UK with emission data from UK environmental accounts, Druckman et al. (2012) show that leisure activities are on average associated with a lower carbon footprint than the average for all other activities (1 kg CO_2eq./h compared to 1.2 kg CO_2eq./h). Emissions in this case include emissions embedded in products that are imported to the UK.

Two studies using household expenditure surveys for the UK and the Netherlands (Pullinger 2012) and for France (Devetter and Rousseau 2011) similarly show that longer working hours are associated with higher Carbon footprints. Using various scenarios for working time reduction (including reduction of weekly hours by 20 %, longer parental leave, one year sabbatical) modelled on existing best practice policies in Belgium and the Netherlands, Pulliger (2012) estimates that a work time reduction by 20 % for all full-time workers would lead to a reduce GHG emissions from working-age households by 4.2 % in the UK and 6.4 % in the Netherlands (Pullinger 2012, 251).

Nässén and Larsson (2010) combine time use data for 1000 Swedish individuals and expenditure data from 1500 Swedish households to assess effects of work time reductions. They conclude that a decrease in work time by 1 % reduces energy use and GHG emissions by about 0.8 % - with the reduction being slightly higher in low-income households and slightly lower in high-income households. The authors also examine how the reduced work time is spent instead. They conclude that the ‘time effect’ does in fact increase energy use (although with the important caveat that reduced commuting to work is not included in the study which assumes that people work less hours per day rather than less days per week). However, the negative impact of the time effect is an order of magnitude smaller than the positive income effect (Nässén and Larsson 2010, 10).

3.6.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Land use is included in the environmental footprint, lower levels of which have been associated with lower working hours. However, we have not identified any detailed studies that look only at land use impacts. Given the basic importance of food which mostly determines land use impacts, one may expect reductions in land use to be somewhat lower than reductions of material use and greenhouse gas emissions.

3.6.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Freshwater use is included in the environmental footprint, lower levels of which have been associated with lower working hours. However, no detailed studies exist that would looking only at the impact on freshwater use. Given that freshwater is directly or indirectly used in most production processes, reductions in freshwater use can be assumed to be similar in magnitude to overall reductions in environmental footprint.

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3.7 Skill enhancement programme

3.7.1 Overview

The purpose of this programme is that resource efficiency would be incorporated into relevant academic and vocational curricula (economics, engineering, marketing, architecture, design, business accounting, land management, craftsmen, etc.). It would also aim at the training professionals to develop skills and techniques relevant for implementing resource efficiency measures in existing firms or developing new business models. The target is both white collar and blue collar workers to provide necessary competence and change in leadership and management, as well as the reallocation of workers to different roles needed for decoupling.

3.7.2 Assumptions made, and identified conditions necessary to generate an environmental impact

We have based our assessment on the information set out in the fiche describing the policy, and on the description in Ekval et al (2015) – in particular, we assume that sufficient funding is made available for the programme, and that it is targeted effectively at training that improves the sustainability and resource efficiency skills of relevant workers.

3.7.3 Change in the extraction of raw materials

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

Skill enhancement could certainly aid the transition but the actual effects of such a skills programme are extremely difficult to assess. Comparative policy analysis for vocational education and training (VET) is undeveloped and there are very limited data available (OECD, 2010). However countries such as Germany, with strong VET systems, have been quite successful in tackling youth unemployment (OECD, 2010). The success of VET's can be measured by comparing the level of employment and earnings between "completers and non-completers" of courses. A study in the UK showed that after 7 years VET completers, compared to non-completers, were earning 3-10% more (with higher level achievers earning more) and an employment boost of up to 8% (Patrignani and Conlon, 2011).

If one extrapolates this to a possible resource efficiency programme then there is cause to conclude that the effects could also be quite strong. That is to say the potential for successfully training people could be assumed to be quite high. However, within the chain of effects one also has to consider the participation rates of skills training courses and whether any such courses could be compulsory. Next is the actual desire to apply what one has learnt and then the actual potential for creating change within the position that a person holds, and finally the system or market that they operate within.

It is not scientifically possible to make the jump to actually judge how this could affect the reduction of raw materials extraction. But it is possible to say that skills and training are a necessary prerequisite for a successful transition to a resource efficient economy. It is therefore possible at least to say that a well-developed and applied skills enhancement programme could make a strong contribution to a reduction in raw materials extraction if the correct policy environment was in place to enable its application.

However, it is questionable whether the proposed programme would go far enough as resource efficiency and/or sustainability should be a fundamental pillar of all education. It is proposed that the most effective approach to develop the necessary skills for change across society is to integrate sustainability literacy onto all courses in all disciplines (HEPS, 2004). Hence the current proposed skills enhancement programme, proposing to “mainstream resource efficiency into relevant academic and vocational curricula” may be too weak to achieve the necessary societal transformation. One could in addition argue that it is not only practitioners that require education but also the customers and consumers that must understand, accept and purchase new products and services.

3.7.4 Change in the GHG emissions

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

A similar argument applies to the previous section in that a well-developed and applied skills enhancement programme could make a strong contribution to a reduction in GHG emissions if the correct policy environment was in place to enable its application.

3.7.5 Change in the global land requirement required for EU consumption and production.

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

Again, a similar argument applies to the previous section in that a well-developed and applied skills enhancement programme could make a strong contribution to a reduction in land use if the correct policy environment was in place to enable its application.

3.7.6 Change in the water use

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

Again, a similar argument applies to the previous section in that a well-developed and applied skills enhancement programme could make a strong contribution to a reduction in water use if the correct policy environment was in place to enable its application.

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3.8 A “circular economy tax trio” – taxes on the extraction and imports of selected virgin materials and on landfilled and incinerated waste

3.8.1 Overview

The fiche introduces taxes on use of virgin materials, landfills and waste incineration. This would aim to: reduce raw virgin resources extraction, encourage recycling and internalise externalities. The materials covered would in particular be: marble, chalk, dolomite, slate, limestone, gypsum, sand, gravel and metals. The revenues would be utilised by supporting research such as the UK’s WRAP programme. There would be no tax on metal imports or any exemption for exports.

3.8.2 Assumptions made, and identified conditions necessary to generate an environmental impact

We have worked on the basis of the information set out in the fiche describing the policy; and in particular have assumed that the tax on raw materials would be levied on EU extraction industries.

3.8.3 Change in the extraction of raw materials

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BAU)

One of the best examples of a similar instrument is the UK aggregates tax that was combined with a landfill tax. Use of primary aggregates in the UK fell by almost 50% from a peak of 262 MT (BDS 2009) in 1990 to 145.9 MT in 2011 (BGS, 2011). The combination of the aggregates levy and an escalatory landfill tax appears to have encouraged the use of recycled and secondary materials. From 1990 to 2011, the use of recycled and secondary materials (by-products from other processes) increased by four fold, from 10 MT to 54 MT and 2 MT to 8 MT respectively (MPA 2012; BDS. 2009).

A comprehensive natural resources tax introduced in Latvia in 1991 covers the extraction of bulk materials, waste disposal and emissions to air (Ferdrigo et al, 2013). Although data is limited, the tax seems relatively ineffective. It shows only a weak relative decoupling at best of dolomite and sand and gravel mix from economic growth (Withana et al, 2014). However, this is probably due to a weak price signal with the rate being unchanged from 1996 until 2009.

Hence used in the combination of an incentive at both the extraction and disposal can have a significant effect if applied at the correct rate and in an escalatory fashion that sends a clear signal, whilst allowing time to adjust. This effect would be felt mostly on bulk materials and less on metals which are relatively inelastic to price changes. In addition, it have the effect of increasing imports as these will not be taxed, further weakening the effects. Therefore the effects on actual extraction would be minimal as particularly as the price of metals is fairly inelastic.

The effect of an incineration tax is less clear. There is no obvious correlation between incineration charges and the quantity of municipal solid waste being sent to incineration (Watkins et al., 2012).

3.8.4 Change in the GHG emissions

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

Through reducing the extraction of materials in the EU the tax trio could reduce GHG emissions. The aggregates tax in the UK reduced life cycle global warming potential (not including the construction or use phase) from 3.38 MT CO2-eq to 1.80 MT CO2-eq per year, a reduction of 46.7% (Fedrigo-Fazio et al., 2014).

This is likely to be felt most for bulk materials rather than for metals which are likely to further shift outside of the EU, potentially creating a marginal increase in overall transport.

About 3000 MT of non-metallic minerals (mostly construction minerals such as sand and gravel) were extracted in 2012 that accounted for about 40% of the domestic material input for Europe (Eurostat, 2014). If the figures from Fedrigo-Fazio et al. (2014) are extrapolated (295 MT of aggregates resulted in 3.38 MTCO2), this is only responsible for about 34 MTCO2, or 0.008% of the 4.2 GTCO2 emitted in the EU during 2012. However, if one considers both mining and quarrying, it is reported as representing 2% of GHG emissions for Europe (Eurostat, 2014b). Hence if the materials tax only effects bulk materials, it is likely to make only a minor contribution to the reduction of GHG in Europe.

The greatest opportunities for change lie in the manufacturing industry that accounts for 23% of Europe's GHG emissions. Typically remanufacturing represents only 1-2% of a country's turnover for the manufacturing industry (Lavery and Pennell, 2013). Comparing the remanufacturing rate of some top performers such as Xerox, Lavery and Pennell (2013) proposed a figure of 50% for a realistic potential (with today's technology) on remanufacturing rate. Whilst reuse can potentially save almost 100% of materials, energy and water, remanufacturing can save 85% and 80% for materials and energy respectively used in manufacturing (Steinhilper, 2006). However, assuming the tax is not applied to imports, there would be some scope for the manufacturing industry to shift their consumption to imported materials, thereby reducing the effectiveness of the policy, and creating risks of misallocation of resources and increased emissions from transport.

3.8.5 Change in the global land requirement required for EU consumption and production.

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The effects on land use are difficult to assess, but in general a lowering of extraction rates would mean that expansion of mines and quarries within the EU are slowed, thereby reducing overall land requirements. The effects are likely to apply primarily for bulk materials.

3.8.6 Change in the water use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The use of water is not directly targeted and there will not be significant effects on the use of water through this tax trio. There may be some slight reduction in water use in quarries, for example water used for dust suppression, due to reduced extraction rates. An increase in material efficiency in manufacturing would also result in an associated improvement in water efficiency.

3.8.7 Other environmental impacts

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

The policy will not encourage the use of less damaging metal ores (e.g. due to ore concentration or co-contaminants such as acid mine drainage causing elements) and hence overall environmental impact will not be reduced. On the contrary the policy could promote imports of metals with increased transport, and that have been mined in countries with less stringent environmental regulations, therefore increasing overall pollution.

Local environmental benefits can be significant both from reduced mining and from reduced effects of waste management. In the case of UK aggregates, research has shown that the reduction of environmental impacts for a range of indicators such as ecotoxicity, human toxicity, acidification, eutrophication, human toxicity and ozone layer depletion was at least 30% and up to 80% (Fedrigo-Fazio et al., 2014).

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4 Land Use Policy Mix

4.1 Stronger and more effective environmental and climate dimension for EU land management in the CAP

4.1.1 Assumptions made, and identified conditions necessary to generate an environmental impact

This assessment assumes that (i) the limited environmental benefits likely to be generated by the 2013 CAP reform are delivered, and that (more importantly) a new CAP from 2020 has a much stronger set of requirements in terms of environmental sustainability, particularly including:

- Enhanced protection of high nature value farming areas;
- Requirements at national or regional level for reducing net GHG emissions from agriculture, accompanied by farm level requirements on improved targeting of fertiliser application;
- A range of other environmental enhancements, as set out in the policy fiche.

Some of the other policy measures in the land use policy mix are important for maximising the environmental impact of this measure, in particular:

- On the production side, measures aimed at improved pesticide management and reduced pesticide use; and measures aimed at improved management of the nitrogen cycle.
- on the consumption side, measures aimed at reducing effective demand for livestock products, and aimed at reducing food waste general.

In addition, some accompanying measures appear to be important, particularly research and innovation, and then outreach to farm businesses, on sustainable intensification of agriculture.

4.1.2 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

Limited impacts are expected; although improved efficiency of fertiliser use could lead to reduced fertiliser consumption, and thus (i) reduced consumption (and therefore reduced extraction) of potassium and phosphorus; and (ii) a reduction in the energy demand linked to fertiliser manufacture. The risk of rebound effects (for example, increased efficiency leading to the same or greater quantities being used in order to achieve significant yield increases) should be limited, particularly by a combination of tighter requirements on nitrous oxide and nitrates emissions under the CAP itself, and by the measure on reduced emissions and improved nitrogen management (see separate assessment). The Commission's impact assessment for the 2011 CAP reform proposals noted that "Using precision farming systems can lead to a reduction of 30% in fertiliser use. No fertilisation in autumn and winter might lead to a reduction of emission from crop residuals between 8 (other arable land) and 40% (sugar beet). The decrease in fertiliser depends on manure type, use of manure in spring and other variables." It is expected that the CAP reform proposals agreed in 2013 will have minimal impacts on fertiliser use; with significant potential therefore for future reforms to encourage the uptake of precision farming approaches to fertiliser use.

4.1.3 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

The impact of a more effective environmental and climate dimension in CAP is likely to drive down both net climate emissions per se, and particularly net climate emissions per unit of output. It is unlikely, given limitations on measurement of carbon sequestration and emissions of GHGs from agriculture, for quantified limited to be set and enforced for agricultural holdings claiming CAP payments. However, if specific requirements in terms of the farming methods used are set, and enforced, the impacts could be significant. In particular, emissions reductions could be achieved through (i) reduced use of inorganic fertiliser (including some upstream emissions savings, as noted above); (ii) more targeted use of both organic and inorganic fertilisers leading to reduced emissions of nitrous oxide, as a result of greater nitrogen uptake in soil; and (iii) reduced methane emissions from improved livestock farming techniques.

However, as noted in the fiche and in the policy mix chapter, there are some countervailing risks, particularly in relation to emissions outside the EU: if the practical impact of tighter environmental controls and greater environmental conditionality on CAP payments is a reduction in yields, and a reduction in production, without any corresponding reduction in EU consumption of GHG-intensive agricultural products, then EU demand will need to be supplied through imports of products from other agricultural economies; which could, in turn lead to increased emissions and pressure on land use in those economies (including, for example, the impact of on deforestation of expanded agricultural production in economies with significant tropical and sub-tropical forest). It is therefore likely to be important that policy is developed in ways that avoid significant yield penalties (there would, in any case, be significant practical political difficulties in introducing measures that led to reduced production); and accompanying policy, including advice to farm businesses, could usefully focus on improved understanding of sustainable intensification in agriculture, in both organic and non-organic farming systems, in order to ensure that environmental objectives are delivered without significant negative impacts on production.

The impact of rules on the protection of grassland under the CAP 2013 legislation will require careful evaluation. In principle, it should have a positive impact on net sequestration, by reducing the loss of soil carbon when grassland is converted to other uses, particularly through ploughing to convert to arable use. However, extensive pasture farming of livestock can have a higher impact in terms of GHG emissions per unit of production.

The inclusion in a more ambitious post-2020 CAP of rules or incentives aimed at improving carbon sequestration in arable soils – for example, through reduced tillage or zero-tillage approaches could have further potential for improving overall levels of carbon sequestration.

4.1.4 Assessment of impact on land use

Estimated magnitude of change: ↗ or ↘ : Medium low (between 10-50% deviation from BaU)

The impact on land use is difficult to assess in aggregate terms, since specific requirements would be likely to be targeted at particular types of land (for example, improved identification and protection of high nature value areas; requirements for the rewetting of carbon-rich peat soils; etc.). Overall, there is likely to be reduced pressure for production on sensitive areas and sensitive soils. Given the issues noted in the policy mix, and noted above, about the risks for non-EU land use associated with reduced EU production without an accompanying reduction in EU consumption of livestock products, this can be expected to lead to some land use increase elsewhere. The extent to which more appropriate management of sensitive environments can be accompanied by an environmentally sustainable improvement in yields from other EU agricultural land is therefore an important issue for the success of the policy.

4.1.5 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

It is unlikely that the measure would have a significant direct impact on the quantity of freshwater consumed by the agriculture industry; it would be difficult to envisage rules and policies being set on water consumption at EU level in a sufficiently targeted way to reflect the local realities of farming methods, water availability, and impacts on water quality. However, a combination of the reduced use of inorganic fertiliser, and more targeted use of fertilisers generally, under a more environmental CAP, together with other measures in the policy mix (particularly pesticide reduction targets, and revised emissions under the National Emissions Ceilings Directive), can be expected to lead to a significant reduction in negative impacts of the agriculture industry on water quality, and therefore on water availability. This may, in turn, depending on local circumstances, reduce the pressure generated by water abstraction demands on sensitive habitats.

4.1.6 Assessment of biodiversity impacts

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

A key element in a more environmentally sustainable Common Agriculture Policy would be the introduction of more powerful incentives, and stricter enforcement penalties, to ensure the appropriate management of sensitive habitats, particularly those protected under the Habitats Directive, and more appropriate management of features relevant to protected species. The EEA's 2010 biodiversity assessment noted that "Decreases in the diversity of crops, the simplification of cropping methods, use of fertilisers and pesticides and the homogenisation of landscapes all have negative effects on biodiversity in agricultural areas"; and also notes the biodiversity risks associated with the abandonment of agricultural land in some areas. A shift in subsidy payments away from generic per hectare payments, and towards payments targeted more at specific sensitive areas and the practices required in those areas, could therefore have significant benefits.

4.1.7 References

As per the policy fiche, and deliverable 4.2, with the addition of:

European Commission, 2011: Commission Staff Working Paper “Impact Assessment: Common Agricultural Policy towards 2020”, SEC(2011) 1153 final/2

Annex 2A: Fact sheet – Biodiversity and Agriculture;

Annex 2B: Assessment of selected measures under the CAP for their impact on greenhouse gas emissions and removals, on resilience and on environmental status of ecosystems.

European Environment Agency, 2010. EU 2010 Biodiversity baseline. EEA Technical report No 12/2010.

4.2 Development of food redistribution programmes/food donation to reduce food waste

4.2.1 Short description of the policy

This policy instrument aims to reduce the generation of food waste through the development of food redistribution programmes, with the additional benefit that food donation provides a crucial support for the most deprived groups in society. The measure would be targeted at the retail (including food restaurateurs) and food supply chain sector and would target the retail/use and disposal phase of food products. It would encourage householders, retailers and other relevant food stakeholders to donate eligible food products to food distribution programmes.

4.2.2 Assumptions made, and identified conditions necessary to generate an environmental impact

Data suggests that food waste is a major issue in the EU. One pan-European study on food waste (BIO Intelligence Service, 2010) estimated that 89 million tonnes of food (about 180 kg per person) is wasted each year in the EU, excluding the primary agricultural and fisheries production phases of the supply chain. In the UK, food manufacturing, distribution and retail generate 4.3 million tonnes of food waste each year (WRAP, 2015). Food production and food waste have major environmental impacts in terms of ghg emissions, land use and freshwater use (see sections below). In the EU, food production and consumption generate an estimated 20% to 30% of all EU environmental impacts (European Commission, 2014).

Food redistribution is one way of preventing food from become waste, by redistributing it to charities, organisations or individuals who can use it. However, it is necessary to do this in collaboration with retailers, suppliers and redistribution organisations to ensure that there are no impacts on food safety or brand integrity. It has been estimated that in the UK, only around 2% of surplus food generated by food retailers, manufacturers and suppliers that is fit for consumption is currently redistributed, with 98% composted, incinerated for energy recovery, or landfilled (Carr and Downing, 2014).

In 2013, the European Federation of Food Banks (FEBA), which brings together 256 food banks in Europe, redistributed 402,000 tons of food, equivalent to 804 million meals, to 5.7

million people in partnership with 31,000 charitable organisations and social centres in Europe (FEBA, 2015). In 2013 FareShare, one of the largest food redistribution charities in the UK, received and redistributed 7,150 tonnes of food which would otherwise have gone to waste (this represents only about 1.5% of the 300,000 to 400,000 tonnes of surplus food believed to be available in the UK annually) (BIO Intelligence Service, 2014). The Company Shop organisation in the UK set up a pilot Community Shop in Goldthorpe (Yorkshire) in 2013 to provide surplus food from large stores to shoppers on the cusp of food poverty at up to 70% less than normal prices. The pilot store has redistributed around 400 tonnes of surplus food to date, and Company Shop hopes to open around 20 stores nationally, which could deliver over 20 million meals to 10,000 members (WRAP, date unknown). A trial by two UK redistribution charities (FareShare and FoodCycle) and several retailers found that on average, around 35kg of surplus food could be collected on each store visit (enough for around 86 meals); the categories of food with the highest surpluses were bakery and produce, which suggests that redistribution could have an impact on the amount of land and water resources used to produce such foodstuffs (WRAP, date unknown).

The majority of food waste arises at the manufacture and distribution phases, but some high value food waste also occurs at the retail stage, or 'back of store'; this waste offers a particular opportunity for retailers, through local charities, to redistribute surpluses of quality food quickly before they become waste (WRAP, 2015).

4.2.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (i.e., less than 10% deviation from BaU)

Since the extraction of raw materials in terms of the DYNAMIX project relates mainly to the use of virgin metals, the impact of this policy instrument in this area is likely to be negligible.

4.2.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗↗ Medium high (ie between 50-100% deviation from BaU) or ↗↗ Medium low (ie between 10-50% deviation from BaU)

A reduction in the generation of food waste as a result of food redistribution is likely to have an associated impact on greenhouse gas emissions. Firstly, it may result in a reduction in the total amount of food produced (since less is wasted), which would mean that emissions associated with food production are reduced. Secondly, it may result in less food waste going to landfill (where it emits methane) or being incinerated for energy (which produces associated emissions).

The global carbon footprint of food produced and not eaten has been estimated at 3.3 billion tonnes of CO_2 equivalent, making food loss and waste the third top emitter of ghg emissions after USA and China (FAO, 2013). In 2011, it was estimated that avoidable food waste in the UK was responsible for ghg emissions of 20 million tonnes CO_2 equivalent per year (accounting for its whole life cycle), which is equal to around 3% of the UK's domestic ghg emissions, or emissions from over 7 million cars (Carr and Downing, 2014).

Each tonne of food waste prevented results in 4.2t of CO2-equivalent emissions avoided compared with landfilling, or 500kg avoided for each tonne processed through anaerobic digestion, or less for other options (e.g. composting or other energy recovery) (Defra and DECC, 2011). Therefore food waste prevented through redistribution could generate avoided emissions in that order of magnitude.

4.2.5 Assessment of impact on land use

Estimated magnitude of change: ↗↗↗ Medium high (ie between 50-100% deviation from BaU) or ↗↗ Medium low (ie between 10-50% deviation from BaU)

A reduction in the generation of food waste as a result of food redistribution may have an associated impact on land use, since it may result in a reduction in the total amount of food production required (since less is wasted). This may mean that some land that was previously used for food production is no longer required for that purpose.

According to the FAO, around 1.4 billion hectares of land used annually to produce food are lost or wasted (FAO, 2013) – only a little less than the total land area of Russia. Meat and milk waste accounts for around 78% of the land occupation of food waste, even though their contribution to total food wastage is only around 11%; the next biggest contributor in terms of land occupation is cereals with around 9% (and around 26% of food waste) (FAO, 2013). This suggests that if the amount of meat, milk and cereals wasted could be reduced, the amount of land required for producing them could be reduced by a proportional amount.

4.2.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗↗↗ Medium high (ie between 50-100% deviation from BaU) or ↗↗ Medium low (ie between 10-50% deviation from BaU)

A reduction in the generation of food waste as a result of food redistribution may have an associated impact on freshwater use, since it may result in a reduction in the total amount of food produced (since less is wasted). This may mean that the amount of freshwater needed for the production of food (e.g. crop irrigation, water used during processing of foodstuffs) is reduced.

Globally, around 70% of water used is for food production (this includes both direct and indirect use) (Asian Development Bank, 2013). According to the FAO, around 250km^3 of water used annually to produce food are lost or wasted (FAO, 2013). The table below (Hoekstra, 2014) shows estimates of the water footprint (i.e. the amount of water needed to produce a specified amount) of various foodstuffs; a kg of any food waste avoided would therefore result in that amount of water not being required and being ‘saved’. For fruit and vegetables this may be between 130 and 1,600 litres, for meat between 3,900 and 15,500, and for grains/cereals between 900 and 3,400 litres.

Table 1. The water footprint of different food items.

Food item	Unit	Global average water footprint (litres)
Apple or pear	1 kg	700
Banana	1 kg	860
Beef	1 kg	15,500
Beer (from barley)	1 glass of 250 ml	75
Bread (from wheat)	1 kg	1,300
Cabbage	1 kg	200
Cheese	1 kg	5,000
Chicken	1 kg	3,900
Chocolate	1 kg	24,000
Coffee	1 cup of 125 ml	140
Cucumber or pumpkin	1 kg	240
Dates	1 kg	3,000
Groundnuts (in shell)	1 kg	3,100
Lettuce	1 kg	130
Maize	1 kg	900
Mango	1 kg	1,600
Milk	1 glass of 250 ml	250
Olives	1 kg	4,400
Orange	1 kg	460
Peach or nectarine	1 kg	1,200
Pork	1 kg	4,800
Potato	1 kg	250
Rice	1 kg	3,400
Sugar (from sugar cane)	1 kg	1,500
Tea	1 cup of 250 ml	30
Tomato	1 kg	180
Wine	1 glass of 125 ml	120

In the UK, the total water footprint of household food waste is 6,262 million cubic metres per year, of which 5,368 million cubic metres per year/243 litres per person per day (8% of the UK's total water footprint) is avoidable and a further 894 million cubic metres/41 litres per person per day (1% of the UK's total water footprint) could possibly be avoided (WRAP and WWF, 2011).

4.2.7 References

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4.3 Targeted information campaign to influence food behaviour towards: reducing food waste and changing diets

4.3.1 Introduction

This measure is an awareness campaign that aims to encourage and achieve a reduction in food waste and a change in diets. The measure would provide information on the serious issue of food wastage in order to increase respect for food, to decrease wastage of food as well as to promote healthy, more environmentally friendly and less resource intensive diets. Advice and guidance could also be provided to consumers on how they could more efficiently consume food by providing information and tips on shopping, shelf life, storage, preparation, recovery and disposal options.

Depending on the scope and quality of the information campaign, e.g. how well the campaign is disseminated in terms of the number of people it reaches and the means and channels of dissemination used, environmental impacts related to the reduction in food waste and changes in dietary habits could be expected.

4.3.2 Assumptions made, and identified conditions necessary to generate an environmental impact

The basic assumption made here is that a well-prepared and targeted information campaign will lead to a change in consumer behaviour towards a reduction of meat and dairy product consumption and avoidance of food waste. Few studies have been carried out to evaluate the effectiveness of information campaigns, especially in regard to campaigns and initiatives aiming to reduce meat and dairy product consumption as well as to reduce food waste (EC 2010; Dibb & Fitzpatrick, 2014). In general, the effect of information campaigns is difficult to measure (Vittersø & Tangeland, 2015). Some relevant findings can be extrapolated from evaluations of public health campaigns (e.g. EATWELL, 2012; Snyder, 2007). It can be stated that information measures overall have a small but positive effect on healthy eating. Snyder (2007) found that well-prepared information campaigns on health issues have an average effect size of about 5 percentage points. This means, for example, that if 60% of people were engaging in the target behaviour before the campaign, about 65% can be expected to do so after the campaign. Regarding food waste, based on the UK “Love Food Hate Waste” campaign, the food waste prevention potential of a targeted waste campaign can be estimated at 3% of avoidable food waste (BIO, 2010). As information campaigns are relatively cheap compared to other policy instruments, they are also generally cost-effective.

Furthermore, it is assumed that a decrease in the demand for meat, dairy products and eggs – (co-)triggered by information campaigns – will lead to a reduction in the number of livestock being reared. This would generate a number of positive effects for the environment, which are described in detail in the following sections. However, it is contestable whether the expected, rather small reduction in the demand for meat and milk products would effectively translate into a reduction in the livestock production. Instead, the changes in demand could possibly lead to an increase of exports of meat and milk products produced in Europe.

In order to support the assumption of changes in the production patterns (reducing livestock rearing in favour of other land uses), it can be argued that the avoided animal products will be substituted by other food items. Thus, at the same time as the demand for meat, milk and eggs is reduced, an increase in the demand for cereals or other vegetarian products can be expected, including, inter alia, for soybeans with the environmental impacts associated with large-scale soybean cultivation (see below section 4.3.5).

Moreover, the targeted information campaign could have the effect that, while overall demand for meat and dairy products decreases, the demand for high quality livestock products increase, e.g. for products from organic agriculture and free range husbandry. While this should be considered a positive tendency, extensive agricultural production practices to feed a larger share of population may yield greater demand for space/land compared to conventional production practices.

With regard to food waste, we assume that the measure achieves a reduction of food wastage in households, leading to a slight decrease in the demand for agricultural products. This is assumed to result in a decrease of agricultural production.

4.3.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

The measure can be expected to have effects on the extraction of raw materials as regards mineral fertilizer use. Reducing food waste and thus reducing the overall amount of food which has to be produced would presumably lead to a decrease in fertilizer use. This, in turn, leads to a reduction in the consumption of phosphorus and potassium, which are finite resources (Grizetti et al. 2013; Odegard & van der Voet 2014). Furthermore, a reduced need for mineral fertilizers involves reductions in the energy consumption associated with fertilizer production and transport (Cordell, Drangert, & White, 2009; Sutton et al. 2013). This would simultaneously reduce further negative environmental effects linked to mineral extraction, e.g. waste generation or water pollution.

Furthermore, reducing food waste requires less packaging materials for processed food, hence possibly leading to reduced demand for materials used for packaging. In addition, energy demand could be reduced in the course of decreasing need for packaging, and also for food production and processing. A reduction of meat and dairy consumption could also lead to energy savings, as plant-based products require less energy for processing and transport than livestock products (Stehfest et al. 2009).

The expected reductions in the use of mineral fertilisers made us estimate that there will be a low beneficial change compared to the BAU.

4.3.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

As described above, information campaigns are expected to only have small effects on people’s behaviour (see also Umpfenbach, 2014). However, in regard to greenhouse gas emissions, even small changes in behaviour can have a significant effect due to the large direct and indirect greenhouse gas (GHG) emissions associated with both meat consumption and food waste. Thus, if the change in behaviour were only a roughly 10% reduction in meat consumption, accompanied by shifts to less GHG-intensive meat, together with a reduction in wastage of meat and dairy products, the impact on GHG emissions associated with food consumption could be expected to be above 10%.

Within the food production sector, meat and dairy products account for the largest share of GHG emissions globally (Garnett, 2009). Assuming that a decreasing demand for meat and milk, resulting from the information campaign, effectively leads to a reduction of livestock, a reduction of various GHG emissions could be expected. First, methane (CH_4) emissions caused by the digestive process of ruminants would be reduced. Second, a reduction in the number of livestock would lead to a reduction of manure, which in turn would decrease emissions of nitrous oxide (N_2O) (Stehfest et al. 2009; Westhoek et al. 2014). Further effects could be reductions in CO_2 emissions related to food processing, transport or production of mineral fertilizers needed to grow feed crops. The production of phosphorus and nitrogen fertilizers in particular involves significant carbon emissions. For the former, the transport of rock phosphate to production sites as well as the transport of the products around the world causes GHG emissions (Cordell, Drangert, & White 2009), while for the latter, the industrial manufacturing by means of the Haber-Bosch process is very energy-intensive (Sutton et al. 2013). However, GHG emissions caused by the pre-farm and post-farm stages are generally perceived to play a comparably minor role (Garnett 2009, Westhoek et al. 2014).

Westhoek et al. (2014) examined the effects of halving the consumption of meat, eggs and dairy products in the EU by means of biophysical models and methods. Considering only methane emissions, the authors concluded that a reduction of 50% of livestock in the EU would lead to a reduction of 108 million tonnes CO_2 equivalents per year. Overall, a 25-40% reduction of greenhouse gas emissions can be expected. An information campaign would probably lead to a much lower reduction in meat and milk consumption; hence GHG emission savings would be lower.

Garnett (2009) points out that livestock farming involves further indirect CO_2 emissions, caused by the conversion of forest into cropland for feed production. In particular, soy is a popular feed crop in the EU and is largely imported from South American countries. In Brazil, for example, the need for land to cultivate soy for exportation is a major driver of deforestation of the Amazon rainforest, which results in considerable GHG emissions (Garnett 2009)(Garnett, 2009, S. 493-494). Reducing the demand for meat and thus the demand for feed crops could reduce the driving forces for land conversions.

Apart from meat consumption, the information campaign targets the avoidance of food waste. Food waste generates environmental impacts throughout its life cycle: during food production, harvesting, transport, packaging and finally, treatment of food waste. Taking on the life cycle perspective, it is estimated that at least 1.9 tons of CO_2 equivalents per tonne of wasted food are emitted in the EU. In 2006, emissions associated with food waste amounted to 170 million tonnes CO2 equivalents for the EU27, which corresponds to approx. 3% of total GHG emissions (EC 2010). Moreover, Parry, James and LeRoux (2015) found that in regard to reducing GHG emissions, avoiding food waste is much more effective than any form of food waste treatment. Treating one tonne of food waste in a landfill causes 536 kg of CO_2 equivalents.

The potential for emission savings also depends on how the land previously used for livestock production or food going to waste will be used. Returning vegetation would lead to further reductions in emissions, while other uses might cause GHG emissions.

The various GHG emissions associated to both food waste and meat and dairy consumption leads us to the conclusion that a targeted information campaign can result in considerable emission savings. Thus we estimate the impact of the measure to be medium low.

4.3.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

The information campaign is likely to drive down the global land required for EU consumption and production. Both a reduction in the consumption of animal products and a reduction in food waste can be expected to lead to a decrease of the land needed for agricultural production – both within and beyond the EU.

A number of studies found that a shift towards healthier, more vegetable based diets leads to decreases in the land needed for agricultural production (Odegard, van der Voet 2014; Wirsenius, Azar & Berndes 2010; Stehfest et al. 2009). A reduction in livestock production would decrease the need for pasture land. For example, Stehfest et al. (2009) analyzed the effects on land of a transition towards a healthier diet, which encompasses an average daily per capita intake of 10 g beef, 10 g pork, and 46.6 g of chicken meat and eggs. On the global level, this would result in a reduction in pasture area by 1360 Mha compared to the reference case based on FAO projections. Additionally, decreases in livestock rearing would reduce the land needed to grow feed crops (Westhoek et al. 2014; Garnett 2009). In their analysis of the effects of cutting the meat and dairy intake in Europe, Westhoek et al. (2014) make the assumption that feed from grassland is favored over feed produced on cropland. Based on this assumption the authors come to the result that a reduction of 50% in meat and dairy consumption in Europe would lead to a reduction of 90% of feed grown on arable land. Furthermore, the decrease of land needed for feed production would be larger than the increase of land needed to grow food cereals to substitute the renounced meat and dairy products.

A further aspect to be considered is indirect land use change. Soy cultivation for feed production is a major cause for indirect land use changes. As cropland uses up large areas of arable land, this indirectly provokes the conversion of natural areas (e.g. rainforests) into other land uses such as smallholder farming or cattle rearing (Garnett, 2009).

For land use, a positive, albeit small impact is expected due to the reduction of land needed for livestock farming (as pastureland, and in terms of the land needed to grow feed crops) as well as a reduction of land associated with food waste. There are some risks that a shift to consumption of organic products, while it would have other environmental benefits, would create a small countervailing increase in land use, given the productivity penalty associated with many practises in organic farming.

4.3.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

By avoiding food waste, a reduction of freshwater needed for the irrigation of crops can be achieved. Also, livestock farming is associated with significant water use (Steinfeld et al. 2006). While the quantity of water needed as drinking water for the animals is comparably low, high quantities are needed for the irrigation of feed crops (Odegard and van der Voet 2013). The reduction in freshwater needed for livestock farming due to a dietary shift might in part be compensated by an increase of water needed for irrigation of food crops, as the reduced meat and dairy product consumption may be substituted by the consumption of cereals, pulses or other plant-based food. However, this increase in water use is likely to be small compared to the water savings.

At the same time, the measure can be expected to decrease the pressure on water quality caused by diffuse pollution. Major pollutants are nitrate and phosphor compounds, stemming for example from manure or fertilizer use. Both a reduction of food waste – which includes a reduction of waste of meat and dairy products – and a reduction in the consumption of animal products would contribute to a reduction of pollutants. Westhoek et al. state that a reduction in livestock production would result in a “significant decrease of reactive nitrogen input and losses across Europe” (Westhoek et al. 2014, p. 200). Grizetti et al. (2013) found that approx. 0.7 Tg of nitrogen per year are emitted to water bodies in the EU due to food waste. The nitrogen is mostly emitted in form of NO_3, which contributes to eutrophication in coastal waters. Grizetti et al. argue that a reduction in food waste would decrease nitrogen emissions with positive effects for aquatic ecosystems. Besides, a decrease of nitrogen emissions could reduce the threat on drinking waters located in agriculturally used areas (Grizetti et al. 2013).

Although the impact of the expected changes in livestock farming is difficult to assess, we estimate that the measure is likely to slightly decrease the freshwater use, as savings can be achieved by food waste avoidance. In addition, the measure indirectly affects freshwater availability by decreasing negative impacts on water quality.

4.3.7 Lessons learnt

- The Business as usual scenario encompasses no information on development on food production / consumption
- Information is missing on whether info campaign promotes reduction of food intake or substitution by certain other products
- More information is needed on the format of the information campaign: Success in raising awareness = prerequisite for reducing negative environmental effects → measured by the number of people the campaign reaches. Depends e.g. on language, format, targeting the right audiences, etc.
- Success factors for food waste / changing diets campaigns encompass:
    - Food waste = avoidable costs for the consumer → changing behaviour can save costs
    - Less meat / eggs / dairy products = increasing health, as it equals a reduction of the intake of saturated fats. Overconsumption of meat and other animal products is seen as a major cause for obesity and coronary heart diseases (Vranken et al. 2014, Westhoek et al. 2014). → changing behaviour can generate health benefits
    - Also relevant for changes in consumer behaviour: concerns for animal welfare (compare Vranken et al. 2014, p. 96).
    - In general: for higher income (which is related to higher degree of education and awareness) there is a tendency seen for a decrease in meat consumption (Vranken et al 2014)
- Interrelation of the two topics addressed in the information campaign (on the one hand, reducing food waste, and on the other hand, reducing the consumption of meat and other animal products): reducing food waste contributes to the target of reducing meat consumption (Grizzetti et al. 2013).

4.3.8 References

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4.4 Regulation for Land Use, Land Use Change, and Forestry (LULUCF)

4.4.1 Short description of the policy

Currently, EU Member States' delivery of their greenhouse gas reduction commitments includes only emissions from emitting sectors. Those sectors are covered either by the EU Emissions Trading System (EU ETS) which regulates emissions from large carbon-emitting plant, particularly carbon-based power plant, and from aviation; or by the Effort-Sharing Decision (decision 23/04/2009) which covers emissions from other sectors (heat; transport; agriculture, etc.). Land Use, Land Use Change and Forestry sectors (LULUCF) refer to a category of emissions covered by the UNFCCC and the Kyoto Protocol which may either be a net sink (through sequestration of carbon in forests and in soil) or a net source of emissions

The proposed policy would involve including the LULUCF sector in the EU's mix of policies to achieve emissions reductions. Kuikman et al[4] consider options including incorporating the sector in the Emissions Trading System (EU ETS), and incorporating it into the Effort Sharing Decision (which governs Member State emissions outside the EU ETS), but suggests that a separate pillar covering LULUCF emissions may be preferable. A European Commission consultation on the options for integrating LULUCF emissions into the EU's 2030 climate and energy policy framework[5] also includes an option which would create a combined Agriculture and Land Use sector, bringing together direct emissions from agriculture (principally of methane and nitrous oxide), with sequestration from land use, in a single sector.

Regardless of the specific design chosen, three elements underpin our preferred policy: (i) that overall mitigation targets for the EU should be increased to reflect the availability of mitigation in the LULUCF sector; (ii) the non-permanent nature of carbon sequestration is fully recognised, with binding requirements on Member States to make up any future carbon losses; and (iii) genuine incentives are created for Member States to introduce policies to encourage additional mitigation activity in the sector.

4.4.2 Assumptions made, and identified conditions necessary to generate an environmental impact

The key assumptions made are that the three elements referred to above are incorporated into policy design. There are, however, some risks that Member State and sectoral industry pressure will lead to carbon sequestration from sinks being made available to cover Member State obligations to deliver emissions reductions from the current ESD sectors, without any adjustment to the overall mitigation target. This could lead to a net reduction in mitigation

[4] Kuikman et al, November 2011: "Policy options for including LULUCF in the EU reduction commitment and policy instruments for increasing GHG mitigation efforts in the LULUCF and agriculture sectors"; Synthesis report for the European Commission. Available online at: http://ec.europa.eu/clima/policies/forests/lulucf/docs/synthesis_report_en.pdf

[5] European Commission, March 2015: "Consultation on the integration of agriculture, forestry and other land use into the 2030 EU climate and energy policy framework"; online questionnaire at: http://ec.europa.eu/clima/consultations/articles/0026_en.htm

activity, since some removals of CO^2 through sinks are likely even under business as usual; and counting them towards delivery of the reduction targets would simply displace emissions reductions from other sectors.

Careful design of the instrument to secure the three elements identified above is therefore important; but not currently guaranteed.

4.4.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ (?) Low (less than 10% deviation from BAU)

There is a limited direct impact of LULUCF regulation on the extraction of raw materials, in that providing incentives for mitigation in the land use sector would make the opportunity costs of using land for the purposes of raw material extraction (particularly through open-cast mining) slightly greater; however, the costs in comparison to the economic benefits of extraction are expected to be negligible.

There may also be a small indirect impact, although the direction of this impact is difficult to determine, and depends heavily on the assumptions made about how a LULUCF regulatory instrument is developed. If LULUCF targets are included in the targets Member States are required to reach across sectors outside the emissions trading system, without an adjustment being made to those targets to reflect the additional potential for mitigation, the impact is likely to be to allow room for increased emissions in other sectors, including those involving use of raw materials. And some carbon mitigation measures in the forestry sector could have the effect of slowing the availability of timber for use in the wider economy, reducing the potential for replacement of more carbon-intensive materials.

4.4.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

Subject to effective design of the instrument, the impact on greenhouse gas emissions should be environmentally beneficial. Mitigation potential in the LULUCF sector exists in the form of (i) opportunities for carbon sequestration through forestry management; (ii) opportunities for afforestation; (iii) cropland management techniques (iv) grazing land management and (v) the restoration of degraded soils.

The EU's technical mitigation potential in soil carbon management has been estimated at 200 $MtCO_2/yr$[6]; however, some of these options have limited cost-effectiveness because of a negative impact on productivity in the short-term, which in turn could be expected to lead to increased land use (and hence, increased carbon emissions) elsewhere; and many of those which are cost-effective may already have been implemented since the publication of the

[6] Smith, P., Powlson, D.S., Smith, J.U., Falloon, P. and Coleman, K. (2000) Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. Global Change Biology, 6, 525-539.

study. A more recent study[7] suggests that the cropland management mitigation potential for the EU is around 67 $MtCO_2$/yr – equivalent to 1.5% of total EU emissions.

Afforestation and forest management potential is estimated by Kuikman et al to be around 120 $MtCO_2$/yr, and 45-60 $MtCO_2$/yr, respectively, once cross-sectoral GHG impacts (for example, reduced availability of timber as a result of lower . Taken together, these estimates suggest a potential of around 200 $MtCO_2$/yr.

4.4.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The impact on land use is likely to be broadly positive, by creating additional disincentives to the conversion of land to other uses; in addition most carbon mitigation measures from land use have side benefits of improved ecological health of land (see below under biodiversity impacts). There are some risks that if incentives

4.4.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ (?) Low (less than 10% deviation from BAU)

While afforestation measures may place some additional pressures on water resources, these are likely to be offset by the enhanced water management, reduced flooding, and enhanced water quality likely to result. Improved peatland management, where relevant, could contribute significantly to improved drinking water quality, reducing overall water resource pressures and reducing the resource costs of water treatment.

4.4.7 Biodiversity impacts

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The Intergovernmental Panel on Climate Change special report on LULUCF in 2000[8] notes that "any LULUCF climate mitigation project that slows deforestation or degradation will help conserve biodiversity." While the biodiversity resources present in EU forests and agricultural land are less significant than those in tropical and sub-tropical regions at risk of deforestation, the statement above is likely to hold true. As Zanchi et al[9] point out, monospecies afforestation has historically had negative impacts on biodiversity in a number of regions in the EU; however, support for forestry under rural development programmes has largely

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[7] Lesschen, J.P., Schils, R., Kuikman, P., Smith, P. and Oudendag, D. (2008) *PICCMAT: implementation of measures to mitigate CO_2 and N_2O from agricultural systems across EU27*. Wageningen. PICCMAT Deliverable D7.

[8] IPCC, 2000: "Land Use, Land Use Change and Forestry", ed R T Watson, I R Noble, B Bolin, N H Ravindranath, D J Verardo and D J Dokken.

[9] Zanchi, G et al, 2007: "Afforestation in Europe". Available at http://www.ieep.eu/work-areas/agriculture-and-land-management/future-of-the-cap/2007/01/meacap-afforestation-in-europe

overcome this risk in recent years, and the design of policies to encourage LULUCF mitigation which enhances biodiversity is not likely to pose a significant challenge to policymakers.

4.4.8 References

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4.5 National Emissions Ceiling Directive

4.5.1 Short description of the policy

The proposed policy would involve a combination of measures, including (i) a progressive tightening of member state targets under the National Emissions Ceiling Directive, particularly for nitrogen dioxide and ammonia; (ii) enhanced regulation of emissions at Member State level, backed if necessary by cross-compliance requirements under the Common Agricultural Policy; (iii) action under the CAP to incentivise compliance in ways which avoid productivity penalties, and (iv) further measures at Member State level to reduce emissions to air and water, including

4.5.2 Assumptions made, and identified conditions necessary to generate an environmental impact

The underlying assumption for this policy package is that it is possible to reduce emissions without a significant negative impact on productivity, or even with a positive impact on productivity, through improved understanding of the most effective approaches to nitrogen fertilisation, ensuring that more of the nitrogen applied is taken up in crops and soil, leading to reduced overall demand for fertilisers, and reduced availability of nitrogen for emissions to air and water. However, careful analysis of the reasons for sub-optimal nitrogen application will be needed in order to ensure that advice, training, and incentives are effective. The chapter on costs and benefits of nitrogen in the environment in the “European Nitrogen Assessment”[10] estimates that internalising the environmental costs of nitrogen would lower the optimal rate of application by farmers by around 50 kg/ha.

A particular challenge for the policy is that fertiliser use in agriculture involves a number of elements of risk management: farmers do not have the luxury of a hindsight view of the weather conditions relevant to their nitrogen application, both in the medium term (growing season) and in the short term (risk of run-off due to unexpected rain); and the response of crops to nitrogen is generally non-linear. Farmers are therefore likely to take a relatively risk-averse approach to application, and err on the side of excess nitrogen. Tackling this risk aversion, and enabling greater precision in choices on nitrogen application, is critical to the success of this policy.[11] Some factors, including increased uncertainty of weather patterns, may, however, make this challenging.

4.5.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

[10] The European Nitrogen Assessment, ed. Sutton et al, Cambridge University Press 2011; chapter 22: “Costs and benefits of nitrogen in the environment”, Brink and van Grinsven

[11] See Niggli et al, 2009

A minor impact is expected in terms of the extraction of raw materials, as a result of the expected reduction in demand for inorganic fertilisers, which would lead to a reduction in demand for the energy and materials required to produce it.

4.5.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The policy is intended to have an indirect impact on reduced greenhouse gas emissions through reduced availability of nitrogen, leading to reduced emissions of nitrous oxide, a gas with a Global Warming Potential assessed by the IPCC at 298 times that of CO_2, and to a lesser extent reduced emissions of methane, which has a GWP of 25 times that of CO_2, through better soil incorporation of organic fertiliser. It seems likely that both effects would be achieved. However, without greater certainty over the policy's ambition of delivering reduced emissions without a penalty in terms of productivity, there are risks that the constraints on nitrogen use by the agriculture sector in the EU would lead to reduced production in the EU (including both through reduced crop yields, and higher costs for livestock production leading to reduced levels of activity), which would in turn lead to greenhouse gas emission leakage through the higher levels of production in other economies necessary to make up the shortfall.

4.5.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The impact on land use should be broadly positive, both in terms of quantity (improved productivity), and in terms of quality (improved soil nitrogen, reduced levels of acidification). However, as noted above, the risk of a productivity penalty, notwithstanding the objectives of the policy to ensure improved productivity, need to be considered carefully. In the event of such an impact, there would be increased demand for land in the EU and elsewhere, to meet a given level of food consumption.

4.5.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

The policy would have the strongest, most direct, and least uncertain impact on water quality (through reduced nitrate pollution), leading to greater availability of water sources for extraction, with reduced costs on water providers, leading in turn to greater efficiency of water extraction and reduced pressure on water sources in areas of water stress.

4.5.7 Biodiversity impacts

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The policy would have a number of positive impacts on biodiversity, particularly as a result of reduced nitrate pollution of water bodies, but also as a result of improved air quality (reduced incidence of acidification and reduced tropospheric ozone concentrations). Stevens et al (2004), for example, identified a broadly linear function of inorganic nitrogen deposition and plant species richness in UK grasslands. Exposure to high ozone concentrations reduces rates of photosynthesis in plant species.

4.5.8 References

Sutton et al (ed), The European Nitrogen Assessment, Cambridge University Press 2011; chapter 22: “Costs and benefits of nitrogen in the environment”, Brink and van Grinsven

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4.6 Payment for Ecosystem Services

4.6.1 Short description of the policy

The measure consists in encouraging the establishment of PES programmes aiming at reducing the environmental impact of agricultural activities and financed by private actors (e.g. water companies, tourist operators, insurance companies and others with an interest in reducing flood risk). While development of PES programmes is intended to be financed by private companies, public authorities can facilitate and encourage the process by offering 1) fiscal incentives and 2) support, including mediation, control activities, and also, when appropriate, guarantees to ensure long term planning (e.g. guaranteeing the payment even in case the company goes bankrupt or cannot afford to pay). Sectors with potential for making such payments include the water sector (to encourage farms to avoid practices which lead to diffuse water pollution); energy undertakings or other businesses with an interest in offsetting carbon emissions; and tourism businesses which rely on high levels of landscape value and biodiversity.

4.6.2 Assumptions made, and identified conditions necessary to generate an environmental impact

We have assumed in particular that PES mechanisms are not associated with a weakening of existing nature protection requirements, for example by providing project developers with a significantly increased flexibility to take forward developments which impact on protected sites and habitats in exchange for offsets elsewhere. This is because, as noted by Tucker et al (2013), there could be "offsetting might not always achieve NNL in practice, as a result of the difficulties associated with restoring or creating some habitats, avoiding time-lags, ensuring the additionality of offsetting measures and achieving equitable outcomes when biodiversity and ecosystems are changed or moved." However, this also reduces the potential beneficial impacts of an offsetting policy – if it were possible to guarantee that net loss of habitats and biodiversity would be avoided, a PES-based approach could in theory improve the resource efficiency of development overall, for example by facilitating the development of sustainable energy sources. The voluntary and non-regulatory nature of the mechanism put forward is both an important element in ensuring that its impact is positive, and a constraint on its potential scope.

4.6.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

In principle, an approach which enables a better recognition of the environmental impacts of the agricultural sector should help to ensure a more sustainable approach to the management of agricultural land, including, for example, reduced extraction of raw materials to provide inputs such as fertiliser. The mechanism could, for example, include water undertakings paying farmers to reduce diffuse pollution through excess fertilisation, leading to more accurate application of fertilisers, which in turn will reduce the total volumes required. In practice, the impacts are likely to be relatively minor, unless there are also drivers for reduced inputs from the primary markets for agricultural products. In the event of PES leading to a reduced level of production from the farms which receive them, without any changes to the level of demand for agricultural products, that unchanged level of demand is likely to be met by increased production on other agricultural holdings, and the associated inputs. Wherever possible, therefore, PES systems should be devised so as to enable beneficiary farms to develop approaches to deliver of ecosystem services which avoid negative impacts on productivity as far as possible.

4.6.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

As with the impact on extraction of raw materials, the impact on greenhouse gas emissions is expected to be positive, but limited. Payments with an impact on greenhouse gas emissions could either be directly-related to the GHG-intensity of farming (for example, supermarkets could, as part of their wider corporate social responsibility commitment, support on-farm efforts to reduce emissions of methane and nitrous per unit of production); or could be co-benefits associated with reduced nitrate pollution of watercourses (since measures to reduce emissions to water through reduced fertiliser application or better storage of slurry are likely also to reduce nitrous oxide emissions to air). However, there remain risks that, without careful design of payments, they would lead to reductions in production, which would in turn cause increased environmental impacts from the likely indirect land use impacts resulting from increases in production elsewhere.

4.6.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low or ↘ (less than 10% deviation from BAU)

The land use impact is likely to be partly positive, with incentives created for rewilding (in part directly for landscape and wildlife purposes, but also including afforestation and other planting measures aimed at slowing water flows through the landscape, and thereby reducing flooding risks). However, there could also be some negative impacts, in addition to the indirect land use impacts noted above, through for example the facilitation of land use in exchange for expenditure on offsetting the environmental impacts elsewhere.

4.6.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BAU)

The impact on freshwater use is likely to be positive; by ensuring that water undertakings are able to make more direct use of the available water supply, with less need for treatment, the demand for further sources of freshwater supply can be expected to be reduced. While it is unlikely that the measure will have any noticeable impact on water use itself, the environmental impacts of supply current levels of demand should be reduced.

4.6.7 Assessment of impact on biodiversity

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

The principle benefit of the measure is likely to be in terms of improved biodiversity in the farmed environment, both as a result of land taken out of production and given over to more biodiversity-favourable uses (in a mechanism equivalent to the ecological focus areas measure under the Common Agricultural Policy – see separate assessment above). Measures to reduce flood risk and improve water quality are also likely to have positive associations with improved biodiversity outcomes. As noted above, there are risks associated with the use of offsetting to compensate for biodiversity damage caused by new development, but the proposed measure appears to mitigate those risks effectively (see under “assumptions” above).

4.6.8 References

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4.7 Pesticides: Stronger pesticide reduction targets, and guidance to farmers on integrated pest management

4.7.1 Short description of the policy

The proposed measures are to strengthen pesticide reduction targets in national pesticide action plans under the Sustainable Use of Pesticides Directive, improve pesticide licensing regimes to encourage full implementation of integrated pest management, ensure Farm Advisory Services provide all farmers with advice on integrated pest management; and improve the incentives for uptake of integrated pest management, including through links to the policy option on a stronger environmental dimension to the CAP. Existing Member State National Action Plans under the Sustainable Use Directive could be strengthened; with more demanding requirements in terms of reduced use of pesticides, and improved pest management. In the absence of further measures on a voluntary basis by Member States, the policy would need to await the 2018 review of the Directive, to be accompanied by any necessary proposals.

The measures primarily require action by Member States. They are all supported by legislative obligations on Member States, but states have considerable flexibility in implementation. The Commission's role is therefore mainly to encourage and facilitate Member States to be more ambitious in their actions. Regulatory authorities, pesticides manufacturers farmers and land managers, and other users of pesticides eg local authorities will be responsible for implementation. Environmental organisations and apiculturists are important influencing groups. Consequences will depend on Member State choices on implementation, but could involve fines and/or taxes (Withana et al 2013), or the compulsory withdrawal of some products from the market.

1.1 Assumptions made, and identified conditions necessary to generate an environmental impact

It is likely that some pesticide active substances with significantly negative environmental impacts will be withdrawn from use during the next decade. The European Food Safety Authority has an ongoing programme to review pesticide active substances for their human and environmental safety according to the Plant Protection Products Regulation 1107/2009, which might result in the withdrawal of some active substances with proven or likely ecotoxicity, eg endocrine disrupting effects, by 2020. The Water Framework Directive could also trigger the de-approval of active substances that are an obstacle to achieving good water status, such as metaldehyde. Pesticide bans can trigger farmers to switch to alternative pest control strategies that use less pesticide and lower environmental impacts, but they can also result in increased use of equally damaging (because less efficient) pesticides as a short term replacement if farmers do not have any alternative solutions.

In the absence of complete bans, pesticide use reduction will rely on increased investment in farmer support, training, and research and operationalisation of effective non-pesticide control strategies. The policy option therefore assumes an increased investment by Member States in their farm advisory services and agricultural research and innovation support networks. This is a stated priority of rural development spending in the current programming period to

2020; however it is not yet possible to say whether farmers really will benefit from increased support and training for integrated pest management and reduced pesticide use.

The spread of new and invasive alien pests and parasites in Europe could present sudden and new pest damage risks, which could trigger increases in the use of environmentally harmful pesticides, and may undermine advances in pesticide use reduction on the affected crops. *Xylella fastidiosa* is a current example (European Commission 2015).

4.7.2 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Pesticide production requires only small amounts of raw materials other than oil or gas, so the policy option is unlikely to have a noticeable effect on raw material extraction. The use of copper-based pesticides may decline as a result of more efficient and appropriate pest control, particularly if viable alternatives are promoted for organically managed vineyards; however, their use is unlikely to decline significantly unless legislative restrictions are put in place to reduce copper pollution of soils. This could happen as many vineyards already exceed the legislative limits for copper concentration in soils, with a risk of groundwater pollution (Komárek et al 2010).

4.7.3 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗ or ↘ = Low (ie less than 10% deviation from BaU)

Pesticide production relies heavily on fossil fuels, but the energy use is low compared to fertiliser production as the chemicals are produced in relatively low volumes (Woods 2010). It is unlikely that reductions in greenhouse gas emissions from pesticide production and application as a result of reductions in pesticide use will be significant. It is possible that farmers' changes in crop cultural practices as a result of the adoption of low pesticide crop systems may increase greenhouse gas emissions from agriculture because of increased use of soil tillage and mechanical weed control, but may also reduce emissions in some crop systems because of better soil management practices and increased soil organic matter. It is not possible to estimate the volume of the overall impact as it will depend on which cropping systems are altered and how individual farmers adapt.

4.7.4 Assessment of impact on land use

Estimated magnitude of change: ↗ or ↘ = Low (ie less than 10% deviation from BaU)

The pesticide use reduction measures are unlikely to trigger a decrease in the area of land occupied by agriculture in the EU unless the policy restricts pesticide availability whilst failing supporting the implementation of alternatives to such an extent that particularly vulnerable cropping systems are driven out of business, thus strengthening the already existing economic drivers of the conversion of agricultural land to other uses. A more likely policy failure scenario is the expansion of arable land area at the expense of extensive grassland as a compensation for reduced arable yields due to less effective pest control systems.

4.7.5 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Pesticide application requires relatively small amounts of freshwater. There may be an indirect effect if there is a reduction in the area of some irrigated intensive crops due to the lack of effective alternative pest management strategies for these intensive crop monocultures. This could result in a reduction in groundwater or river water abstraction for irrigation in certain water-stressed areas, but the effect is likely to be temporary as viable alternative pest control systems are developed for these high economic value cropping systems. It is not possible to quantify the volume of the impact.

4.7.6 Assessment of impact on land use environmental impact targets

Biodiversity: Estimated magnitude of change: ↗↗↗↗ = High (above 100% deviation from BaU)

There is evidence that pesticides are having consistent negative impacts on wild plant diversity, carabids and ground-nesting farmland birds across Europe (Geiger et al 2010). Current pesticide use is reducing the biodiversity of stream invertebrates both at the local and at the regional level (Beketov et al 2013). Furthermore, current monitoring systems are measuring pesticide concentrations with eco-toxicological impacts throughout the year, indicating that pesticides are exerting an evolutionary force in agriculturally impacted aquatic ecosystems (Bundschuh et al 2014), and the concentrations and effects of the diverse mix of chemicals entering freshwater are likely to be underestimated (Malaj et al 2014). There is evidence that the toxicity of pesticides to amphibians is currently underestimated and that pesticides are having a large-scale negative effect on amphibian populations in Europe (Brühl et al 2013). It is therefore to be expected that reductions in the quantity and ecotoxicology of pesticides entering freshwater will have significantly positive impacts on biodiversity in Europe. However, the effects are occurring at pesticide concentrations that current legislation considers environmentally protective (Beketov et al 2013). Therefore the pesticide measures in this policy mix option will have to reduce pesticide use beyond the mere compliance with current environmental standards in order to have this beneficial impact.

Soil functionality: ↗↗↗ = Medium high (ie between 50-100% deviation from BaU)

It is likely that the negative impacts of pesticide residues in soil on soil functionality are currently underestimated (Chagnon et al 2015). A reduction in the use of persistent eco-toxic pesticides can therefore be expected to have a beneficial impact on soil functionality. However, soil organic matter is a dominant driver of soil functionality, and if reduced pesticide use cropping systems result in reduced soil organic matter because of the necessity for increased tillage, this could cancel out the beneficial effect. However, organic farming systems have significantly higher soil functionality than conventional systems, so an increase in the area under organic arable farming would have a noticeable beneficial impact on soil functionality.

Water quality: Estimated magnitude of change: ↗↗↗↗ = High (above 100% deviation from BaU)

Currently, pesticide concentrations in freshwater in Europe are in exceedance of the drinking water standard in many areas (Eurostat 2013). The reduction or ban from use of certain pesticides will have a significantly beneficial effect on water quality. Candidates for restrictions or bans on the basis of their impact on water quality are metaldehyde and certain herbicides. However, environmental concentrations of certain pesticides will take a long time to fall even after an EU-wide ban, as shown by the example of atrazine (Nödler et al 2013).

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4.8 Value added tax (VAT) on meat products

4.8.1 Short description of the policy

This policy instrument aims to change dietary habits by reducing meat consumption through the application of VAT on meat products, thereby raising the price of meat products in most MS (a reduced VAT rate is currently applied to meat products in all but six MS - BG, DK, EE, HU, RO and SK). National governments would implement and enforce the tax measure, and the meat industry would need to comply by applying VAT to meat products. Exemptions could be envisaged for certain types of meat products that promote environmental protection and health e.g. organic meat products, meat produced according to strict environmental criteria, meat donated to charities and food donation programmes, etc). VAT on meat products would follow the already established rules on VAT, which applies to almost all goods and services bought and sold in the EU. As a consumption tax, increased prices (through VAT) are ultimately paid for by the final consumer. Tax revenues from VAT on meat products could perhaps go to MS funds to finance environmental protection initiatives and associated education and public campaigns.

4.8.2 Assumptions made, and identified conditions necessary to generate an environmental impact

Studies have suggested that around one third of total environmental impacts from consumers are related to food and beverage consumption (EEA, 2005 and Tukker et al, 2005). The agricultural production stage of meat and dairy products involves intensive use of land for grazing and feed production, with associated impacts on water use and ghg emissions (see sections below on ghg emissions, land use and water use), therefore any measures that can reduce consumption (and/or wastage) of meat and dairy products could be expected to generate an associated reduction in those impacts.

It is assumed that the measures would influence consumer purchasing decision on meat products due to the increased purchase/sale price of meat. Demand for meat and dairy products is price inelastic. According to one study, a 12% price increase would be expected to reduce demand for meat in the EU between 2% and 7%, and for dairy products between 2% and 5% (IVM, 2008).

4.8.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ or ↘ = Low (ie less than 10% deviation from BaU)

Since the extraction of raw materials in terms of the DYNAMIX project relates mainly to the use of virgin metals, the impact of this policy instrument in this area is likely to be negligible.

4.8.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗↗ = Medium high (ie between 50-100% deviation from BaU) to ↗↗ = Medium low (ie between 10-50% deviation from BaU)

One recent FAO estimate suggests that livestock (including poultry) accounts for about 14.5% of global ghg emissions (Gerber et al., 2013). A previous FAO study had estimated the figure at 18% of total global ghg emissions measured in t CO2e (more than the transport sector); including 9% of anthropogenic CO2 emissions and 37% of anthropogenic methane emissions (FAO, 2006).

A study cited in IVM et al. (2008) suggests that ghg emissions per tonne of food produced range from 1.1 tonnes CO2 equivalent for milk to 4.6 tonnes for poultry, 6.4 tonnes for pigmeat, 15.8 tonnes for beef and 17.5 tonnes for lamb (based on UK data), whilst emissions for vegetable based alternatives are much lower (though direct comparisons are difficult due to the variety of different crops involved and uncertainties about the changes in diet that would be brought about due to changes in meat consumption).

A 12% increase in the price of meat products in the EU (i.e. the application of the standard rate of VAT), which may reduce meat demand by between 2 and 7% and dairy product demand by between 2 and 5%, could bring about a gross reduction in ghg emissions of between 9.2 and 27.5 million tonnes CO_2 equivalent for meat and between 3.4 and 6.9 million tonnes CO_2 equivalent for dairy products, or a total in the order of 12-21 Mt CO2 equivalents per year (IVM et al, 2008). Another study suggests that if beef, dairy, pork, poultry and egg consumption in Europe were halved, agricultural ghg emissions could be reduced by 19% if the land was used to grow cereal crops, and up to 42% if the land was used for bioenergy crops (both figures based on a baseline figure of 464 million tonnes in the reference year of 2004). (European Commission, 2014)

4.8.5 Assessment of impact on land use

Estimated magnitude of change: ↗↗ = Medium low (ie between 10-50% deviation from BaU) to ↗ = Low (ie less than 10% deviation from BaU)

Reduced meat consumption would mean that the amount of land used to produce meat (including land used to grow animal feed) could be reduced. In total, livestock production accounts for 70% of all agricultural land globally, and 30% of the entire land surface of the planet (FAO, 2006). In Europe, each cow needs to consume between 75 and 300kg of dry matter (grass or grain) to produce 1kg of protein (Walsh, 2013).

One study suggests that if beef, dairy, pork, poultry and egg consumption in Europe were halved, the demand for animal feed would drop from around 520 million tonnes (for the baseline year of 2004) to 285 million tonnes, which would contribute to 23.7 million hectares of land previously used for pasture and growing animal feed becoming available for other uses. The amount of land needed to grow food for each EU citizen would fall from 0.23 to 0.17 hectares. (European Commission, 2014)

Another study (Woltjer, 2011) suggests that the total land use effect of a reduction in the consumption of livestock products in the EU is likely to be relatively small compared with the reduction in consumption, since the CAP incentivises keeping land in agricultural production.

The table below summarises the estimated percentage changes in land use for producing various agricultural goods under the following scenarios:

- Ref_CattleRed40: 40% reduction in consumption of red (cattle) meat, replaced with consumption of white (pork and chicken) meat;
- Ref_WHODiet: change in diet to match WHO recommended amounts, including for meat and dairy intake;
- Ref_GlobalFoodWasteRed15: a 15% reduction in global food waste;
- Ref_EUFoodWasteRed15: a 15% reduction in EU food waste;
- Ref_EUAnimalsRed10: a 10% reduction in consumption of cattle, other animal products and milk;
- Ref_EUAnimalsRed20: a 20% reduction in consumption of cattle, other animal products and milk; and
- Ref_EUAnimalsRed50: a 50% reduction in consumption of cattle, other animal products and milk.

Table 5.7c Land use change in the EU27 for the consumption reduction scenarios

	Primary Agriculture	Arable	Livestock	Cattle meat	Pork/ poultry	Milk and dairy
Ref_CattleRed40	-0.2	2.9	-5.1	-17.1	4.4	2.4
Ref_WHODiet	-0.8	-0.1	-1.9	-0.7	-4.3	-2.7
Ref_GlobalFoodWasteRed15	-1.5	-1.8	-1.0	-1.1	-4.4	-0.9
Ref_EUFoodWasteRed15	-0.6	-0.1	-1.4	-0.9	-2.9	-1.8
Ref_EUAnimalsRed10	-1.1	1.7	-5.3	-3.4	-10.5	-6.4
Ref_EUAnimalsRed20	-1.6	2.5	-7.9	-5.5	-16.6	-9.4
Ref_EUAnimalsRed50	-2.4	4.5	-12.8	-9.3	-34.4	-14.9

4.8.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU) to ↗↗ = Medium low (ie between 10-50% deviation from BaU)

In total, livestock production accounts for over 8% of global human water use, with most of that water being used to grow feed crops (FAO, 2006). Animal products, including dairy, make up 46% of the EU's water footprint (European Commission, 2013).

The water footprints for producing 1kg of various types of meat have been estimated as follows: 15,500 litres for beef, 4,800 litres for pork and 3,900 litres for chicken (Hoekstra, 2014). These estimates include water used to grow animal feed. Any reduction in meat consumption would therefore result in that amount of water not being required for meat production and therefore being 'saved'.

A study in 2013 (European Commission, 2013) compared the water footprints of four diets in the EU: the average diet for a European citizen; a healthy diet as recommended by nutritionists; a vegetarian diet (with meat substituted with pulses, nuts and oil crops, and including dairy products); and a combined diet midway between the healthy and vegetarian. In the vegetarian diet, meat was substituted with pulses, nuts and oil crops and also contained milk, yoghurt and cheese. The current average EU diet requires more water than is available within the EU-28; the water footprint of the vegetarian diet was 38% smaller; that of the combined diet 30% smaller; and that of the healthy diet 23% smaller.

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5 Metals policy mix

5.1 Green fiscal reform: internalisation of external environmental costs

5.1.1 Short description of the policy

The fiche is effectively an increase in environmental taxes and fees (IET&F) on emissions and all natural resources, to eventually incorporate 100% of environmental externalities. This includes materials, energy and water, and emissions from all economic sectors in Europe. Revenues are used to fund other instruments, such as sharing schemes, research and development, information and reducing labour taxes.

5.1.2 Change in the extraction of raw materials – reduce virgin material use by 80%

Estimated magnitude of change: ↗↗ Medium low (10-50% deviation from BaU)

The combination of both tax on first industrial use and emissions could send a strong message to industry to induce improvements. Hence there is good potential to decrease the extraction of raw materials within the EU. However, because the coverage of the policy is very comprehensive and such a tax has not been introduced before, the effects are difficult to assess.

A concern is that it could increase costs for European manufacturing, but the tax will be increased in steps to enable stakeholders to react in time, and reductions in labour costs. This is similar way to how the landfill tax was introduced in the UK (Withana, 2014; pg 68), which was also designed to be revenue neutral (reducing employers labour costs through reduced national insurance contributions).

The UK introduced an aggregates tax in 2002, which in conjunction with the landfill tax appears to have been effective at achieving absolute decoupling (Mazza et al, 2013). The use of primary aggregates in the UK has fallen by almost 50% from a peak of 262 MT (BDS 2009) in 1990 to 145.9 MT in 2011 (BGS, 2011).

In Latvia, a comprehensive natural resources tax was introduced in 1991 that covers the extraction of bulk materials, waste disposal and emissions to air (Ferdrigo et al, 2013). Although data is limited, the tax seems relatively ineffective. It shows only a weak relative decoupling at best of dolomite and sand and gravel mix from economic growth (Withana et al, 2014). However, this is probably due to a weak price signal with the rate being unchanged from 1996 until 2009.

A further concern is how it will affect “green growth” and the need for increased use of certain materials. For example, more copper will be needed in the coming electrification of vehicles. From a more complex life cycle perspective some materials such as rare earth metals, may improve the life cycle performance but have a perceived greater impact in the extraction phase, thereby increasing the initial costs. How it will affect shifts in the use of substitute materials is also difficult to predict. Since oil and gas are the basis of most polymers but are very efficient it could increase their use, but in some cases the recyclability may be less (e.g.

epoxy resins are not recyclable). Hence other policy support may be required to ensure life cycle thinking. There is also an increased need for education and marketing to support this.

Without policies that even the playing field this policy fiche could shift burden elsewhere by:

- supporting extraction and mining outside of the EU, where environmental restrictions may be less.
- reducing manufacturing in Europe and the export industry. This could lead to increased transport and production emissions on the whole. Even recycling activities could shift outside the EU due to the emissions component.

The IET&F has a good potential to reduce the extraction bulk materials but may have minimal effects on the extraction of metals primarily because there is only a very small mining industry in Europe (particularly Sweden, where iron is mined). It may not work so effectively for metals where transport is less of a defining factor (than for bulk materials), if only applied in Europe.

5.1.3 Change in the GHG emissions

Estimated magnitude of change: ↗↗Medium low (10-50% deviation from BaU)

Since both raw materials and emissions will be taxed then the IET&F can potentially encourage a large reduction in GHG emissions. The tax is likely to encourage a shift to renewable energy and help balance the playing field against fossil fuels.

In the Czech Republic the tax on air pollutants significantly reduced GHG emissions between 1990 and 2007 (CENIA, 2008), reducing total GHG emissions by 23% in the first 8 years[12]. Without a tax on CO2 introduced in Sweden in 1991, it is reported that CO2 emissions would be 20% higher (Cottrell, 2010). However, the system is characterised by a complex system of exemptions for several industry sectors in order to protect national competitiveness (Withana, 2014).

Through reducing the extraction of materials in the EU the IET&F could greatly reduce GHG emissions. The aggregates tax in the UK reduced life cycle global warming potential (not including the construction or use phase) from 3.38 MT CO2-eq to 1.80 MT CO2-eq per year, a reduction of 46.7% (Fedrigo-Fazio et al., 2014).

This is likely to be felt most for bulk materials rather than for metals which are likely to further shift outside of the EU, potentially creating a marginal increase in overall transport. An increase in emissions tax could see an increase in carbon capture and storage (CC&S) methods being used to capture emissions. This could see an increase on private sector R&D in CC&S technology in addition to any support from the recycled tax.

5.1.4 Change in the global land requirement required for EU consumption and production.

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BaU)

[12] http://www.un.org/esa/agenda21/natlinfo/countr/czech/natur.htm

The effects on land use are difficult to assess, but in general a perceived increase in reuse, remanufacturing and recycling would result in a lowering of global land requirement.

There could be a shift of agricultural production outside the EU due to increased costs of production, which would not result in an overall reduction of global land requirement.

5.1.5 Change in the water use

Estimated magnitude of change: ↗↗Medium low (10-50% deviation from BaU)

Since the use of water has been largely neglected by EU policy the IET&F should encourage more efficient use of water.

Denmark has the highest prices in OECD countries for water supply and treatment. The tax on water supply has a direct incentive for water companies to reduce leakages, as they are required to cover part of the tax if the water delivered to customers is less than 90 percent of the abstracted water (i.e. if water leakage is greater than 10 percent) (Withana et al, 2014). The price of water increased by 54 percent from 1994 to 2004 leading to a decrease in urban water demand by 24 percent, from 155 to 125 litres per person per day (Withana et al, 2014). Subsequently, water leakage had been reduced to only 10 percent compared to 30-40 percent for many EU cities (Eunomia and Aarhus University, 2014).

5.1.6 Other environmental impacts

Estimated magnitude of change: ↗↗ Medium low (10-50% deviation from BaU)

Reductions in the use of fossil fuel will also reduce other associated pollution in addition to GHG emissions, such as SO_2, nitrogen, particulates, toxic substances, petrochemical pollution through spills, leaks and urban runoff.

Reductions in the extraction of raw materials within Member States will see localised effects such as reductions in dust, transport and noise. There could be a shift to more organic farming due to increased costs of pesticides and fertilisers and their emissions. However, there could be an influx of cheap food imports, which could increase the overall life cycle emissions of food for Europe.

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5.2 Green fiscal reform: materials tax

5.2.1 Short description of the policy

The value-based tax is applied on all materials used in the EU and includes recycled materials. It is similar to VAT but only added on the first industrial use of the material and not throughout the value chain. The primary target is the manufacturing industry of the EU-27, with the aim of increasing the material efficiency of processes and systems.

It also applies to imports to level the playing field, but is applied based on the percentage of value that the material represents. In 2020, the threshold is set at 50% (meaning it is applied on imported products whose material value is 50% or more), decreasing to20% in 2030 and 10% in 2050. Exported materials and products are exempt where the material value is above the threshold. Revenues will be used for reducing taxes on labour.

5.2.2 Change in the extraction of raw materials

Estimated magnitude of change: ↗Low (Less than 10% deviation from BAU)

It is unclear how much effect this tax would have on extraction, since demand for natural resources and virgin materials tends to be own-price inelastic (Söderholm, 2011). For example, iron ore has fluctuated from around $13/tonne in 1997, to over $180/tonne in recent years without significant effects on material efficiency. The tax would not actively encourage recycling because the tax is also paid on recycled materials. Söderholm (2011) argues that additional policies are required as a tax on virgin materials does not create an incentive for some generators of recycled materials to enhance waste sorting activities.

One of the best examples of a similar instrument is the UK aggregates tax, where use of primary aggregates in the UK fell by almost 50% from a peak of 262 MT (BDS 2009) in 1990 to 145.9 MT in 2011 (BGS, 2011). However, this instrument was combined with a landfill tax to encourage the use of recycled and secondary materials. It saw an increase in the use of recycled and secondary materials (by-products of other processes) by four fold (MPA 2012; BDS. 2009).

A comprehensive natural resources tax introduced in Latvia in 1991 covers the extraction of bulk materials, waste disposal and emissions to air (Ferdrigo et al, 2013). Although data is limited, the tax seems relatively ineffective. It shows only a weak relative decoupling at best of dolomite and sand and gravel mix from economic growth (Withana et al, 2014). However, this is probably due to a weak price signal with the rate being unchanged from 1996 until 2009.

Hence if the tax is not used in combination with a waste tax or similar then it may struggle to encourage material efficiency, and simply raise the price of material goods. However, because recycled materials will also increase in price and labour is made cheaper (by utilising the tax revenues) it may encourage the more efficient circular resource flow approaches of reuse, refurbishment and remanufacturing which are more labour intensive.

The tax will eventually be 200% of the value which is fairly high, and could send a large signal to increase material efficiency. For instance the UK’s aggregates tax which saw a strong improvement in material efficiency was only 20% of the value (European Environment Agency, 2008). The tax was in fact based on an economic valuation study that estimated the total external costs of aggregate extraction in the UK (London Economics 1999).

Reuse and remanufacturing can make huge savings on resources. Whilst reuse can potentially save almost 100% of materials, energy and water, remanufacturing can save 85% and 80% for materials and energy respectively used in manufacturing (Steinhilper, 2006). The potential for reuse, refurbishment in Europe and remanufacturing is not well known, although research is strengthening.

Typically remanufacturing represents only 1-2% of a county's turnover for the manufacturing industry (Lavery and Pennell, 2013). Comparing the remanufacturing rate of some top performers such as Xerox, Lavery and Pennell (2013) proposed a figure of 50% for a realistic potential (with today's technology) on remanufacturing rate. One of the most critical barriers is infrastructure that will require additional policy support (All-Party Parliamentary Committee, UK, 2014).

There are a couple of disparities with the tax:

- The tax assumes all resources are the same so will not actively encourage better environmental management or selection of environmental superior resource supplies (e.g. from a more efficient and less environmentally damaging mine).
- The tax is applied according to a threshold based on the percentage of material value, but a product with a higher portion of value compared to material value is not necessarily more environmentally sound. For example, the high market value of an Italian sports car means the material value is proportionally lower than compared to an average family car (and hence has a better chance of not being covered by tax due to the threshold), but is generally much less efficient and typically underutilised.
- Some products may require more material in order to have a longer life span and enable opportunities for reuse and remanufacturing. However, the material tax may not encourage this and in fact could inhibit this approach. This requires a life cycle approach in marketing also so that customers understand an increase in the initial cost.

5.2.3 Change in the GHG emissions

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

Through reducing the extraction of materials in the EU the tax could (potentially) reduce GHG emissions. The aggregates tax in the UK reduced life cycle global warming potential (not including the construction or use phase) from 3.38 MT CO2-eq to 1.80 MT CO2-eq per year, a reduction of 46.7% (Fedrigo-Fazio et al, 2014). However, the effects of the tax are likely to be felt most for bulk materials rather than for metals, whose extraction and production may begin to further shift outside of the EU. This could potentially create a marginal increase in overall transport.

About 3000 MT of non-metallic minerals (mostly construction minerals such as sand and gravel) were extracted in 2012 that accounted for about 40% of the domestic material input for Europe (Eurostat, 2014). If the figures from Fedrigo-Fazio et al. (2014) are extrapolated (295 MT of aggregates resulted in 3.38 $MTCO_2$), this is only responsible for about 34 $MTCO_2$, or 0.008% of the 4.2 EU emissions of the 4.2 $GTCO_2$ for 2012. However, if one considers both mining and quarrying, it is reported as representing 2% of GHG emissions for Europe (Eurostat, 2014b). Hence if the materials tax only effects bulk materials, it is likely to make only a minor contribution to the reduction of GHG in Europe.

5.2.4 Change in the global land requirement required for EU consumption and production

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

The effects on land use are difficult to assess, but in general effects from this tax specifically are expected to be a small reduction at best, due to a possible increase in reuse, remanufacturing and recycling.

The land use for domestic extraction of aggregates requires relatively small areas of land compared for example to biomass (Bleishwitz and Bahn-Walkowiak, 2006). For example, Germany the largest extractor used 14,000 km2 in 2001 which his less than 0.005% of Germany's total land area. The total for mining and quarrying amounts to 0.5% of the total area of Germany. However, the effects of quarrying and mining generally mean that the land cannot be utilised for other purposes for quite some time.

There could be a shift of agricultural production outside the EU due to increased costs of production, which would not result in an overall reduction of global land requirement.

5.2.5 Change in the water use

Estimated magnitude of change: ↗ Low (Less than 10% deviation from BAU)

Water would not be directly affected and any reduction in water use would be associated with increases in material efficiency. The effects on water use are therefore only expected to be marginal.

5.2.6 Other environmental impacts

Reductions in the extraction of raw materials within Member States will see localised effects such as reductions in dust, transport and noise

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5.3 Product standards

5.3.1 Short description of the policy

This measure entails the development of standards to regulate the design of specific products and components. The standards should initially cover a few select products or components only, but be gradually extended to more and more products. The measure might include, for example, standards to improve modularity to increase repairability and reuse of components (taking into account impacts on energy efficiency), standards to reduce the unnecessary use of material, and standards to substitute metals for other materials when appropriate (e.g. shifting from copper water-piping to polymer piping). The aim is to reduce the use of virgin metals through product redesign, increased longevity, and increased recycling. The EU should attempt to initiate the development of international standards within the framework of the International Organization for Standardization (ISO), or if that fails, to develop European standards within the framework of the European Committee for Standardization (CEN). It is anticipated that the measure would be introduced by 2030, to allow time to gain acceptance among policy-makers and industry for the idea of product standards with an explicit environmental purpose, and to allow adequate time for the development of the first standards.

5.3.2 Assumptions made, and identified conditions necessary to generate an environmental impact

The idea of product standards with an explicit environmental purpose might be more easily accepted if such standards are part of a dynamic policy package that begins with the establishment of EU strategies for material efficiency, resource efficiency and/or sustainability, and perhaps that includes green fiscal reform (e.g. a materials tax, increased R&D on material efficiency and the removal of environmentally harmful subsidies).

Since this measure is in the metals policy mix, the assumption has been made that the main focus of this measure will be to reduce environmental impacts from the use of metals (e.g. by reducing metal use, increasing substitution of alternative materials etc); the assessment therefore does not focus on the energy-related aspects of ecodesign, or take particular account of assessments of the impacts of ecodesign on the environmental performance of energy-using products.

It is worth noting that most metals can be recycled any number of times without loss of quality. It should also be noted that design that encourages recycling will only have a significant impact on resource efficiency if the products are actually collected and recycled at the end of their lifetime; there will be no material savings if this does not happen.

In addition, it is possible that if the EU were to introduce standards for recycled content in metals products, this would shift the majority of use of virgin metals outside the EU. An international standard is therefore likely to be more effective.

5.3.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗↗ = Medium low (ie between 10-50% deviation from BaU)

The environmental benefits will differ widely for different standards, depending on the volume of products affected and the requirements of the specific standard concerned. Product standards are likely to be effective when they specify the type or quantity of material in the product, in the sense that the environmental benefit from reduced material use can be estimated in advance. The environmental benefits are more difficult to predict for standards that aim at modularity to increase repairability and reuse of components. Standards should aim to encourage product design that is more environmentally friendly and less resource intensive.

Eco-design strategies relevant to raw material use include: reducing the material intensity (i.e. using less material for the same functionality, or product lightweighting), eliminating or reducing the dispersion of harmful substances, increasing the amount of recycled and recyclable material, optimise the product's durability (e.g. by making it durable, repairable and upgradeable), and making the product's environmental features visible to the user/consumer (McAloone et al, 2008). Although the Ecodesign Directive currently mainly focuses on promoting energy efficiency, a recent study suggests that it could, at least in theory, be used to regulate almost any resource efficiency parameter of energy-related products provided that the parameter can be measured and there is significant potential for impact/improvement (VHK, 2014). Potential resource efficiency related parameters that could be employed include: reusability/recyclability/recoverability (RRR) rates, recycled content, use of priority resources (RRR benefit rates), use of hazardous substances (which can be a barrier to end-of-life treatment), and durability (VHK, 2014); these could presumably therefore also be used for the development of product standards. As noted above, international standards would likely be more effective than solely EU-wide standards, in particular when it comes to encouraging recycled content; although the mechanisms for securing such international standards are less obvious. Test cases to date on RRR benefit rates have suggested that for energy-related products (e.g. washing machines and televisions) the environmental gain is likely to be less than 10% for most environmental impact categories; evidence is likely to be easiest to gather for measures related to recycled content, durability of components, use of hazardous substances, and use of materials of certified origin (VHK, 2014). New Ecodesign-type requirements for non-energy related products could in theory bring substantial environmental improvements, e.g. for food and beverages, clothing, furniture and cleaning chemicals, which have significant environmental impacts; however significant research would be needed into technical feasibility, data availability and necessary measurement/testing methods (CSES, 2012).

Design can play a critical role in material use and material savings; the shape and dimensions of the product, the choice of materials, and the possibility of reusing/recycling the product at the end of its useful life are all defined during the conception of the product, and all directly influence the type and amounts of materials used and their corresponding environmental impacts during the product's life cycle (BIO IS et al, 2011). It has been estimated that around 80% of a product's environmental profile is set during the concept creation stage (McAloone et al, 2008), and that around 80% of a product's environmental impacts can be eliminated through better design (House of Lords, 2008).

Assuming a market share for ecolabelled items of 5%, the EU ecolabel was estimated to have led to a saving of 25.11 ktonnes of titanium dioxide (AEAT, 2004). Actual contributions of the Ecodesign Directive to material use and efficiency have not been significant to date since the main aim is to promote energy efficiency and none of the implementing measures to date have specifically addressed material issues; the main material savings have been made as an indirect result of change in energy efficiency technology for domestic lighting (compact fluorescent lighting (CFL) bulbs, which have replaced incandescent light bulbs, use more material per light bulb, but the product's lifetime is much longer) (BIO IS et al, 2011).

One study estimates the amount of metals recycled if EU recycling targets were to be fully met. The total is 21,420 ktonnes (3,856 kt from municipal solid waste (MSW), 7,998 kt from construction and demolition (C&D) waste, 4,352 kt from end-of-life vehicles (ELV), 3,275 kt from packaging waste, 787 kt from waste batteries and 1,151 kt from WEEE) (BIO IS et al, 2011). If product standards were to include requirements for recycled material content, it may be the case that they would help to drive greater material savings. Indeed, the same study estimated the future potential for the amount of metal recycled to be a total of 26,454 kt (5,013 kt from MSW, 9,800 kt from C&D waste, 4,597 kt from ELV, 3,382 kt from packaging waste, 843 kt from waste batteries and 2,819 kt from WEEE) (BIO IS et al, 2011).

The weight of packaging has tended to decrease significantly since the 1950s, and also slightly since the entry into force of the Packaging and Packaging Waste Directive in 1994 (although it is not certain whether the decrease in packaging weight has been a direct result of the PPWD). For example, the weight of a metal can to contain 400g of soup has reduced from 90g in the 1950s to 57g in the 1990s and 49g in 2008, whilst a metal drinks can (330ml) has reduced from 60g in the 1960s to 21g in the 1990s and 14g in 2008 (BIO IS et al, 2011).

Product lightweighting (and to some extent also dematerialisation and miniaturisation) reduces the weight of products during design whilst considering the environmental impacts throughout its entire lifecycle, leading directly to reduced raw material use. One study gives the technological advance from a Walkman to a miniature mp3 player as an example of a product providing the same function that has undergone drastic lightweighting (BIO IS et al, 2011). Another suggests that changing car manufacturing production structures could improve material productivity by up to 29%, substituting material could save 20% and product design could reduce up to 84% of material usage in car production, but that further R&D would be required to achieve these potentials (Wuppertal Institute, 2007). The BIO IS (2011) study suggests that there is evidence that it would be economically feasible to redesign packaging to use 5% less materials (by weight), electronic and electrical equipment (EEE) to use 10% less materials (e.g. through miniaturisation), and vehicles could be redesigned to use up to 30% less materials (provided consumers are willing to accept smaller cars).

Every tonne of aluminium recycled saves 4 tonnes of bauxite, whilst every tonne of steel packaging recycled saves 1.5 tonnes of iron ore (Every Can Counts, 2015).

The EU Packaging and Packaging Waste Directive (PPWD) includes provisions on 'essential requirements' for packaging, relating to manufacturing and composition of packaging, the reusable nature of packaging, and the recoverable nature of packaging. In addition, CEN standards and ISO standards on packaging and the environment provide guidance on the resource-efficient design and ecodesign of packaging. However, although the PPWD has mandatory material-specific recycling and recovery targets and this has helped to contribute to natural resource savings (e.g. water and land), overall generated packaging waste is still increasing, reuse of household packaging is decreasing, and the strong focus on lightweighting of packaging does not always deliver environmentally optimal results. (BIO IS et al, 2014)

Regarding end-of-life vehicles, the ELV Directive and other regulations such as Directive 2005/64 have forced car manufacturers to integrate ecodesign into their processes (e.g. Directive 2005/64 requires new vehicles to demonstrate reusability and/or recyclability of at least 85%, and reusability and/or recoverability of at least 95% by weight, as measured against the international standard ISO 22620) (BIO IS et al, 2014). To comply with the ELV Directive, the car industry has also significantly limited its use of hazardous substances: lead emissions have been reduced by 99.6%, cadmium by 96% and hexavalent chromium almost completely (99.99%) (Öko-Institut, 2010).

Green public procurement (GPP) criteria currently exist at the EU level for 22 groups of products/services. Those of potential relevance to metals include: construction; electrical and electronic equipment used in the health care sector; furniture; gardening products and services; imaging equipment; office IT equipment; road construction and traffic signs; sanitary tapware; street lighting and traffic signals; and transport (European Commission, 2015a).

5.3.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ = Medium low (ie between 10-50% deviation from BaU)

With regards to cars, EU Regulation (EC) No 443/2009 sets emission performance standards for new passenger cars, requiring manufacturers to achieve an average of 130 grams of CO2 per kilometre (g/km) by 2015 across their whole new car fleet, and an average of 95g/km by 2021. The EU car fleet as a whole was estimated to have already met the 130g/km target in 2013, two years ahead of the deadline (the average specific emissions of the new European car fleet in 2013 was 126.7 g CO2/km, a reduction of 4.1% compared to 2012) (EEA, 2014). If the 2021 target is met, this will represent a reduction of 40% compared with the 2007 fleet average of 158.7g/km (European Commission, 2015b). This is therefore an example of a successful ‘product standard’ (i.e. emission reduction standard) that has already been applied in the EU on a large scale.

For other products, the main impact on GHG emissions would come from factors such as the magnitude of the reduction of the amount of energy used to extract raw materials for products that are designed to be more environmentally friendly and less resource intensive, and the reduction in fuel use by, or used for the transportation of, products. This is likely to be correlated to the achieved reduction in raw material use. According to the British Metals Recycling Association, using recycled steel instead of virgin ores cuts down CO_2 emissions ‘significantly’, and reduces air pollution by 80% (EMR, 2015). European Metal Recycling, a global leader in metal recycling that operates at 150 locations worldwide, claims to save 15 million tonnes of CO_2 every year through recycling, compared to using virgin ores (EMR, 2015). Another source suggests that every tonne of aluminium recycled saves 9 tonnes of CO_2 emissions, 97% of GHG emissions, and 95% of the energy need to make aluminium from raw materials; every tonne of steel packaging recycled reduces CO_2 emissions by 80%, saves 75% of the energy needed to make steel from virgin material, and reduces air emissions by 86% (Every Can Counts, 2015).

5.3.5 Assessment of impact on land use

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU)

There may be an associated reduction in land use for mining virgin metal ores if product standards are designed to encourage the use of recycled metals.

5.3.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ = Low (ie less than 10% deviation from BaU) OR ↗↗ = Medium low (ie between 10-50% deviation from BaU)

If product standards are designed to encourage the use of recycled metals, there may be an associated reduction in the amount of water used (since recycling metal requires less water than extraction and processing of virgin ore). According to the British Metals Recycling Association, using recycled steel instead of virgin ores reduces water use by 40%, and water pollution by 76% (EMR, 2015). Another source suggests that every tonne of steel packaging recycled saves 40% of the water required in production and reduces water pollution by 76% (Every Can Counts, 2015).

5.3.7 References

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5.4 Increased spending on research and development (R&D)

5.4.1 Short description of the policy

This instrument would involve continued and strengthened public funding of research and development (R&D) for recycling and material efficiency, primarily targeting research at universities and institutes and R&D in the manufacturing industry. The aim is to increase recycling and material efficiency, reduce the use of virgin material, and enhance the competitiveness of European industry. The R&D for recycling will include:

- Design for recycling;
- Efficient and consumer-adapted systems for collection, and identification of the role for the public sector in ensuring their provision;
- Technology for dismantling and separation of components and material; and
- Technology for recycling.

The R&D for material efficiency will include, for example:

- Improved processes and products;
- New business models; and
- Non-material alternatives for safe investments (to find ways to substitute metals, particularly gold, with other ways of delivering the service safe investments).

It is envisaged that the policy would be introduced by 2020, and the results of the R&D should directly affect the manufacturing, wholesale, retail and waste management sectors.

5.4.2 Assumptions made, and identified conditions necessary to generate an environmental impact

It is assumed in the policy fiche that public spending on R&D for recycling and material efficiency will be doubled by 2020 and kept constant (in terms of its share of total public R&D spending) after that date.

The effectiveness of R&D in terms of increasing recycling and material efficiency in the EU-28 is almost impossible to predict. The environmental impacts will depend on many factors including the aim of the R&D, the level of take-up of available R&D funding, the potential scale of application of any new technologies/processes etc. that are developed (e.g. whether it can be applied to the manufacture of numerous, widely used products or only to a specific niche product), the actual scale of application in the real world context (including whether companies are willing to share their R&D results), the types of projects funded and so on. It would also likely be beneficial to ensure that industry and SMEs are adequately involved in, and benefitting from, the outcomes of research, to enable industry to act as a bridge between research and commercialisation of research findings, and to encourage innovation (European Commission, 2010).

However if effective, and if translated from research results into real-world applications, increased R&D spending should contribute to recycling and material efficiency, and reduce the energy-intensive production of virgin materials and associated environmental impacts. It is also worth noting that impact can also be achieved through R&D findings contributing to national and international policy, for example by strengthening the knowledge base and helping to develop methods and tools for environmental policy (European Commission, 2009).

Note that it is impossible within the scope of this assessment to look at the results of all EU-funded R&D to assess environmental impacts. However, some literature on the subject has been found and reviewed, and a small number of brief examples are included where environmental benefits (actual or theoretical) have been demonstrated.

5.4.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

R&D on recycling and material efficiency should, by its nature, have the goal of increasing recycling and material efficiency. It will be important to measure the impacts of relevant R&D to identify its potential to contribute to increased recycling and improved material efficiency. Funding for eco-innovation can contribute to reduced use of natural resources per unit of output, and it has been recognised that research in this area has a major role to play, for example by substituting existing technologies, facilitating novel products/processes that are more resource efficient, and promoting more holistic assessments of environmental impacts (European Commission, 2011).

A report on the impacts of the EU FP6 sub-priority 'Global Change and Ecosystems' suggests that in areas of applied research such as environmental technologies, a focused approach (rather than large-scale instruments) is the most effective approach to take (European Commission, 2009). The same report suggested that R&D projects can have several benefits (many of which could potentially contribute to reducing raw material extraction if they are targeted in that area), for example: strengthening the knowledge base for greater consideration of sustainability in building/construction; contributing to progress on the harmonisation of standards in the EU; moving forward the technological state-of-the-art (e.g. through modelling, tool development, prototypes or demonstrations); the creation of pilots, new products and services (e.g. web-based tools); creating forums for discussion within scientific communities and between scientific experts.

The EU-funded ZeroWIN (Towards zero waste in industrial networks) project aims to contribute to the objective of achieving zero waste and emissions through industrial symbiosis and integrated systems that are more resource efficient. The project addresses almost 3 million companies (mainly SMEs) in sectors including electronics and construction, that generate 400 million tonnes of industrial waste per year, and aims to help them reuse/recycle 70% of their waste. (European Commission, 2014)

One EU-funded R&D project (SORT IT) developed a full scale sorting line using innovative technologies (sensors and machinery) to improve sorting processes in industrial paper recycling/production. The environmental sustainability performance of newsprint paper and packaging paper life cycles were optimized by achieving improved quality of recovered paper, and lower consumption of the improved quality recovered paper, raw materials and energy. (European Commission, 2014)

5.4.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: ↗↗ Medium low (between 10-50% deviation from BaU)

An interim evaluation of the EU's 7th Framework Programme for Research (FP7) suggested that from the perspective of EU policy, research efforts going forward should increasingly focus on recognised 'Grand Challenges' faced by the EU and its citizens, including climate change, competitiveness and energy supply (European Commission, 2010). This suggests that increased R&D funding could generate greater impacts with regard to ghg emissions.

The EU-funded C2CA (Advanced technologies for the production of cement and clean aggregates from construction and demolition waste) project aims to develop innovative technologies to recycle end-of-life concrete to obtain material that can be used as a secondary raw material in cement kilns, thereby replacing virgin materials and reducing construction and demolition waste. R&D in this area could contribute to reducing CO2 emissions, since the production of cement used in concrete is estimated to contribute over 5% of global CO2 emissions. (European Commission, 2014)

The EU-funded SORT IT project (see above) demonstrated a reduction in energy consumption in paper mills, which was the major environmental benefit (and would have associated ghg emissions reductions). (Escabasse and Blasius, 2011)

5.4.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Very limited literature has been found to support an estimate of the magnitude of the impact of increased R&D funding on land use.

The SORT IT project (see above) aimed to improve the quality of recycled paper, which would in turn reduce the demand for virgin pulp, which could reduce the pressure on forests to provide material for paper production. (European Commission, 2014)

5.4.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (less than 10% deviation from BaU)

Very limited literature has been found to support an estimate of the magnitude of the impact of increased R&D funding on freshwater use.

The EU-funded ZeroWIN project (see above) is aiming to help almost 3 million companies in sectors including electronics and construction to become more resource efficient; one of the targets is to achieve a 75% reduction in fresh water consumption. (European Commission, 2014)

5.4.7 References

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5.5 Sharing systems

5.5.1 Short description of the policy

This policy would see the establishment (e.g. by local authorities) of sharing systems for cars, bicycles, tools, and equipment, to make leasing and sharing of products more convenient compared with ownership of the same products. Typically, users of the scheme would pay an annual membership fee to gain access to the shared products/services. Depending on local and national conditions, this could include:

1) Local authority-led schemes for sharing of cars, bicycles, tools, and equipment;
2) Local authority support for the creation of private sharing systems, by funding part of the investment cost;
3) National authority support to private sharing systems, e.g. through deductions in income tax to consumers for rental costs, or through differentiation in VAT between goods and services.

The aim of the policy is to reduce the use of metals and other materials through a reduction in the number of products used (because products are being shared by more users/consumers). The main target for the measure is households, but commercial leasing companies and non-profit organisations could also be targeted.

5.5.2 Assumptions made, and identified conditions necessary to generate an environmental impact

Although this policy focusses on the use phase of products' life-cycles, it is anticipated that it would also indirectly affect the production phase, since manufacturers/producers may see a reduction in demand for their products.

Bicycle sharing schemes have become increasingly popular in recent years as a means of encouraging cycling as an alternative means of urban transport; large rental systems now exist in many cities worldwide (e.g. Lyon, Paris, Toulouse (France), Stockholm (Sweden), Barcelona, Seville (Spain), Hangzhou, Guangzhou (China), Milan (Italy), Brussels (Belgium), Montreal (Canada), Mexico City (Mexico) and London (UK)) (Rojas-Rueda et al, 2011). A survey of car club trends in the UK found that there was an unprecedented use of car clubs in 2014-15, that more car clubs are being created, they are becoming better integrated with public transport, that car clubs are increasingly becoming a lifestyle choice, that there is significant and continuing government support for car clubs in Scotland and England, and that the types and ages of people using car clubs is becoming more diverse (CarPlus, 2015). If trends for shared hired transport such as this continue, this will certainly bring about a reduction in ghg emissions, and may also result in a reduction (although most likely a small reduction) in private vehicle ownership.

In much of the literature reviewed for this assessment, environmental concerns/benefits were cited as a reason/motivation for various types of sharing/collaborative consumption etc, but little solid evidence was found of the actual environmental benefits of sharing systems. In addition, much of the positive opinion or data found was presented by organisations with an interest in the sharing economy; there therefore appears to be a current lack of neutral, unbiased reports on concrete, quantified environmental benefits.

5.5.3 Assessment of impact on extraction of raw materials

Estimated magnitude of change: ↗ Low (ie less than 10% deviation from BaU)

Initially, it is estimated that sharing systems would likely have a limited impact on the total use of materials, since a relatively small share of materials is used for producing cars, bicycles, tools, and equipment. The materials affected, however, are mainly metals and polymers, which are associated with relatively high environmental impacts in the production phase. Provided that the shared items (cars, bicycles, tools etc) are well maintained/kept in a good state of repair, it might indeed lead to the purchase of fewer new, privately owned items, leading in turn to a reduction in the extraction of raw materials to make the items concerned.

An annual survey of car clubs in England, Wales, Scotland and London found that for each car club vehicle, between 3.5 (in Scotland) and 8.6 (in London) cars have been removed from the road (as a result of car club members selling a car), amounting to almost 23,690 cars in total (20,150 of which in London) removed from the roads. In addition, almost a third of car club members in Great Britain would have bought a private car if they had not joined a car club, representing approximately 55,060 fewer cars purchased (46,500 of which in London). This means that one car club car 'replaces' on average 12.8 private cars (9.2 in England and Wales, 9.3 in Scotland and 19.8 in London). The survey also found that car ownership amongst new members falls after joining (e.g. in England and Wales, 54% of new members owned at least one car before joining, falling to 35% afterwards; around 50% of longer-term (six months or more) members Britain-wide owned at least one car before joining, falling to between 20 and 29% afterwards); and in England and Wales 12% of longer-term members had sold or disposed of a car in the 12 months prior to completing the survey, with 40% of these citing their car club membership as a major factor in their decision). In addition, car club cars tend to have a higher occupancy rate than private cars (an average occupancy of 2 people compared with 1.6 people). (data compiled from Steer Davies Gleave, 2015a, 2015b and 2015c)

One article suggests that the picture on the environmental impacts of the sharing economy may not be entirely positive. For example, Airbnb (a website through which people can book a room or whole property owned privately by someone else, rather than a hotel) may mean that fewer hotels get built, saving concrete, steel, construction waste and carbon emissions; on the other hand, it may mean that people are able to travel more, or buy a steak dinner (with its related climate impacts) with the money they have saved by not staying in a hotel. (Gunther, 2014) On the other hand, a study commissioned by Airbnb (based on a survey of 8,000 Airbnb hosts and guests) found that accommodation booked through the site results in significant reductions in waste, that fewer than half of Airbnb hosts in Europe provide single-use toiletry products for their guests (therefore possibly reducing waste) (Airbnb, 2014).

5.5.4 Assessment of impact on greenhouse gas emissions

Estimated magnitude of change: in the range of ↗Low (ie less than 10% deviation from BaU)

The London ‘Barclays Cycle Hire’ scheme allows people to hire bikes from docking stations throughout the city centre, and from the scheme’s launch in July 2010 to the end of March 2015 there had been 38,746,226 hires (TfL, 2015). A report from the Mayor of London suggests that if 14% of journeys in central London were made by bicycle, it would reduce central London emissions of NOx by 30% (453 tonnes per year), and reduce emissions of particulate matter by 24% (33.8 tonnes per year); in addition, it may help to reduce the health issues related to air pollution from vehicles (according to the Massachusetts Institute of Technology, air pollution from vehicles prematurely kills 2,200 Londoners every year) (Greater London Authority, 2013). A cyclist making 160 trips of 3.9km[13] per year by bicycle instead of single-occupancy car would result in an estimated total saving of 112,000 grams of CO2 per person, or 112 metric tonnes of CO2 per 1,000 people (SQW, 2007). The ‘Bicing’ public bicycle sharing initiative in Barcelona, is estimated to have led to an annual reduction of over 9,000 tonnes of CO2 emissions (Rojas-Rueda et al, 2011); the system registers over 1 million uses per month (Bicing, 2015). In Paris, the Vélib’ bicycle sharing system registers between 50,000-70,000 hires per day, or around 22 million per year (Planetoscope, 2015).

One study of car sharing schemes in several EU countries identified that the specific CO2 emissions of car sharing cars was, on average around 19.7% lower than the specific CO2 emissions of the average new private car in the same country (ranging from 7.6% to 36% lower). The same study suggested that each car sharing vehicle replaced at least four to eight private cars. (Bundesverband CarSharing, date unknown)

On average across England, Wales, Scotland and London, car club members’ annual mileage is around 2,790 miles, lower than the national average of 5,333 miles; the average car club member travelled 1,787 fewer miles per year after joining, which represents a carbon saving of around 450kg CO2/annum (Steer Davies Gleave, 2015). Car club cars in Great Britain typically have emissions that are a third lower than the average private car, emitting around one tonne less of carbon per year for the same mileage; 90% of car club vehicles in England, Wales and Scotland (and 80% in London) are in the lowest three emission bands (A, B and C), whereas the majority of the UK vehicle fleet is in bands E to M (data compiled from Steer Davies Gleave, 2015a, 2015b and 2015c). A survey of car club trends in the UK suggested that the proportion of hybrid, plug-in hybrid and electric vehicles in car clubs is increasing (CarPlus, 2015), which would also lead to a reduction in ghg emissions. Since car club cars tend to be lower-emission than private vehicles, this should mean that whenever they are used (even if infrequently, by people who also own cars, or only for short journeys) they should still have a benefit in terms of lower ghg emissions.

In addition, the reduction in resource use associated with the reduction in vehicle numbers would itself provide a benefit in terms of a reduction of emissions associated with extraction of raw materials and vehicle manufacture.

There is, however, some potential for some countervailing rebound effects, including through increased vehicle use by those who find car club membership feasible, but vehicle ownership impractical; and it is noticeable that many of the reports we have identified are commissioned by or written by organisations with an interest in promoting car clubs, potentially leading to some optimism bias.

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[13] The average distance of a cycle trip according to the UK National Travel Survey 2005.

A study commissioned by Airbnb found that accommodation booked through the site results in significant reductions in energy use and ghg emissions; for example, in one year it is estimated that Airbnb guests in Europe avoided ghg emissions equivalent to 200,000 cars, nearly 79% of Airbnb hosts in Europe reported owning at least one energy efficient appliance, and Airbnb guests are 10-15% more likely to use public transportation, walk or cycle as their primary mode of transport than if they had stayed at a hotel (Airbnb, 2014).

5.5.5 Assessment of impact on land use

Estimated magnitude of change: ↗ Low (ie less than 10% deviation from BaU)

Car sharing schemes and to some extent bicycle hire schemes may in some cases result in a reduced need for areas dedicated to parking vehicles; there may therefore be a small amount of land freed up for other uses, such as green spaces or parks. This is unlikely to be a significant land use change (benefits would likely be felt at a local level), unless car sharing becomes a very significant share of the mobility mix. For example, in a new neighbourhood (Vauban) in the southern German city of Freiburg, car-owning households had to pay to purchase a parking space in a central community garage. Car-free households, meanwhile, could offset the legal requirement of one parking space per new-build flat by declaring their car-free status to the local authority and joining an association that owned a plot of land in the neighbourhood, where green space and play areas were established. This cost a fraction of the amount of an actual parking space. (Bundesverband CarSharing, date unknown)

5.5.6 Assessment of impact on freshwater use

Estimated magnitude of change: ↗ Low (ie less than 10% deviation from BaU)

Very limited literature has been found on the impact on freshwater use of sharing systems. However, it may be the case that if raw material consumption is reduced due to reduced ownership of items (e.g. cars) as a result of sharing systems, this could lead to an associated reduction in water consumption at the production stage.

A study commissioned by Airbnb found that accommodation booked through the site results in significant reductions in water use; during one year, the report suggests that Airbnb guests in Europe saved the equivalent of 1,100 Olympic-sized pools of water (Airbnb, 2014).

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