Estimating the abatement costs of hazardous chemicals

A review of the results of six case studies

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1. Introduction

In 2011, ECHA commissioned six studies to estimate the costs of abatement for a range of hazardous substances. The aim of the work was to improve capability in assessing the costs of reducing the use, emissions or exposure of hazardous substances, to explore and develop some of the theoretical and methodological aspects specific to hazardous chemicals, and to understand what barriers there might be to useful estimation. The work builds on the 2010 project *Abatement Costs for Substances of Concern*, sponsored by the UK Environment Agency, and supported by ECHA, the UK Health and Safety Executive and RIVM in the Netherlands.[1]

The studies commissioned by ECHA were undertaken by two separate contractors in 2012 and considered the costs of abating the following substances:

- Nonylphenol (NP) and nonylphenolethoxylates (NPE) used primarily as surfactants in the textile sector;
- Four phthalates (DEHP[2], DBP[3], BBP[4] and DIBP[5]) used as plasticisers in a wide range of processes and products;
- 1,4-dichlorobenzene (DCB) used in air fresheners and toilet blocks;
- Lead used for pellets in shotgun cartridges;
- HBCDD[6], a brominated flame-retardent used in polystyrene insulation material;
- MDA[7] for various uses, including polyimides and as a hardener in resins and adhesives.

For each of the substances, available data from the literature were supplemented with information from stakeholder consultation to construct cost curves. The primary aim of the exercise was to aid in developing the methodology for abatement cost and cost curve estimation in the chemicals sphere. However, some of the results have also contributed to assessments by ECHA and Member States of the feasibility of substitution. The purpose of this review is to summarise the results and discuss the principal findings, qualifications and so on, thereby providing a 'reader's guide' for the individual study reports which are attached as annexes to this document. When considering the results and findings, it should be remembered that, in undertaking the work, the contractors needed to make a number of assumptions to fill data and evidence gaps, which means that the resulting cost curves are subject to significant uncertainty. ECHA takes no responsibility for the assumptions and definitions employed in the reports, and does not necessarily agree with the findings or any opinions expressed in them.

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[1] Environment Agency (2011) *Abatement costs for substances of concern: Report on a pilot study – Methodology and indicative examples*, available at: http://a0768b4a8a31e106d8b0-50dc802554eb38a24458b98ff72d550b.r19.cf3.rackcdn.com/scho0811bucc-e-e.pdf

[2] Bis (2-ethylhexyl)phthalate

[3] Dibutyl phthalate

[4] Benzyl butyl phthalate

[5] Diisobutyl phthalate

[6] Hexabromocyclododecane

[7] 4,4'-Diaminodiphenylmethane

2. Why estimate abatement cost curves?

Abatement cost curves show what the marginal cost of abatement is at any given level of abatement, that is, the cost of achieving one additional unit of abatement. The units in which abatement is measured can vary depending on the context – use, consumption, emissions, exposures *etc* – but will always have some relationship to a 'bad' which is generated by the activity in question. In the case of the use of hazardous substances, the 'bad' will generally be negative impacts on human health and/or the environment. 'Abatement' refers to any measure which can be adopted to reduce the amount of 'bad' occurring – from the use of protective equipment to reduce personal exposure, filter mechanisms to reduce the emissions of hazardous substances to air or water, to substituting the use of the chemical with an alternative substance or technology. Conventially in economics, abatement costs curves are depicted as smoothly upward-sloping, as in Figure 1, which reflects the so-called 'law of diminishing returns'. Accordingly, as more abatement measures are adopted, the incremental amount of abatement (i.e. reduction in exposure, emissions *etc*) obtained from each successive euro spent declines; equivalently, as the total amount of abatement increases, the euro cost of each successive unit of abatement increases. This does not have to be the case (e.g. if there are significant economies of scale associated with particular abatement technologies) but is a reasonable starting point for illustrating the value of estimating abatement cost curves.

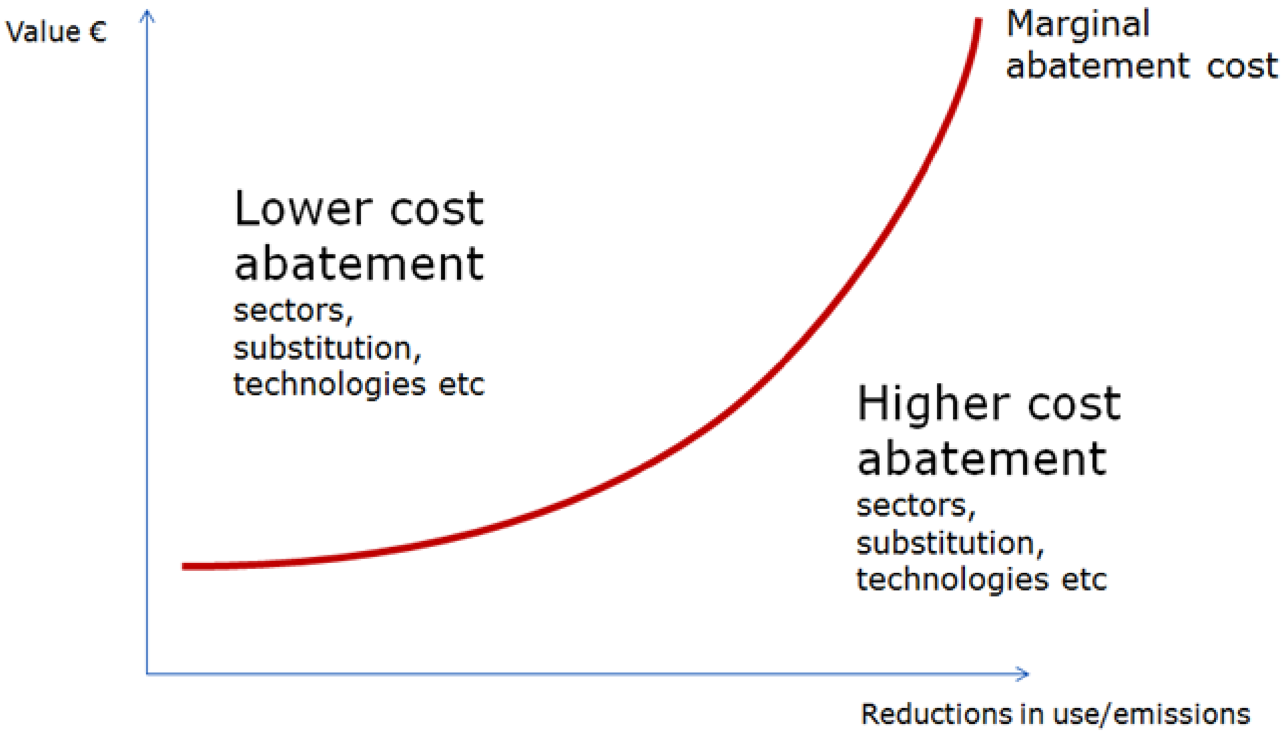

Figure 1: The `textbook view´of abatement cost curves

Knowledge of the marginal abatement cost curve can be useful in a number of ways. For instance, the marginal cost of abatement can be compared with the marginal benefit of abatement to establish what the optimal intervention level is, as in Figure 2. If the marginal cost of abatement is relatively low and the marginal benefit relatively high, it suggests that the adoption of abatement measures should be extended because the benefits of an additional unit of abatement outweigh the costs. In Figure 2, this is anywhere to the left of point X. Conversely, if the marginal cost of abatement is high relative to the marginal benefits, abatement is excessive and at least existing measures should be reduced or removed because the benefits they generate do not justify their costs. In Figure 2, this is anywhere to the right of point X. Only at point X, when the marginal cost of abatement just equals the marginal benefit, is no increase or reduction in abatement justified on cost-benefit grounds – that level of abatement is the economically optimal one.[8]

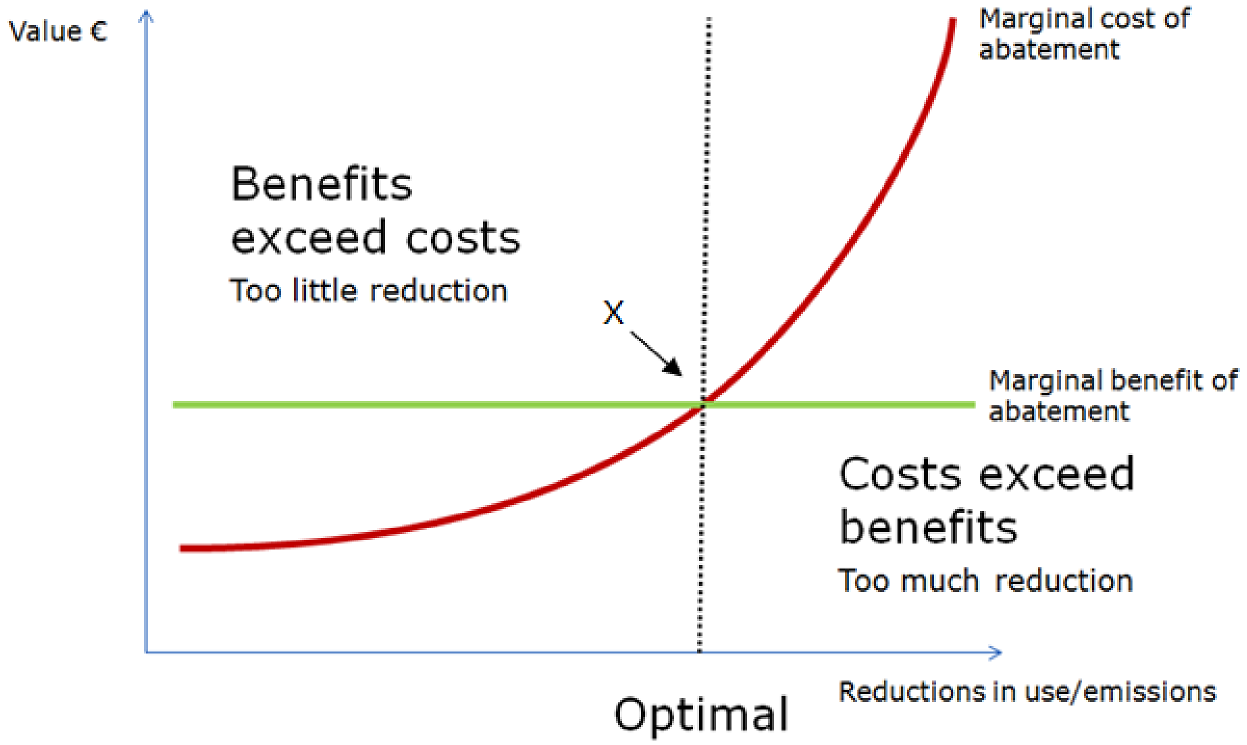

Figure 2: The optimal level of abatement

Obviously, such conclusions can only be drawn if appropriate estimates of the marginal benefits of the abatement are available. Often, however, particularly in chemicals regulation, these marginal benefits of reducing substance emissions or exposures will be unknown due to issues such as scientific uncertainty or a lack of quantitative evidence on human health or environmental impacts, or an absence of economic values of the impacts. In these cases, it might be possible to undertake cost-effectiveness analysis instead. This essentially compares the cost of abatement across different contexts – for example, different interventions, use sectors, countries and so on – to see where abatement is relatively cheap and where it is relatively expensive (Figure 3). This knowledge can be used to target abatement at where it is cheapest, so that abatement is achieved at no higher cost than necessary. If the marginal costs of abatement have been estimated previously for the same substance in relation to other interventions, this can be used to assess whether any new intervention seems to offer value for money which is any better or worse than these others.

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[8] Note that this is a 'textbook' representation and does not take account of any other requirements which might necessitate abatement on non-economic grounds.

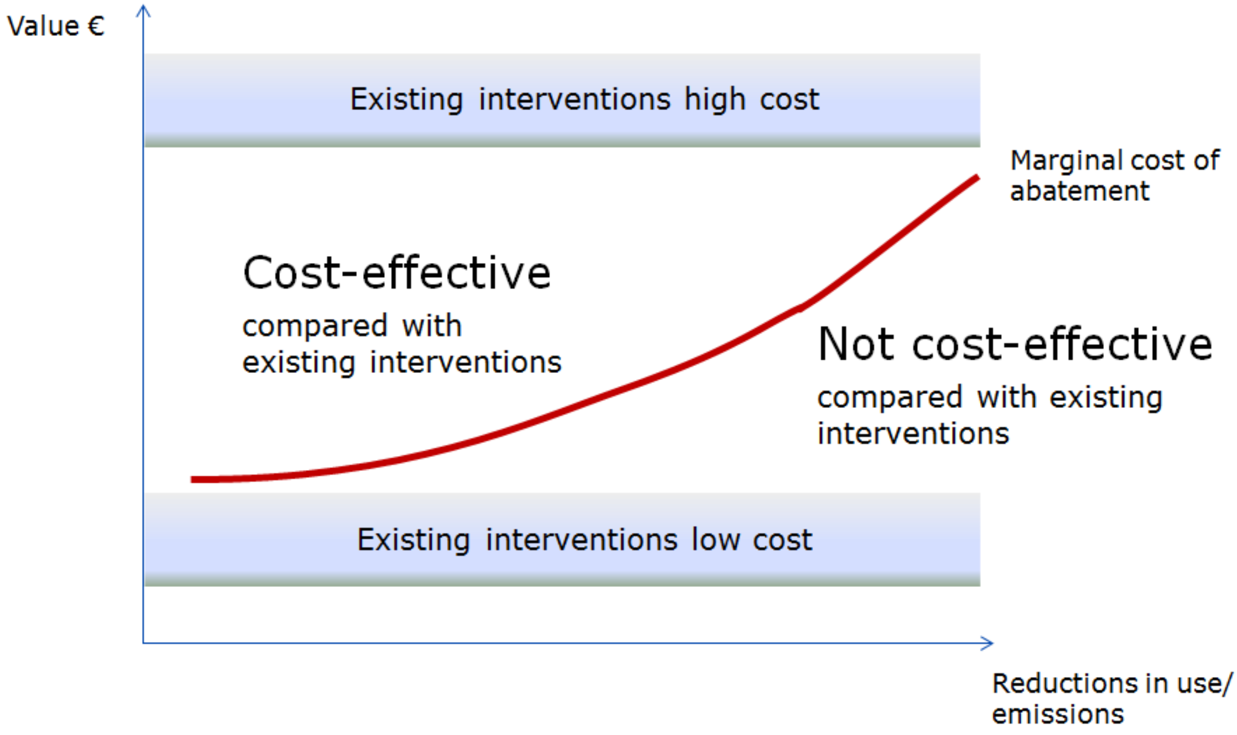

Figure 3: Cost-effectiveness of abatement compared with other interventions

Even if previous abatement cost estimates for the same substance in other contexts are not available, understanding how the marginal costs of abatement vary with the level of abatement – that is, the shape of the cost curve – within a given context can still be useful, because it can indicate the scope for cost savings through targeting. If the marginal cost curve is relatively flat, it means that the costs of abatement measures do not vary significantly within the context in question, and there is no particular advantage to be gained from targeting a particular total quantity of abatement (the '"first" tonne of abatement costs the same as the "last" tonne'), or any particular use or sector ('it costs the same to reduce emissions in one place as in any other'). If the marginal cost of abatement does vary a lot, however (e.g. the cost curve is very steep, or relatively flat and then 'kinked', after which point it becomes steeper, as shown in Figure 4), this means that it does cost significantly more to abate some uses or in some sectors than in others. In this situation, lower levels of abatement can be gained relatively cheaply whereas higher levels of abatement are much more expensive to achieve. Whilst, in the absence of evidence on the benefits of abatement, this information alone is insufficient to say what actual level of abatement is justified, it does imply that it will be possible to reduce total costs (significantly) by targeting the abatement where marginal costs are lowest.

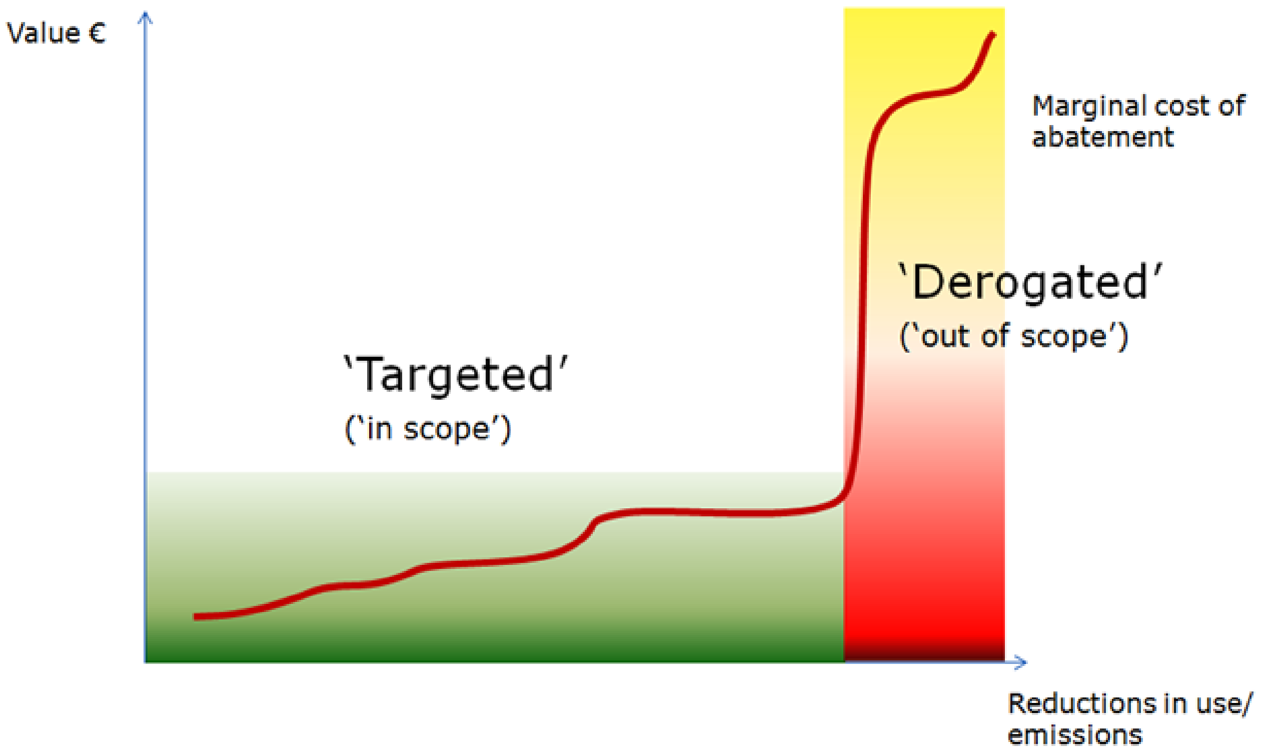

Figure 4: Targeted abatement

Clearly, for the costs of abatement for a particular substance to vary significantly with the quantity of abatement, there must be some variation in what can be done to reduce emissions or exposures by different amounts. This might be because the uses of the substance vary, and the technology needed to effect abatement for each use varies. The substance in question might be used across quite different sectors, which again might lead to quite significant variation in abatement technologies. Abatement costs might vary if the technologies employed rely on other inputs which are related to specific factors such as geography (e.g. availability of large water sources). Differences in life-cycle factors can lead to variations in the extent and timing of emissions, which might mean that the costs of abatement can vary when measured in emissions terms, even if they do not in terms of other units such as volumes used. There might be a large number of different abatement measures available, but their capacity and efficiency in effecting abatement might be limited and variable. This last situation is shown in Figure 4, where the marginal abatement cost curve exhibits a number of specific 'kinks', indicating where the abatement opportunities of a lower cost option (or set of options) are exhausted and the next more expensive option (or set of options) takes over, itself exhausted at the next 'kink'.[9]

Ultimately, for a given substance, the wider is the scope of any study to estimate abatement costs, the higher is the potential number of factors driving abatement cost functions, and the

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[9] Figure 4 shows a kinked curve, but as in the other figures, it is still continuous (unbroken), suggesting that the cost of abatement varies smoothly between different uses and contexts. In practice, curves are based on a number of distinct uses of different cost, and hence tend to be 'stepped' – flat for a given use (because data etc limitations mean it is the average abatement cost which is generally estimated for a use) and then jumping up discontinuously between uses as abatement becomes more expensive. This is how the curves (or perhaps it would be more accurate to describe them as 'envelopes') are presented in the case study reports. The discussion in this section is largely unaffected, however.

more likely it is that the costs will vary. As the scope narrows, the number of cost drivers declines, and the potential for variation in abatement costs is reduced. This would suggest that a more widely scoped study is likely to provide more interesting results from a policy-making perspective, in terms of demonstrating the potential for targeting to reduce the total costs of achieving a given level of abatement. However, when the scope of a study is wider, for a given level of study resources, it becomes more difficult to achieve the same level of analytical detail, and variation in abatement costs becomes more difficult to identify and measure (accurately). Thus, there is likely to be a tradeoff between analytical ‘interest’ and ‘accuracy’ when setting the scope of an abatement cost study.

Table 1: Summary of cost estimates obtained from the six case studies

Case study	*Cost (€/tonne)*	*Comments*
1 NP/NPE	from 80k to 2,000k from 108,000k to 429,000k	Lower cost range for switch to alcohol ethoxylates, with testing of imports. Higher cost range for end of pipe waste water treatment – these figures likely to be overestimates since co-benefits of treatment not accounted for.
2 Four phthalates		All figures per tonne emitted, rather than used (see text). Two ranges for DEHP relate to phthalate and non-phthalate alternatives (latter not expected to be adopted). Differences in functionality not accounted for (e.g. improved performance of DINP/DIDP compared with DEHP). 64% of BBP emissions accounted for one use at €7,000 per tonne.
DEHP	from 1k to 96k; from 2k to 166k	
BBP	from 1k to 170k	
DBP and DIBP	from 1k to 8k	
3 1,4-Dichlorobenzene	from -18k to 8k	Possible differences in odour-masking functionality not accounted for. Negative costs (i.e. savings) (for air fresheners and domestic toilet blocks) should be treated with caution.
4 Lead in shot gun cartridge pellets	9k	Compared with steel alternative. Impact (additional cost) of reduced ballistic performance not accounted for. Costs of testing and gun replacement possibly exaggerated.
5 HBCDD	from -190k to180k	Possible additional (e.g.) construction costs with alternatives not accounted for. Negative costs (i.e. savings) (for extruded polystyrene) should be treated with caution.
6 MDA	Not quantified	Limited data available did not allow quantification with meaningful level of uncertainty.

3. Findings from ECHA’s review of the six case studies

The reports relating to each of the case studies in this project are attached to this document. ECHA has undertaken a review of the content and the methodological learning points and other general findings are briefly presented below; summary assessments of the case studies are provided in the Appendix. The purpose of the review was to understand how successful the cost estimation exercises had been in each case and what particular problems might have been encountered, with a view to assessing the robustness of the results and learning lessons for any future activities in this field. The review was also designed to identify any methodological issues associated with estimating abatement costs in the chemicals sphere specifically. The strengths and weaknesses of the results in each case are also discussed in the reports themselves.

The costing results are summarised in Table 1. A number of points can be made in relation to them. First, it is evident that there is wide variation in the costs of abatement between and within substances. For instance, abatement costs for phthalates seem to start around €1,000 per tonne of substance emitted, rising to €200k per tonne; NP/NPE starts around €80k per tonne; and HBCDD can be as high as €180k. Lead in shotgun cartridges, 1,4-DCB and a large proportion of the phthalates seem to be abated at around €7-9k per tonne. However, just because one substance is cheaper to abate than another, this does not mean it is more justified, since the benefits of abatement will vary across substances too, and it is this comparison which determines the optimal level of abatement for a substance (Figure 2). Significant variation in costs for a given substance does, however, suggest the potential for efficiencies in abatement costs through targeting particular applications (Figure 4). Second, a limited number of cases indicate negative costs of abatement, suggesting that certain limitations, in this case reducing use of the substance, might actually generate savings because the alternatives are cheaper. However, these results might not be reliable because of difficulties in taking account in the analysis of differences in functionality between alternatives, so they should be treated with caution.

3.1 Limitations

In general, the constructed cost curves for these substances provide only limited information on which to base a prioritisation of abatement measures. Little variation in abatement costs was found by the contractor. Where it did seem to occur, it was related to a rather narrow range of cost drivers (e.g. variation in abatement costs for the four phthalates was driven almost entirely by differences in emission factors), or reflected the consideration of a limited range of abatement measures from extreme ends of the cost spectrum (e.g. alternative substances combined with testing in the case of NP/NPE in textiles, versus additional measures at wastewater treatment plants). A number of reasons for the limited usefulness of these results for prioritasion of abatement measures can be identified, including:

1. *Lack of study time and resources* – The amount of time and resources available for each case study was generally limited, which affected the amount and quality of information the contractors could obtain. In addition, the uses in some cases were quite minor and geographically specific (e.g. the domestic use of 1,4-DCB for air fresheners and toilet blocks) making the data gathering more challenging (e.g. in terms of linguistic requirements, market intelligence and finding contacts). Less than complete and perfect information might have reduced the accuracy of the results by limiting variation in abatement costs.

2. *Narrow scope of uses and available alternatives* – The range of uses considered for the substances in question was often narrow, with no obvious limit to the extent to which any single alternative could substitute for it. This meant that the resulting costs curves for each use were effectively straight lines based on a single alternative (e.g. 1,4-DCB in air fresheners and toilet blocks, lead in pellets for gunshot cartridges). Variation in abatement cost then reflected the different costs across uses and the limit of consumption or emissions in each use. Even where uses were quite broad (e.g. four phthalates in plastics production), there was a limited but quite suitable range of essentially drop-in alternatives available, and thus there was little scope or need to consider a wide variety of different contexts, technologies etc which might have generated greater variation in abatement costs.

3. *Unobservable cost drivers* – Even where a reasonable range of alternatives was potentially in scope of the study, it was often not possible to estimate accurate costs for them because their adoption would require changes in other inputs, or in the quality of outputs, which

were difficult to observe and/or value. For instance, 1,4-DCB is reportedly used in toilet blocks and air fresheners because of the substance's strong odour-masking capability in environments which are difficult to clean, but it was generally not possible to determine the value of this additional masking capability which would be lost if alternatives were adopted. Similarly, there is a range of non-lead alternatives to shotgun cartridges which are both higher priced and of lower performance, but the costs of any performance reductions could not be observed and hence could not be incorporated into the cost curves. In the HBCDD flame retardent case study, alternatives were considered which differed in terms of their insulation performance (as well as their inflammability). Adjustments were made to obtain 'insulation-equivalent' market prices, but could not account for changes in other inputs which might be necessary to achieve comparable performance (for example, higher labour and transport costs of using heavier materials). These variations in price and performance (as well as suitability for use in different contexts) between alternatives help to explain why multiple alternatives can continue to be available on the market even when one appears to dominate the others in simple price terms.

Because of these limitations, the results of the studies need to be interpreted very carefully, as described in the Appendix. Nevertheless, a number of the studies generated results which have been useful in the compilation of Annex XV dossiers (e.g. phthalates, 1,4-DCB). The limitations themselves have provided useful insights into abatement cost estimation which will assist in the design of future studies.

3.2 Methodological learning points

Despite the weaknesses of the case study outcomes discussed above and in the Appendix, the studies did provide useful results in some cases, as well as the opportunity to develop some methodological 'learnings points' relating to the definition and measurement of abatement costs, in theory and practice. These can be summarised as follows.

1. *Abatement cost definitions and measurement are endogenous to the substitution context* – The major portion of the abatement costs estimated in the NP/NPE case study was accounted for by the costs of testing clothing articles imported into the EU for the presence of NP. However, testing is not itself required to reduce NP use or emissions. In a voluntary context where a manufacturer chooses not to use NP (for whatever reason), there is no obvious need for such testing, and testing costs should not appear in the abatement cost curve. On the other hand, if there are continuing incentives to use NP and it cannot easily be observed whether it is being used or not, testing might well be needed to ensure regulatory abatement measures are being adopted, and the costs of this monitoring are correctly included in the abatement cost estimates. Thus, the definitions and measurement of abatement cost are dependent on the context in which they are being estimated, and not all types of cost will be relevant in all cases. This was recognised in the Environment Agency (2011) study (e.g. Section 3.2.5), and the NP/NPE case study provides a clear illustration.

2. *Treatment of residual capital equipment values in abatement cost curves is uncertain* – During the course of several case studies, it became apparent that abatement measures involving substitution away from the chemical(s) under study could involve costs to industry in terms of the redundancy, and hence loss of capital value, of existing capital equipment. This raised the issue of the appropriate treatment of these capital costs in the abatement cost function. At first glance, they seem genuine economic costs resulting directly from the regulatory requirement to reduce the manufacture and/or use of a substance. On the other hand, if the residual capital value is a genuine market value, a reasonable question is why the equipment cannot simply be sold. If the answer is that this is not possible because it has no alternative use, then this equipment would normally be classified in economic theory as 'sunk', implying that its opportunity cost is zero and there is no social cost from its redundancy. This in turn implies that any residual value assigned

to this capital is actually rent[10], and effectively a transfer to producers (the owners of the capital) from consumers, who would otherwise enjoy lower product prices and higher levels of consumer surplus.

This logic would point towards residual capital value not being included in abatement cost curves. However, even if a rent, this measure still seems to represent a genuine economic value – albeit one reflecting the utility gained by consumers rather than the opportunity cost of resources used in production. The costs of abatement in these situations are measured by (amongst other things) the increase in price charged to consumers who must switch to substitutes if they are no longer able to use the original substance. But if the original price they are being charged is 'inflated' by the existence of rents paid to producers, then this measure, based on market price differentials, will underestimate the true costs of the switch. Contrary to the previous paragraph, therefore, this logic suggests an argument that residual capital values should be incorporated into estimates of abatement costs (specifically in terms of an annualised value based on the remaining life of the equipment).[11]

This discussion was ongoing during the course of the current project, and the decision was taken to exclude residual capital costs from the estimates until it was resolved. This should be borne in mind when considering the results – including them would increase the estimates. Given their apparently ambiguous status – costs which appear to be borne by one party (firms) might actually be borne by another (consumers) – what seems clear is that there is potential for double-counting in the way residual capital is treated in the estimation of abatement costs. Note also that, even if it was concluded that residual capital costs are indeed 'sunk' and hence should not be included in abatement cost estimates, this does not mean that they might not be associated with real economic impacts – for instance, the effect on a company's financial position might reduce employment, investment and business performance, all of which might imply real economic costs.

3. *Differences in social and private perspectives can affect abatement option costs and prioritisation* – One aspect of estimating abatement costs which was acknowledged in the current costing exercise (although ultimately did not play a significant role in these case studies) related to the possible impact of adopting a private or social perspective. This is related to the issue raised under the first heading in this section, where whether the context was voluntary or regulatory in turn affected which types of cost were assumed would be incurred in the course of 'abatement'. The private-social perspective distinction is relevant because costs can be higher or lower depending on the perspective adopted, which in turn can affect the prioritisation of abatement options, and mean that what is socially preferred might not be privately preferred, and *vice versa*.

   Social and private measures of cost might differ for a number of reasons. Firms will not allways take account of the negative impacts of their activities on, for example, human health and the environment, and hence their costs measured from firms' private perspective will be lower than when viewed from society's. On the other hand, private firms tend to be less able to manage financial risk than broader society (e.g. governments), and hence activities which have significant risks associated with them will tend to be judged

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[10] Rent is value earned in excess of the (risk *etc*-adjusted) opportunity costs of the resources committed to the application. It is relevant to discussions of economic value which is measured by opportunity cost. Rent is generally seen as a source of inefficiency since it means that market prices do not accurately reflect opportunity costs, potentially leading to resource misallocation.

[11] The compliance cost guidance also includes a discussion of residual capital and its treatment (ECHA, 2010. Addendum to the Guidance on Socio-economic Analysis (SEA) – Restrictions: Calculation of compliance costs) (http://www.echa.eu/documents/10162/13576/appendix1-calculation__compliance_costs_case_restrictions_en.pdf).

more costly by firms than by society – examples would be large capital investments with long time horizons and payback periods.[12]

The significance of these social-private cost differences is that, where there is any discretion over what actions to take, firms will always take those which have the lowest cost when judged from their own perspective, irrespective of whether that is desirable from a social point of view. More specifically, the 'social abatement cost curve' might look quite different from the 'private abatement cost curve', and policies which do not take account of any such difference might fail to achieve their objectives and even generate an increase in costs for society rather than a saving. Even if policies can be used to regulate private firms' activities, they can only regulate what a firm *cannot* do, not what a firm *should* do. For example, a firm can always shift production overseas, or shut down altogether, if it finds the costs of so doing are less than those resulting from an action which the regulator attempts to force the firm to undertake. Even when a policy is set up to regulate those activities which should not be undertaken, a firm can still face a choice over whether to comply with the regulation and accept the costs of any required changes in behaviour, or to carry on illegally and accept the risks of sanction if its non-compliant behaviour is discovered. The key is understanding the role of the private perspective in driving policy outcomes, including how it affects the assessment of abatement costs and firms' abatement choices.

4. *REACH regulation provides obligations and incentive to provide data*

Feedback from the contractors undertaking the case studies, and from contractors undertaking similar work for the German Federal Environment Agency,[13] highlighted the difficulties encountered in obtaining useful and comprehensive data for abatement cost estimation. As mentioned above, this partly reflects the fact that much chemical usage and many types of chemical use are quite specific to particular areas, with an associated need to source consultancy resources which have the appropriate linguistic and geographical scope. In addition, these and other abatement costing exercises have often focused on substances whose use is limited to a small number of relatively minor applications and/or is already in decline due to previous regulatory initiatives or other pressures (e.g. 1,4-DCB). This can mean that information is often limited, out of date and difficult to identify. Finally, although standard sources like industry directories and Best Available Techniques Reference Documents (BREFs, produced by the Joint Research Centre of the European Commission) can be valuable for supplying information, in most cases by far the most important source of information will be industry itself, and the incentives for industry to participate in these studies are limited. In that respect, the REACH regulatory instruments of restriction and applications for authorisation could be helpful in providing a reason for firms to supply and use such cost information.

4. Concluding remarks

Abatement costing for chemicals is inherently complex. This was recognised at a workshop held by ECHA as part of the original Environment Agency project,[14] and has been demonstrated during the course of the current case studies. That complexity, especially compared with 'more traditional' abatement costing relating to, e.g., air pollution, is often

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[12] Some of the issues around discounting are discussed in the SEA guidance (ECHA, 2008. Guidance on Socio-Economic Analysis – Restrictions) (http://www.echa.eu/documents/10162/13641/sea_restrictions_en.pdf).

[13] Discussed at the workshop, 'The assessment of abatement costs of chemicals', in Berlin in March 2013

[14] Information on the workshop is available at http://echa.europa.eu/en/view-article/-/journal_content/a606d41a-950d-4db7-8eb0-8bc5833b4cd1.

related to the fact that hazardous substances, as well as potentially having negative impacts on human health and/or the environment, have quite particular functions which are the source of their value and the reasons why they are used in the first place. Abatement options, including substitution to alternative substances or technologies, often impair this functionality, which in turn implies reductions in value.

For instance, 1,4-DCB is used in air fresheners and toilet blocks because of its pungent, masking smell, but has been linked with cancer and lung disease via inhalation; lead used for pellets in shotgun cartridges provides good ballistic qualities lacking in alternatives such as steel; and the flame retardant HBCDD enables polystyrene to be used as an insulation material, instead of the bulkier and heavier alternatives such as mineral wool.

The challenge in estimating abatement costs in these circumstances is to account for these impacts on value and the costs of any mitigating actions, which might be adopted as a response. This is a challenge because the value of the specific function in question is generally not observable in the prices of products and the costs of processes (which are essentially bundles of input characteristics each with their own value or cost). Differences in price between alternatives are an inaccurate measure in these situations since the price of a substance is only one input into the total cost of the process, and using a ‘cheaper’ alternative might involve incurring costs in other ways. Hence, the cost of reductions in value, complementary inputs and/or mitigating actions associated with using alternatives are difficult to observe, measure and estimate, but knowledge of each is necessary to estimate abatement costs accurately.

The results of a number of ECHA’s abatement cost case studies were useful in the preparation of Annex XV restriction dossiers, but the curves estimated in the study in general exhibited limited variation in abatement costs. Hence the curves did not signal the potential for significantly increasing efficiency through targeting. Different factors explain this finding in each case, with the broad conclusion being that the wider the scope of the abatement cost study in question, the greater the potential for variation in costs, and the more likely is it that the abatement cost curve will be ‘interesting’ from a regulatory perspective. Unfortunately, the wider the scope of the study, the more difficult it is to observe relevant cost drivers, and the more costly it is to collect accurate data, so estimates are likely to be more prone to error and inaccuracy. This implies the potential for a tradeoff between cost variability (‘interest’) and project feasibility.

The studies to estimate abatement cost curves could be useful for higher level, strategic assessments of where best to intervene to regulate chemicals use, where the possibility for estimation error is of less concern. Abatement cost studies will still be highly valuable for the assessment of individual interventions as well, where more detail and accuracy are needed.

Abatement costing methodologies are well established, but there are some specific challenges in the area of chemicals, particularly related to the value and complementarity of outcomes, and the ECHA case studies presented here have demonstrated this effectively and usefully. The value of abatement cost curves comes from the potential for variation – it might be useful to try and look for indicators of this, perhaps using pilot studies to identify candidate cases using more rigour in specifying abatement options that has been thus far the case.

Appendix: Summary assessments of the abatement cost case studies

Nonylphenol and nonylphenolethoxylates in textiles

This study focussed on the use of nonylphenol and nonylphenolethoxylates (NPEs) in the textile industry. NPEs are used as surfactants, but have been associated with persistent and endocrine disruptive properties. Abatement costs were assessed for reducing emissions to surface water by substituting NPEs with alternative substances (including through a ban on imported textiles containing NPEs), or by installing emission reduction technologies at wastewater treatment plants. As might be expected, the first option was found to be clearly more cost-effective, based on the assumption that alternative substances would provide the same functionality as NP/NPE on a one-to-one basis.

Abatement costs for the wastewater treatment option were estimated at over €400 million per tonne of NPE emitted to surface water in the EU, compared with values for NPE substitution of up to €2 million per tonne. Wastewater treatment involves significant investment in long-lived physical capital, and amortising these costs over the reduction in emissions of a single pollutant would always be likely to produce figures for the abatement cost per tonne which greatly exceed the costs of a direct substitute for the substance in question. However, technologies to remove NPEs in wastewater treatment plants will also remove a number of other pollutants, and these potentially significant additional benefits, which would reduce the effective cost of NPE abatement, were not considered in the calculations due to information gaps. This omission arguably makes estimating the abatement cost curve a rather pointless exercise, since without any adjustment for the co-benefits of wastewater treatment, the two abatement measures are clearly not comparable in cost terms.

A further uncertainty concerned the additional testing of textile products, which it was assumed importers would need to perform. The costs of testing accounted for the majority of the estimated total costs of this option. Assumptions were made about, for example, the rate and time profile of additional testing and one-off costs of compliance reporting to suppliers and to customers. Clearly, costs such as these relate to particular types of regulatory regime and would not need to be incurred in a scenario in which emission reductions were the result of, for example, voluntary industry action to substitute. Meanwhile, a simple abatement technique such as additional washing of textiles at place of origin was not considered due to difficulties in cost estimation (although whether such an approach is considered as providing additional abatement depends on how much weight is placed on non-EU NPE emissions).

Overall, the scope for possible substitution opportunities was underestimated in this case study, and the costs of substitution overestimated. The potential for reducing total abatement costs through targeting was likely to have been underestimated, due to the contractor's identification of only two abatement measures at opposite ends of the cost spectrum.

Four phthalates (DEHP, BBP, DBP and DIBP)

The assessment of abatement costs of the four phthalates focused only on substance substitution, and did not consider other measures for reducing the emissions of the substances, e.g. pollution control in manufacturing.

Extensive experience exists of replacing DEHP and BBP with alternative phthalates and the uncertainties on the costs of substitution are mainly due to variations in price differences between the alternative phthalates and DEHP and BBP. Alternatives exist which can act as drop-in substitutes for nearly all applications and the costs of substitution of one tonne of the phthalates is largely the same for all application areas. For DBP and DIBP the substitution pattern is more diverse, with a wider range of alternatives, but with similar properties and prices to the existing substances.

Cost curves showing the costs of reducing the consumption of each of the phthalates, using least cost alternatives (principally alternative phthalates), were consequently practically straight lines, indicating that the costs of replacing each of the phthalates did not vary across different applications. The abatement costs are also similar between phthalates – for DEHP and BBP around €200 per tonne, and for DBP/DIBP around €300 per tonne.

The picture changes quite markedly when abatement costs are expressed per unit of emissions, rather than consumption. This was done using life-cycle emission factors for the different application areas, obtained from a variety of different literature sources, and representing average emission factors over different emission sources and compartments over the entire life cycle of different products/applications. From this perspective, phthalate alternatives for all four phthalates would start at around €1,000 per tonne. Most emissions of DBP/DIBP were estimated to be abated at a cost of around €8,000 per tonne, and almost 90% of both DEHP and BBP emissions could be abated at a cost below €7,000 per tonne. However, the most expensive BBP and DEHP substitutions were estimated to rise to €100,000-€200,000 per tonne.

These cost variations were the result almost entirely of differences in the associated emission factors for different applications. The authors of the case study argued that this meant that abatement costs themselves were not useful in prioritising abatement options in this case, since the same ranking would result from using the emission factors alone. This might be true, but this conclusion could only have been reached after the abatement costs had been calculated. Moreover, a ranking based on emissions factors would not be helpful if the intention was to compare the marginal cost of abatement with estimates of marginal damage from the substance, or with estimates of the marginal cost of abatement in other contexts (for cost-benefit or cost-effectiveness purposes respectively). Hence these results illustrate the potential importance of incorporating emission factors into abatement cost analysis, but do not obviate the need to undertake cost analysis in the first place.

1,4-Dichlorobenzene in toilet blocks and air fresheners

The assessment of abatement costs of 1,4-Dichlorobenzene (1,4-DCB) in toilet blocks and air fresheners considered only replacing the products with substitutes based on alternative substances. Accordingly, the main challenge in assessing abatement costs in this case related to the need to compare different characteristics of the products. 1,4-DCB is used principally for its odour-masking function, particularly in situations where effective cleaning is difficult (e.g. due to the physical design of the sewage drainage system at a given site). However, available alternatives tend to be cleaning products primarily, with fragrances added for aesthetic reasons rather than for any odour-masking purposes specifically. As a result, there are few if any one-to-one alternatives which would provide exactly the same functionalities as 1,4-DCB.

The differences in odour-masking and cleaning capability of 1,4-DCB and alternative products are mentioned in the case study report, but practical difficulties in taking account of them mean they are not reflected in the cost calculations. For instance, the cost of any reduction in odour-masking ability depends on the impact on amenity value experienced by users of the toilets and other facilities in question, and this is not observable directly. Any loss of amenity value might be reduced by compensating action such as manual cleaning, but how much additional cleaning would be undertaken would depend on its effectiveness and cost. Hence, the full cost of reduced functionality from switching to alternatives could not be accurately estimated, and price differentials between 1,4-DCB-based products and alternatives cannot be assumed to provide a full reflection of this cost. This is likely to have lead to the costs of switching to non-1,4-DCB-based alternatives being underestimated in the report.

Furthermore, the cost of alternatives is affected by technical parameters. For instance, the effective cost of alternative products is affected by how long they last and how regularly they need to be replaced – and alternatives tend to have a shorter lifespan than 1,4-DCB-based products for a given flushing frequency. This was considered in the cost calculations by assuming an average user with average longevity of alternative products, whereas in practice the circumstances in toilets and associated longevity will differ.

The results of the case study suggest that 1,4-DCB-based air fresheners and domestic toilet block products are more expensive to use than alternative products (*minus* €18,000 per tonne, compared with alternative toilet block products for professional use with a cost of €8,000 per tonne). The fact that these products continue to be used (albeit in a very limited way) when cheaper alternatives exist could be explained by the failure, as described above, to take account of the costs of any reductions in functionality when moving to alternatives, and the costs of any associated compensating actions. If this is the case, the abatement cost estimates can be regarded as unreliable, since they indicate a negative cost of abatement when the opposite is true. If abatement costs are, in fact, negative (indicating that the consumer would save money by using the alternative), a number of reasons could be possible, including that users of 1,4-DCB-based products are unaware of alternatives or misperceive their functionality (e.g. by mistaking odour-masking for cleaning). In the absence of any evidence one way or another, the current estimates should be treated with caution.

Lead in shot

Lead used for pellets in shotgun cartridges has been linked with impacts on the health of wildfowl populations in wetland areas. This study considered the costs of replacing this lead with a number of alternative metals in cartridges for wetland use as agreed by most of the EU Member States and the Union as a whole under the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA). As a comparison the study considered also the costs in other hunting, too. Cartridges using these alternative materials are either somewhat higher in price compared with lead-based ones (e.g. prices of steel-based cartridges assumed to be up to 20% higher), or substantially more expensive (e.g bismuth, four to five times more expensive; tungsten, four to seven times more expensive). There were assumed to be no (long-run) limits to the capacity to supply the cartridge market with products based on alternatives, and hence it was assumed that all affected hunters would simply switch to the lowest-cost alternative. This resulted in an abatement cost curve which is a straight line, with an estimated cost per tonne of lead abated equal to €9,100. The contractor assumed no difference in cost between abatement in wetland areas only and in general hunting.

A number of remarks can be made about this result. First, the quality of information and data available was generally patchy. Comprehensive price data were obtained for only one Member State. Information on hunting and gun ownership across the EU exhibited features suggesting possible incompleteness (e.g. the number of birds reportedly shot annually in France, with just over 1m hunters) was ten-times that reported in Germany (200,000 hunters), and 100-times that reported in Poland (100,000 hunters), differences which seem difficult to explain simply in terms of participation or hunting practice). More generally, it proved difficult to gain good intelligence on the impacts of these policies and how hunters had responded to them (beyond that provided by the 2004 report by COWI). This was despite the fact that two EU Member States (Denmark and the Netherlands) have completely banned the use of lead in shotgun cartridges, and a further 14 have complete or partial bans on the use of lead for hunting on wetlands (due to their legal obligations under the AEWA). In the absence of such information, some of the assumptions made to generate abatement cost estimates could be regarded as speculative and pessimistic. For instance, the contractor assumed that 95% of existing guns would be tested to check their compatibility with steel ammunition, when compatibility might already be known for many guns; in addition, instead of incurring testing costs, hunters might prefer to use more expensive alternatives like bismuth, which, according to COWI (2004), are compatible with existing guns and would not necessitate such testing.

Perhaps the major gap in the cost estimation related to how substitutes for lead affect shooting performance and the impact on the shooting experience. The alternative which was assumed to be the ‘next best’ to lead was steel, which is harder and lighter than lead, leading to reduction in shooting distances and accuracy. This would mean that the implied one-to-one substitution between lead and steel assumed by the contractor does not in fact hold, and that hunters might use more ammunition in total if they switched to steel. Reductions in performance might also reduce the enjoyment that hunters get from their sport. As with other case studies discussed here, it can be seen that alternatives can be expected to be associated with differences in functionality which are difficult to take account of but which might have a significant impact on estimates of the costs of abatement.

Hexabromocyclododecane (HBCDD)

HBCDD has been identified as a Substance of Very High Concern (SVHC) pursuant to Article 57(a) of REACH, as it is classified as a PBT substance, and is now included in Annex XIV and subject to authorisation, with a sunset date in 2015. It is primarily used as a flame retardant for polystyrene foam used in insulation and other construction applications, and the report considered only alternative insulation materials which would not necessitate the use of HBCDD. Alternative flame retardant substances are available for polystyrene foam, or are under development, but insufficient information was available at the time of preparing the report to allow a meaningful cost assessment. Abatement costs were not assessed for other uses of HBCDD.

The main challenge in assessing the costs of substituting plastic insulation materials containing HBCDD with alternative materials, such as mineral wool, stemmed from the different characteristics of the good. As with the 1,4-DCB case study, these alternatives are generally not one-to-one substitutes providing exactly the same functionalities as (e.g.) polystyrene foam. In particular, although flame retardant capabilities are treated as comparable in the report, the insulation performance of the different materials differs. To account for this, the assessment adjusts the prices of competing materials by the differences in thermal properties, to obtain a price ‘per thermal unit’. However, using alternatives might also require the use of additional inputs, and the assessment does not account for these. For instance, heavier products might imply higher transportation and labour costs, while achieving equivalent insulation performance with lower density materials might require wider wall cavities and hence building modifications and ultimately redesign.

The report estimated the costs of using mineral wool in external walls, and polyurethane foam in walls and floors, in place of polystyrene with HBCDD. The costs were generally positive for expanded polystyrene (from €25,000 to €175,000/te) and negative for extruded polystyrene (from -€67,000 to €187,000/te). The report’s authors suggested that this reflects the fact that the costs:

‘do not include the (potentially significant) cost implications that could occur as a result of the lost market, and lost residual value of capital equipment, for the current suppliers of HBCDD-based EPS and XPS to the insulation market. If such costs are included, it is therefore possible that there could be significantly higher costs for substitution of EPS and positive rather than negative costs for substitution of XPS.’

However, although the costs of alternatives would certainly increase if factors such as residual capital values were incorporated into the analysis, the estimates are based on actual market prices, and if materials are being used despite alternatives being available at a lower price, this must be because these lower prices do not reflect the full cost implications of switching. As with the 1,4-DCB case study, in the absence of an explanation as to why the estimated costs do not seem to be consistent with actual behaviour, the results should be treated with caution.

MDA

The assessment of abatement costs for MDA considered only identified non-intermediate uses, as these were considered to be most 'REACH relevant' (MDA is listed in Annex XIV of REACH, and use beyond the sunset date is therefore subject to authorisation; however, intermediate uses of Annex XIV substances are exempt from the authorisation requirement.). Identified non-intermediate uses included use as a hardener in epoxy resins and adhesives, and use in polyimides (PMR-15). The available information suggested that only use in polyimides is currently taking place in the EU, although it cannot be excluded that other non-intermediate uses exist as well.

The main difficulty in assessing abatement costs of MDA in polyimides stemmed from the very limited availability of data, which did not permit the calculation of abatement costs or the construction of abatement cost curves with any meaningful level of uncertainty. As a result, only a qualitative description was provided in the report.

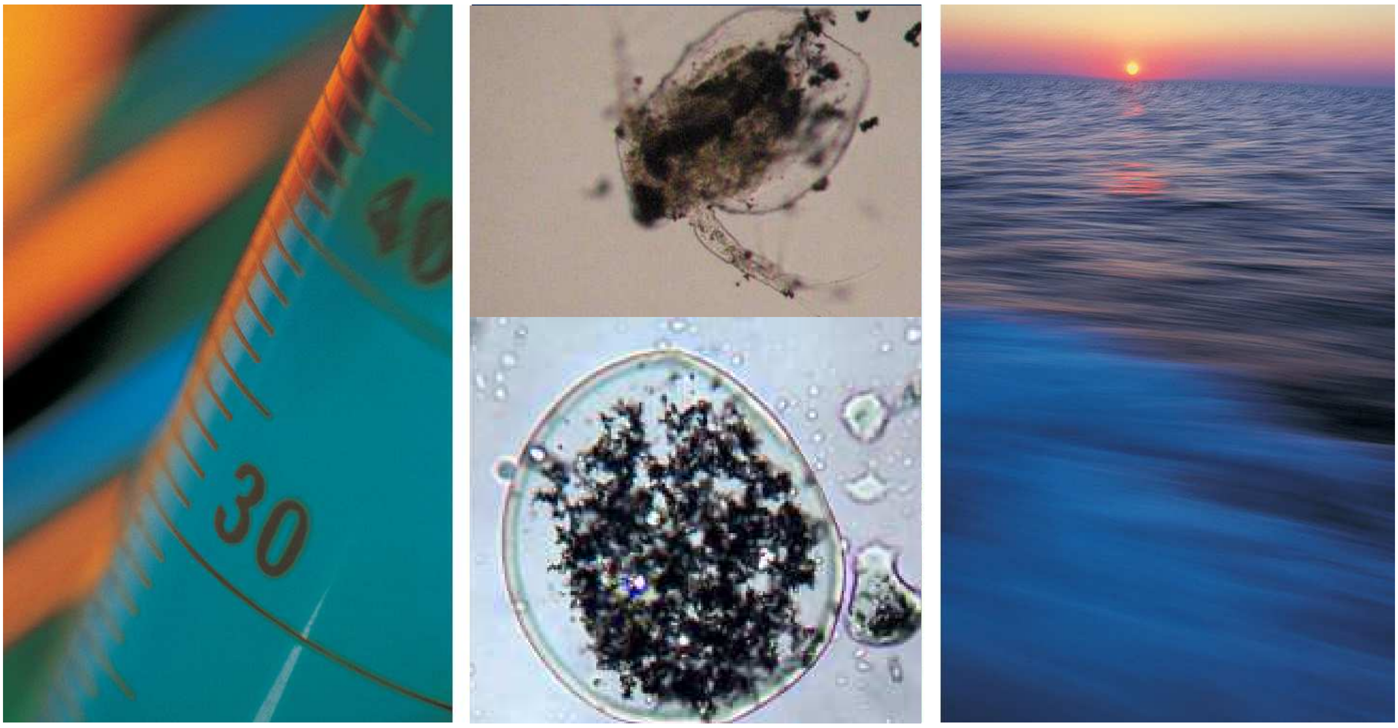
ANNEXES:

Annex 1: Nonylphenol and nonylphenol ethoxylates in textiles
Annex 2: The four phthalates (DEHP, BBP, DBP AND DIBP)
Annex 3: 1,4-Dichlorobenzene in toilet blocks and in air fresheners
Annex 4: Lead in shot
Annex 5: Hexabromocyclododecane (HBCDD)
Annex 6: MDA

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Nonylphenol and nonylphenol ethoxylates in textiles – Final Report



AMEC Environment & Infrastructure UK Limited

June 2012

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European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Nonylphenol and nonylphenol ethoxylates in textiles – Final Report

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Contents

Executive Summary

This report provides the results of work on collection of data on abatement costs of reducing the use and/or environmental emissions of nonylphenol and nonylphenol ethoxylates in textiles. The work involved developing cost curves based on information readily available in the literature in combination with consultation with industry. It has been undertaken on behalf of ECHA by AMEC.

The report includes best estimates of the current production volumes of NP and NPE in Europe. There are thought to be less than five producers of NP in Europe with an overall production level of between 10,000-50,000 tonnes/year (exact figures are confidential). NPE production in the EU is estimated at 32,000 tonnes/year. In terms of usage, it was estimated that approximately 8,000t per year of NPEs were used in the textile industry in Europe in 1999. Following the introduction of a restriction on the use of NPE for textile processing in the EU in 2005, the volume of NPEs used by EU textile mills in processing is thought to have decreased by to a maximum of 5,000t per year. Releases to the environment (assumed to be surface water) from the remaining uses have been estimated as around 10t per year, reflecting the fact that the existing restriction prohibits use except for processing with no release into waste water or where process water is pre-treated to remove the organic fraction completely.

Whilst usage of NPE in EU textile processing has decreased, countries outside the EU are understood to still widely use NPE in textile processing as a detergent after dying as it is inexpensive and highly effective. Typically some of the NPE remains as a residue on the textile and this residue is easily washed out. Therefore, emissions of NPE (and, thereby, NP) are possible via the washing of textiles containing NPE that originated from outside Europe. In total, it is estimated that perhaps 800-900 tonnes of NPE (320-360t NP_{eq}) could be washed out of imported fabrics and clothing apparel each year in this way.

Data have been collated on the potential of a number of different risk reduction measures to reduce use and/or emissions of NPEs from textiles, both domestically-produced and imported. Information on the potential costs of these measures has also been developed.

The cost curves developed illustrate the relative costs, in €/t, of reducing emissions of NPE from textiles to surface water. The cost curves illustrate that legislative measures, in the form of a restriction, could be introduced which would eliminate effectively 100% of NPE emissions to surface water (estimated at close to 400t per year). Alternatively, tertiary WWTP measures, applied at a large number of WWTP across Europe, could be implemented which would eliminate around 40% (about 160t) of NPE emissions to surface water. Whilst the underlying assumptions are subject to a significant degree of uncertainty, the legislative measures (restrictions) are found to be far more cost-effective than the WWTP measures. Specifically, a ban on NPE in imported textiles (above a minimal threshold) is likely to be by far the most cost-effective way in which to eliminate the majority of emissions (97%); removal of the remainder would require a further ban on EU-produced textiles. The total cumulative cost of possible restrictions is estimated to be around €800m, whilst to achieve the maximum reduction identified for WWTP, the total cost could be around €70bn. However, the costs for advanced WWTP measures do not take account of the co-benefits of removing other pollutants at the same time. These have not been quantified for the purposes of the current study but they are likely to be significant.

The costs of restrictions are based on several key assumptions and the calculated results are highly sensitive to these assumptions. Firstly, it is assumed that substitution costs to textile producers located outside the EU would be passed on to downstream European importers. It is not clear whether this would happen in practice as responses from consultation on this issue were mixed. Secondly, the measures assume 100% replacement of NPEs with alcohol ethoxylates (AE). This is a simplistic analysis as, in practice, it is likely that a range of alternatives would be used, depending on the specific fabric and final function. Consultation with industry suggests that for companies using NPEs in polymeric dispersion for coating ‘technical textiles’ in Europe it may not be a simple case of substituting NPE with AE (although this represents only a small fraction of the total use, emissions and costs calculated in this report). More sophisticated cost curves would require further data collection and analysis in order to understand the likely uptake of different alternatives.

Thirdly, assumptions have been made regarding the likely response of industry to a proposed restriction on the placing of the market of imported textiles containing NP/NPE. It has been assumed that 50% of importers would conduct additional spot testing of textiles (the remainder are assumed to rely on communication from suppliers or to already undertake spot testing of NPEs in the baseline scenario). A testing frequency of 0.05% of textiles in the first year following the implementation of the restriction is assumed, with that frequency decreasing to 0.005% in subsequent years. These costs would in practice depend on the scope and definition of a restriction, as well as on enforcement requirements, such as whether companies would need to demonstrate compliance by testing or whether supply chain communication alone might be considered sufficient.

The assumptions regarding the potential to achieve reductions in emissions to surface water through tertiary WWTP measures are also subject to significant uncertainty. The underlying data on the amount of NPEs present in imported textiles (and the associated releases to the environment) are not based on extensive monitoring of imported textiles and the picture of current and potential future releases from existing WWTP are also based on relatively generic assumptions from existing literature sources.

What is clear, however, is that the potential to reduce emissions through a restriction is likely to be significantly greater than through additional measures at WWTP, and the costs are likely to be significantly lower. The drafting of any such restriction, and the activities associated with enforcement, will have a significant effect on how companies achieve and demonstrate compliance, and hence on the total costs.

1. Introduction

1.1 The Project

The European Chemicals Agency (ECHA) has commissioned a project to provide information on "abatement costs for certain hazardous chemicals" (contract number ECHA/2011/140). The work is being undertaken by AMEC Environment & Infrastructure UK Limited ("AMEC").

The present report is intended to provide a summary of the data collected on abatement costs of reducing the use of nonylphenol (NP) and nonylphenol ethoxylates (NPE) in textiles.

The data collected is intended to be used for:

- Supporting the Agency in assessing the most appropriate risk management options for the substances addressed;
- Furthering the understanding of the usefulness of data on use/emissions abatement costs in risk management decision-making; and
- Supporting the Agency (and potentially others) in the preparation of restriction dossiers.

1.2 Project Context

This project follows on from a 2010 project on "*Abatement cost curves for substances of concern*" conducted by AMEC Environment & Infrastructure UK (previously Entec UK) for the Environment Agency, ECHA, the Health and Safety Executive (HSE) and RIVM. The main aim of that project was to develop a suitable method for estimating abatement costs to reduce emissions of chemicals and to apply and test the method with three selected case study substances. That study provided a first illustration of the benefits of being able to compare unit abatement costs amongst different substances and different uses.

The objective of this project is to assist ECHA in establishing capability to assess the abatement costs of reducing the use or emissions of hazardous substances. Under this lot ("Lot 2"), abatement costs of NP and NPE have been assessed specifically in: imported and EU-manufactured textiles.

NP/NPEs are not currently included in the Annex XIV (List of substances subject for authorisation) but are subject to various restrictions under Annex XVII. ECHA was originally going to prepare an Annex XV restriction dossier on request from the Commission but the Swedish Chemicals Agency (KEMI) included their plans to do so on the pre-RoI. Under this Lot, abatement costs to reduce emissions (and exposure) have been assessed as well as abatement costs to reduce usage. Only usage in textiles has been covered.

The main outputs of the work, for this substance and for the other substances being assessed under different lots are expected to be as follows:

- Data on abatement costs of reducing the use or emissions of the chemical. Different applications of the same substance may/will introduce different abatement costs; and

- An overview on the functioning of the markets for the substances in question, including information on prices, amounts of the substance on the markets (including import and export), the relative shares of the substance used for different applications, number of actors involved in the business, as well as possible trends in the relevant market.

1.3 Structure of this Report

This report is structured as follows:

- Section 2 of this report provides an overview of the process of data gathering for this study and presents an overview of the data received;

- Section 3 sets out an overview of the market of NP/NPE;

- Section 4 presents the data analysis and resulting abatement cost curves; and

- Section 5 presents conclusions.

The appendices to this report include various other background data.

2. Data Collection

2.1 Overview

The aim of the study was to gather abatement costs data for NP and NPE specifically in textiles. This chapter briefly summarises the process of data gathering for this study and presents an overview of the data received.

2.2 Review of Existing Data Sources

KEMI provided numerous sources of information at project inception including analysis of available alternatives (KEMI, 2011a) and analysis of NP/NPE occurrence in textiles (KEMI, 2011b). This information has been supplemented with further relevant sources identified during the course of the study.

2.3 Stakeholder Consultation

A questionnaire to collect the required information was developed in collaboration with ECHA. This was used as a basis for collecting information via telephone and written consultation with:

- Manufacturers, formulators and importers of NP/NPEs and their trade associations;
- European textile producers using NP/NPEs or alternatives; and
- European textile importers and retailers.

The questionnaire was also sent to key trade associations representing the textile industry. In total, 41 organisations were contacted (see Table 2.1). Of these, 22 organisations provided information for the study, although only two questionnaires were completed and returned. There are, therefore, some relatively large data gaps. A list of consultees is presented in Appendix A.

Table 2.1 Overview of Stakeholder consultation

Stakeholder categories	Number contacted	Number of responses
Associations		
A.I.S.E.	18	10
AEDT – European Association of Fashion Retailers		
AEPSAT		
British Apparel and Textile Confederation		
CEPAD		

Table 2.1 (continued) Overview of Stakeholder consultation

Stakeholder categories	Number contacted	Number of responses
CESIO		
CIA (UK Chemicals Industries Assocation)		
CIRFS		
Euratex		
Eureau		
The European Outdoor Group (EOG) (Sustainability Working Group)		
Fedustria (Federation of Flemish Textile Companies)		
International Wool Textile Organisation (IWTO)		
LTMA, Lancashire Textiles Manufacturers Association		
TEGEWA		
Textil-Bekleidung		
The Textile Industry		
Union des Industries Textiles de France		
Industry		
Akzo Nobel (Sweden)	23	12
BASF AG (Germany)		
Centexbel		
Clariant		
COHIBA Project		
Concordia Textiles (Waregem)		
Detic		
Dow Chemicals		
Eurofins Scientific		
Huntsman Textile Effects		
Marks and Spencers		
Masureel Veredeling (Wevelgem)		
Oeke-Tex Association		
PCC Synteza SA, Kedzierezyn-Kozle (Poland)		
Rhodia (part of Solvay Group)		
Sasol Germany/UK		
Shirley Technologies		
SI Group Inc. (switz)		

Table 2.1 (continued) Overview of Stakeholder consultation

Stakeholder categories	Number contacted	Number of responses
Stepan UK		
Sasol Germany		
Testex		
Textilimportprerna		
Uniqema UK (now Croda)		

In addition, two Member State competent authorities provided information that has been used in the study: the Swedish Chemicals Agency (KEMI) and the Anglo-Welsh Environment Agency.

3. General Market Overview

3.1 Introduction to the substances

‘Nonylphenol’ (NP) refers to a large group of isomeric compounds of the general formula $C_6H_4OH)C_9H_{19}$. Nonylphenols may vary in two ways: the substitution position of the nonyl group on the phenol molecule; and the degree of branching of the nonyl group. There are several CAS numbers for nonylphenols (see Table 3.1).

Table 3.1 CAS Numbers for Nonylphenols

Substance	CAS Number	Comment
Nonylphenol, branched nonyl chain in para position	84852-15-3	
Nonylphenol, branched nonyl chain	90481-04-2	
Isononylphenol (mixed isomers)	11066-49-2	
4-Nonylphenol, straight nonyl chain in para position	104-40-5	
Nonylphenol, straight nonyl chain, not necessarily in para position	25154-52-3	This CAS No. previously covered all nonylphenols

Nonylphenol ethoxylates (NPEs) are produced by reacting NP with ethylene oxide (EO) under basic conditions. The degree of ethoxylation depends on the molar ratio of NP to EO. There are several CAS numbers for NPEs (see Table 3.2).

Table 3.2 CAS Numbers for commonly used Nonylphenol ethoxylates

Substance	CAS Number
Nonylphenol polyethylene glycol ether	9016-45-9
p-Nonylphenol polyethylene glycol ether	26027-38-3
Poly(oxy-1,2-ethanediyl), alpha(isononylphenyl) omega-hydroxy	37205-87-1
Poly(oxy-1,2-ethanediyl), alpha(4-nonylphenyl) omega-hydroxy, branched	127087-87-0
Poly(oxy-1,2-ethanediyl), alpha(nonylphenyl) omega-hydroxy, branched	68412-54-4

3.2 Manufacture, import and export

3.2.1 Historical information

Nonylphenol was first synthesised in 1940 (Soares et al. 2008). There are three main methods used to manufacture nonylphenol: (1) phenol and mixed nonenes are reacted in the presence of a catalyst in a batch process; (2) phenol and mixed nonenes are reacted in the presence of a sulfonated ion exchange resin in a batch process, or (3) phenol and mixed nonenes are reacted in the presence of a fixed bed ion exchange resin in a continuous process. Commercially produced nonylphenols are predominantly 4-nonylphenol with a varied and undefined degree of branching in the alkyl group. Very little straight chain nonylphenol is produced in the EU (EU RAR, 2002).

According to the EU RAR (2002) in Europe, in 1997, four companies were producing NP. The total EU production was estimated to be 73,500t (RPA, 1999). Imports were estimated to be around 9,000t and exports at around 3,500t. Therefore, net production was estimated at 79,000t

In 2006, there were estimated to be three companies producing NP in Europe (Feenstra, 2009):

- Sasol Germany GmbH;
- Polimeri Europe, Italy; and
- Synteza, Poland.

Globally, the annual production of NP reached 154,200t in the USA, 16,500t in Japan and 16,000t in China (Soares et al., 2008).

However, NP is also a breakdown-product of the non-ionic surfactant nonylphenol ethoxylate (NPE). Thus, NPE sources are also possible NP sources.

NPE is produced by the ethoxylation of nonylphenol. According to the EU RAR (2002), production of NPEs was estimated at 109,808t in 1994 and 118,000t in 1997 in the EU.

3.2.2 Current best estimates

Table 3.3 shows best estimates (based on available literature and consultation) of total volumes of NP/NPE manufactured in, imported to and exported from the EU.

The current production volume of NPs is estimated to be between 10,000 - 50,000t per year in Europe. According to Eurostat, imports were estimated to be around 6,000t and exports from the EU at around 2,000t. Therefore, current net consumption of NPs is estimated to be between 14,000 – 54,000t per year.

According to CEPAD, in 2010 in the enlarged EU (e.g. EU-27, Norway and Switzerland) production of alkylphenol ethoxylates (APEs) was approximately 32,000t, of which the majority was NPEs[1].

Table 3.3 Production volume, exports and imports in EU (amount in tonnes/year)

	1997[1]	**2008**	**2009**	**2010**
Production volume NP, EU	73,500	No data	No data	10,000-50,000
Exports NP from EU	3,500	1,000[2]	1,000[2]	2,000[2]
Imports NP into EU	8,500	3,500[2]	3,000[2]	6 000[2]
Tonnage (Use)[3]	**78,500**			14,000 – 54,000
Production volume NPE, EU	118,000			32,000[5]
Exports NPE from EU[4]	2,200 (5,600)			
Imports NPE into EU[4]	18,000 (46,000)			

Notes:
1) EU RAR (2002)
2) Eurostat; Octylphenol included
3) Production volume + imports – exports
4) Weights as NP (Weights as NPE in brackets). 1 unit weight NPE = 0.4 unit weights of NP
5) CEPAD estimate. Includes all APEs

3.3 Current uses

The use of NP can be divided into five categories:

1. Industrial production: as an intermediate in the production of other substances;
2. Industrial production: in the manufacturing of articles;
3. Professional use in industry;
4. Professional use in areas other than the manufacturing industry;
5. Consumer use.

The European Council for Alkylphenols and Derivatives (CEPAD) has provided a summary of the markets and uses of NP in Western Europe in 2010. This is illustrated further in the figure in Appendix B.

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[1] Personal communication with CEPAD. March 2012.

3.3.1 Intermediate use of NP

NP is mainly used as an intermediate in the production of other chemicals, predominantly nonylphenols ethoxylates but also a number of other nonylphenol-derivatives such as nonylphenoxy acetic acids (RPA, 1999; EU RAR, 2002). Within the EU, NP is used in three main applications:

- In the production of NPEs (the main use);
- In the manufacture of resins, plastics and stabilisers; and
- In the manufacture of phenolic oximes (EU RAR, 2002).

Best estimates of the use of NP as an intermediate in the EU are presented in Table 3.4.

Table 3.4 Use in EU of NP as an intermediate (tonnes/year)

Use	**2009**	**1997[1]**	**1997[2]**
Industrial production: Intermediates			
Intermediate - NPE	Confidential	No data	47,000 [3,4]
Intermediate - Phenolic oximes		2,500	2,500
TNPP Production		4,000	4,000
Plastic Stabilizer Production		No data	1,000
Phenol/formaldehyde resin production		22,500	22,500
Epoxy resin Production		1,500	1,500
Production of other organic basic chemicals		7,000	No data
Sub-total (intermediate, plastic, resins and stabilisers)		**35,000**	**29,000**
Industrial production: Materials			
Monomers in Polymers	Confidential	No data	No data
Emulsion polymerization		3,600[5]	No data

Notes:
1) Andersson et al. (2010)
2) EU RAR (2002)
3) 118,000 tonnes/year as NPE (Andersson et al. 2010)
4) 31,000 was used in EU the rest was exported.
5) 9,000 tonnes as NPE

3.3.2 Use in formulation

NP and NP derivatives are also used in the formulation of certain products. These applications are assumed to be of relatively minor importance. Such applications include:

- Textile and leather auxiliaries;
- Additives in concrete;
- Additive in plastics, food packaging included;
- Additive in photographic chemicals, and
- Component in laboratory chemicals.

Table 3.5 presents the use of NP/NPE in chemical formulations and articles.

Table 3.5 Use in the EU of NP/NPE in chemical formulations

Use	2009	1997[1]	1997[2]
Industrial production: Products - Articles			
Formulation of paints, laquers and varnishes	Confidential	1,600	3,997 (NPE)
Formulation of adhesives		9,000	No data

Notes:
1) Andersson et al. (2010).
2) EU RAR (2002).

3.3.3 Consumer uses

According to Månsson et al. (2008), NP/NPE are still found in a wide variety of products. Apart from textiles, the largest volumes originate from paints/lacquers, adhesives and cleaning agents. Other studies indicate that tyres are a significant source of both NP and its derivatives (KEMI, 2006). Floor coverings have also been found to be a significant source due to the addition of epoxy resins for accelerating the hardening process (Kjølholt et al., 2007).

The most important source of NP and NP derivatives seems to be:

- Coatings;
- Paints;
- Inks;

- Adhesives, and
- Tyres and other rubber products.

Table 3.6 End uses of products containing NP/NPE

Product	2007	1997[1]
Paints, coatings and inks	Confidential	4,000
Adhesives		No data
Tyres and rubber products		No data
Cleaning agents for professional use	No update	23,000
Agriculture products	No update	5,000

Notes:

1) Postleet al. (2003)

3.3.4 Specific uses of NPEs

NPEs are surface active agents (surfactants) that are part of the broader category of surfactants known as alkyphenol ethoxylates (APEs). Surfactants can be categorised according to their electric charge in water. Categories of surfactants include anionic (negative), non-ionic (no charge), cationic (positive), and amphoteric (positive and negative). NPEs are part of the non-ionic category.

Non-ionic surfactants, including NPEs, are used in a wide variety of applications including detergents, cleaners, degreasers, dry cleaning aids, petroleum dispersants, emulsifiers, wetting agents, adhesives, agrochemicals, including indoor pesticides, cosmetics, paper and textile processing formulations, pre-wash spotters, metalworking fluids, oilfield chemicals, paints and coatings, and dust control agents. The wetting properties of NPE surfactants are of particular importance for degreasing (i.e. cleaning), where the surface tension of the cleaning solution has to be low enough in order to wet the entire surface of the material to be degreased (RPA, 1999). In certain applications, NPEs are also used for the other properties they confer (COHIBA, 2012a). According to the US EPA DfE (2011), NPEs are considered to be “workhorse” surfactants given their cost-effectiveness and high performance in multiple applications.

In general, short-chained NPEs (3 - 20 EO units) are used as detergents and other cleaning productions (Ahel et al., 1994). NPEs with chains of medium length (10 - 30 EO units) are used as emulsifiers. Long-chained NPEs (with up to 80 EO units) can be used as dispersants, owing to their ability to retain small particles in solutions.

It was estimated that NPEs represented between 80-90% of the APEs used in the EU in the 1990s (by tonnage) (Postle et al., 2003; RPA, 1999), with the corresponding market share for OPEs estimated to be 10-15%. The EU RAR (2002) provides a summary of the end-use of NPE by industrial sector in 1997 (see Table 3.7).

Table 3.7 Estimates of uses of NPEs in Western Europe in 1997

Column Heading	NPE usage (kt)	% of NPE usage
Industrial & institutional & domestic cleaning	23	29.6
Other niche markets & miscellaneous	12.6	16.2
Emulsion polymerisation	9	11.6
Textile auxiliaries	8	10.3
Captive use	7	9.0
Leather auxiliaries	6	7.7
Agriculture	5	6.4
Paints	4	5.2
Metal industry	2	2.6
Pulp & paper	1	1.3
Total	77.6	100

Source: EU RAR (2002)

As described previously, it is estimated that production of APEs was approximately 32,000t in Europe in 2010 (CEPAD, 2010), with NPEs continuing to account for approximately 80% (26,000 tonnes). Approximately 8,000t is used by manufacturers “captively” e.g. to produce further derivatives. The remaining 18,000t is sold on to other companies in the enlarged EU. No specific information is available on the uses of the 18,000t APEs (CEPAD, 2010).

3.4 Alternatives to NPEs

Over the past decade, significant efforts have been made to identify and characterise alternative surfactants to NPE. The US EPA DfE (2011) identified eight potential alternatives to NPE (see Table 3.8).

According to the textile industry, APEs, alcohol ethoxylates and other ethoxylates are mostly used as alternatives to NPEs (OSPAR, 2009). However, octylphenol ethoxylates (OPE) degrade to form octylphenol which is very toxic to aquatic organisms (exceeding the TGD criterion for toxicity). OP is not easily degraded in the environment and meets the TGD criterion for persistence. OP also has the potential to cause endocrine disruption effects. As a result, OPE are not considered in this study as viable alternatives to NPE, as substitution with OPE would not be expected to reduce risk to the aquatic environment (which is borne out by the consultation and data review undertaken for this study). In addition, neither the cost (much higher than NP) of octylphenol, nor its performance and availability makes it suitable as a substitute for NP (OSPAR, 2009).

By far the most common replacements for NPE are alcohol ethoxylates (AEs) (ToxEcology, 2002; HERA, 2009; AIST, 2009). AEs are a very widely used class of non-ionic surfactants. They have been used in significant quantities in industrial products since the 1930s. Significant quantities of AE are converted to alcohol ethoxysulphates (AES) with the remaining AE used primarily in household laundry detergents. AE have many desirable characteristics such as rapid biodegradation, low to moderate foaming ability, superior cleaning of man-made fibres and tolerance of water hardness (HERA, 2009).

More than 435,000 tonnes of AE were produced in North America and Western Europe in 2000 (Modler et al, 2002). Consumption of AEs in Western Europe in 2000 was estimated at 645,000 tonnes (ToxEcology, 2002). No further information was found on current levels of production and consumption in Europe.

In the past there have been concerns about the availability of detergent-grade alcohols (raw material for AE) which could be a significant constraint to increased use of AEs as replacements for NPE (ToxEcology, 2002). Detergent grade alcohols are alcohols containing twelve or more carbon atoms per molecule and having a carbon backbone with a high degree of linearity. These types of alcohols are employed mainly in detergent production but they also have a number of other diverse applications. It is unclear whether supply remains a problem.

Further information is provided on the alternatives in Appendix D.

Table 3.8 Environmental hazard summary for available surfactants (USA EPA DfE, 2011)

Chemical	Fate		Aquatic toxicity			Synthesis
	Persistence	Degradates of concern	Acute	Chronic	Degradate aquatic toxicity	
Octylphenol ethoxylates (OPEs)	H	Y	H	H	VH	Octylphenol is prepared from phenol and diisobutylene, yielding a highly branched, predominantly para-substituted alkylphenol. Reaction of octylphenol with ethylene oxide yields OPE surfactants.
Linear alcohol ethoxylates (LAE)	VL	N	VH	H	L	Linear alcohols, derived from fatty acids or alpha-olefins, are reacted with ethylene oxide to yield LAE surfactants. Many detergent grade LAEs make use of alcohols in the C10-C18 range.
Ethoxylated/propoxylated alcohols	L	N	M	M	L	2-Ethylhexanol is reacted with ethylene oxide and propylene oxide to yield this product. Other surfactants in this class use linear alcohols in place of 2-ethylhexanol.
Alkyl polyglucose (APG)	VL	N	M	M	L	Fatty alcohols are reacted with glucose in the presence of an acid catalyst. Similar products may be prepared from other sugars, such as sucrose.
Linear alkylbenzene sulfonates (LAS)	VL	N	H	H	L	Benzene is alkylated with a linear olefin (either internal or terminal) in the presence of an acid catalyst, yielding a linear alkyl benzene (LAB). The LAB intermediate is sulfonated and neutralized to yield a linear alkyl benzene sulfonate surfactant

Table 3.8 (continued) Environmental hazard summary for available surfactants (USA EPA DfE, 2011)

Chemical	Fate		Aquatic toxicity			Synthesis
	Persistence	Degradates of concern	Acute	Chronic	Degradate aquatic toxicity	
Alkyl sulphate esters (AS)	VL	N	H	H	L	Fatty alcohols are sulfated and neutralized to yield alkyl sulphate ester salts.
Alkyl ether sulphates	L	N	H	H	L	Linear alcohol ethoxylates are sulfated and neutralized to yield alkyl ether sulfate salts.
Sorbitan esters	L	N	H	H	L	Fatty acid methyl esters are reacted with sorbitan in the presence of a basic catalyst to yield sorbitan esters.

VL = Very low; L = Low; M = Medium; H = High; VH = Very high.

3.5 Trends

Companies responding to the questionnaire indicated that regulatory pressures have impacted demand for NP and NPEs and that this trend is expected to continue in the future.

The data presented in 3.2.2 suggests that the total production volume of NP, as well as number of producers, has decreased in Europe over the past two decades. Similarly, the total production volume of NPE has decreased (by around 70% from 118,000t in 1997 to around 32,000t today).

There has also been a significant decrease in the use of NP and NPEs in Europe especially in the sectors posing high emission factors to water (i.e. where a high percentage of used NP enters the waste water). This is due to the implementation of legislation in Europe over recent years at both a regional and national scale following concerns over risks posed by NP/NPEs to both human health and the environment.

A number of Member States implemented voluntary agreements with industry not to use NP or NPEs. For example, in the UK industry agreed on a voluntary action to phase out the use of NPEs in domestic cleaning products in 1976. In addition, in OSPARCOM, with membership covering many countries, a recommendation to ban the use of APE in household detergents by 1995 and in industrial detergents by 2000 was adopted (AIST, 2009).

European-wide legislation relevant to the emissions of NPs includes the Water Framework Directive (Directive 2000/60/EC). NP is identified as one of the “priority hazardous substances” under which its emissions, discharges and losses will be ceased or phased out by the year 2021.

NP and NPE were included on the first list of chemicals for priority action towards achieving the OSPAR Convention target of ending discharges, emissions and losses of all hazardous substances to the marine environment of the north-east Atlantic by 2020 (OSPAR, 1998).

Furthermore, the use of NP and NPE is restricted for a number of uses within the EU since 1/1/2005 due to Directive 2003/53/EC (REACH Regulation, Annex XVII (46)). This requires that NP/NPE shall not be placed on the market or used as a substance or constituent of mixtures in concentrations equal or higher than 0.1% by mass for the following applications:

- Industrial and institutional cleaning except controlled closed dry cleaning systems where the washing liquid is recycled or incinerated, cleaning systems with special treatment where the washing liquid is recycled or incinerated;
- Domestic cleaning:
- Textiles and leather processing except processing with no release into waste water, systems with special treatment where the process water is pre-treated to remove the organic fraction completely prior to biological waste water treatment (degreasing of sheepskin);
- Emulsifier in agricultural teat dips:
- Metal working except uses in controlled closed systems where the washing liquid is recycled or incinerated;
- Manufacturing of pulp and paper;
- Cosmetic products;
- Other personal care products except: spermicides;
- Co-formulants in pesticide and biocides (subject to Article 1.2).

Therefore, the use of NPE for textile processing has been restricted in the EU since January 2005. Whilst there are some derogations and residues of NPE continue to be present in some EU-produced textiles, the volume of NPEs used by EU textile mills in processing has decreased since the introduction of this restriction (from 8,000t to 5,000t per year). However, NPE is still widely used in large concentrations in industrial processes and in products manufactured in countries outside the EU (COHIBA, 2011a). NPs and NPEs can therefore enter the EU market since it is not prohibited to import products containing NP and NPEs. This is discussed in further detail in the following section.

4. Data Analysis

4.1 Uses and Releases

4.1.1 NPEs in the textile industry

NPEs have been used historically in several processes of textile manufacture, including: scouring, fibre lubrication and dye levelling. Table 4.1 summarises in which specific textile sub-sectors NPE may be used.

The main use has historically been in wool scouring where natural oils are removed from the wool. NPEs are used because of their detergent and fibre lubricating (conditioning) properties and because they are not adsorbed in to the wool (EU RAR, 2002).

To a lesser extent, NPEs are also used in some textile dyeing and finishing processes. For example, they are used as emulsifying agents in certain textile treatments, in order to get the treatments in the right phase to interact with the textile (RPA, 1999). It seems that NPE may be present in both fabric (Seppänen, 2010) and printing sections of textiles (Pedersen and Hartmann, 2004).

Table 4.1 NPEs in the textile sector

Sub-sector	Where NPE may be found in process
Fibre, yarn thread mills	Mercerizing, bleaching, dyeing
Fabric mills	Woven mills – de-sizing, scour and wash, mercerize and wash, bleach and wash, printing, dyeing and washing, finishing and drying (in the same establishment) Non-woven mills – dyeing, chemicals finishing (in the same establishment) Knit mills – scour, bleach, dyeing , printing, wet finishing (in the same establishment) Wool mills – bleach and rinse, light scour, dye, carbonize, piece dyeing, chemical finishing (in the same establishment).
Textile and fabric finishing and fabric coating	Woven mills – finishing and drying Non-woven mills – chemical finishing Knit mills – wet finishing Wool mills – chemical finishing

Source: XCG (2006)

It has been estimated that approximately 8,000t per year of NPEs were used in the textile industry in Europe in 1999 (see Table 3.7), with most of this being used for wool scouring (RPA, 1999). Despite the restrictions (since 2005) on the use of NP and NPE in textile industries in the EU due to Directive 2003/53/EC, their use is still permitted in closed systems with no releases to waste water.

Consultation with industry has indicated that a number of companies in Europe continue to use NPEs in the manufacture of certain textiles. In Germany, a small number of companies are known to use NPE in polymeric dispersions for coating technical textiles[2] (for use in, for example, tents). The NPE acts as a glue to bind the coating. The coatings are applied to the surface of textiles to confer particular technical functions to the textiles[3]. NPEs remain in a residue on the surface of the textile. It is understood that these technical textiles are not intended to be washed in a washing machine. The Anglo-Welsh Environment Agency also indicated that a number of companies in the UK use NPEs in the same application.

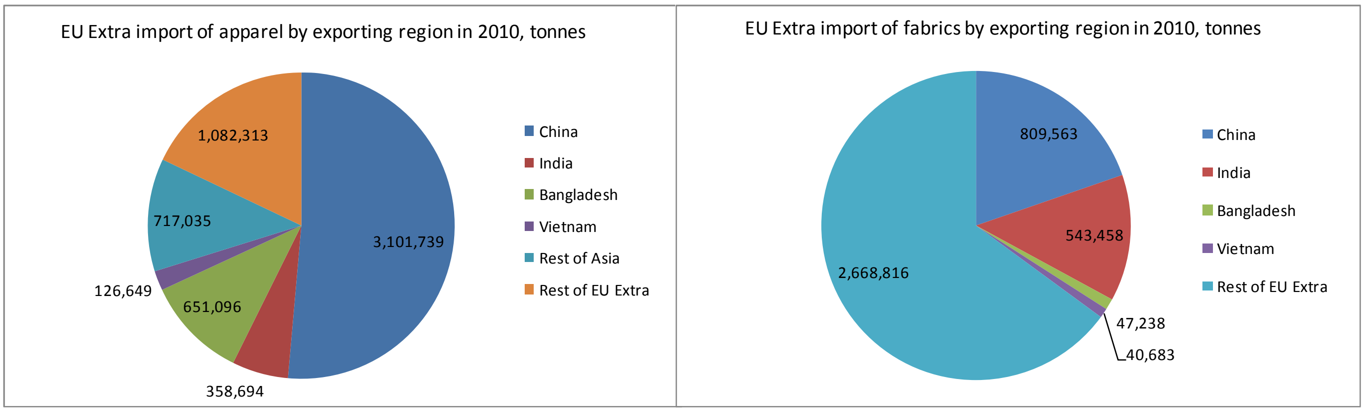
Available literature also suggests that NPEs continue to be used in manufacture of textiles in Estonia (COHIBA, 2012b). No further information was available on this.

The German textile association, TEGEWA, estimates that up to 5,000t of NPEs may be used in Europe for such uses.

In general, however, manufacturing of textiles in the EU has decreased over the years largely due to cheaper manufacturing costs in other countries. Many textiles and clothing articles are imported to the EU from non-European countries, such as China, India, Vietnam and Bangladesh (see Figure 4.1). These countries continue to use NPE in their textile processing as a detergent after dying as it is inexpensive and highly effective[4]. Typically some of the substance remains as a residue on the textile, however NPEs with a chain length greater than six are readily soluble in water and therefore the residue is easily washed out (CEPA, 1999).

Thus, emissions of NPE (and, thereby, NP) are possible, for instance, via the washing of textiles and via leaching from landfills that contain textiles and other articles containing NPE that originated from outside Europe. The emissions from washing of textiles imported from outside the EU are estimated to be much more significant than emissions from landfills (COHIBA, 2012b).

Figure 4.1 EU Extra import of apparel and fabrics in 2010 by exporting country, tonnes



Source: Eurostat data

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[2] Personal communication with Textil-Bekleidung, March 2012.
[3] Personal communication with Huntsman Textile Effects, March 2012.
[4] Personal communication with Textil-Bekleidung, March 2012.

Note: ‘Apparel’ refers to finished articles of clothing ready for wear. ‘Fabrics’ refer to material that may still need to be finished (dyeing, printing, coating, finishing) and made-up (arranged into article) for consumer use.

4.1.2 NPEs in imported goods

The presence of NPEs in a product indicates that it was used during the manufacture of the product. However, the level of NPEs in the articles is not necessarily indicative of the amount of NPEs used during manufacture (Greenpeace, 2011). It is possible that NPEs are washed out from materials during manufacture (e.g. in the manufacture of jeans that are ‘stonewashed’ prior to finishing), resulting in a low level of NPEs in the final product. Therefore, a finished article found to contain a low level of NPEs could have been manufactured using far more NPEs than a finished article that was found to contain a higher level. The OECD (2004) emission scenario document estimates that, in general, 20kg of surfactants are used to treat 1 tonne of textile.

Numerous studies have sought to assess the level of NPEs contained in imported textiles (see Table 4.2). However, it is difficult to draw any conclusion concerning which types of textiles contain particularly high concentrations of NPE. The reviewed studies indicate a wide range of NPE content both concerning garment type and fibre type (from just above the limit of detection to 27,000 mg/kg with a mean value of 202 mg/kg in the 8 studies). This is further supported by communication with industry that suggests that NPEs are so ubiquitous in textile processing outside of Europe that it is virtually impossible to attribute NPEs contained in a textile article to a particular source or processing stage.

Table 4.2 Summary of reviewed studies

Study	Year of analysis	Number of samples	Number of samples under detection limit	Range (mg NPE/kg textile)	Mean value (mg NPE/kg textile)	Median value (mg/kg)
Greenpeace	2011	78	26	Nd (1) - 27,000	435	6
Swedish Television	2009	6	2	Nd (7) - 2,200	456	59
Testfakta/Children's winter overalls	2007	13	0	2-1,200	421	420
Prevodnik, A. et al. (2008)	2007	20	2	Nd (1.6) - 10,608	685	9
Prevodnik, A. et al. (2008)	2008	17	1	Nd (3) - 940	132	33
Klif, Norway	2011	22	3	<10 - 360	83	20
RådochRön	2012	97	60	<10 - 2,040	57	5
Confidential	2012	100	72	Nd (3) - 1,760	71	17
Total: 8 studies		**353**	**166**	**Nd - 27,000**	**202 mg/kg (96 excluding two highest values)**	**5 mg/kg**

Note: Older studies (e.g. pre-2007) have been excluded from the analysis.

A European textile testing company provided the following information based on extensive testing of textiles over a number of years[5]:

- In 30-40 % of the tested textiles no or low concentrations (<10 mg/kg) of NPE is present. At these low levels it is not likely there is an intentional use of NPE in the manufacturing process. This may result from traces of NPE in contaminated water in the manufacturing process or contamination by other fabrics during transport or storage;
- Half of the tested textiles, show NPE content from 10-20 up to 400-500 mg/kg (with an average of 100-150 mg/kg). These levels demonstrate, according to experts at Eurofins, a use of NPE in the manufacturing process; and.
- In about 5% of analysed samples 500 to 1000 mg/kg or more are found. An explanation for these high levels could be the use of colour pigment contaminated with a high concentration of NPE during the colouring process of fabrics.

These conclusions, based on industry experience and several years of testing a large range of garment types and fabrics, have been used to derive the abatement cost curves.[6] The conclusions provide a view on both the proportion of textiles containing NPE and concentration where present. Furthermore, as compared to the studies reported in the table above, the results are assumed to be more representative of the wider presence in textiles, because some of the reported figures in the table were intended specifically to highlight the presence of the substance, rather than to provide a representative estimate.

4.1.3 Environmental releases

There are two potential sources of EU emissions of NP and NPE arising from use in textiles:

1. Releases from EU-manufactured textiles containing NPE; and
2. Releases from imported textiles containing NPE.

4.1.3.1 Textiles produced in the EU

As set out previously, a certain quantity of NPE is used by manufacturers of technical textiles in the EU, with a proportion remaining on the finished article. Very limited information is available on the extent of this use or the quantity remaining on the textiles. For the purposes of development of the cost curves, the following has been assumed:

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[5] Communication with Eurofins, October 2011.

[6] In the cost curves, we have assumed that 50% of imported textiles have an average concentration of 125 mg/kg, 5% of imported textiles have an average concentration of 750 mg/kg and 45% have zero NPE present (see supplementary spreadsheet).

- A maximum of 5,000t of NPEs are used in manufacture of technical textiles each year (as set out above);
- Of the amount used, an average of 0.6% of the NPE used is assumed to remain on the textile[7] (i.e. 30t per year); and
- The technical textiles in question are not generally assumed to be washed in the same way as clothes, but may be wetted (e.g. through rain), leading to at least some release to the environment. It is therefore assumed that, over an average 10 year lifetime[8], 5% of the remaining NPE in the textile articles is released to surface water each year. The remainder is assumed to remain in the textiles during disposal.

Based on these assumptions, the releases to the environment (assumed to be surface water) have been calculated according to equation (5) in the OECD (2004) emission scenario document as: 12.0t per year.

4.1.3.2 Release estimates from imported textiles

The ongoing "Control of hazardous substances in the Baltic Sea region" (COHIBA)[9] study has estimated emissions from the washing of textiles imported from outside the EU. The COHIBA (2012a) study relied on three existing studies to extrapolate total emissions of NPE to wastewater from washing of imported textiles (see Table 4.3). The study concluded that the estimated yearly emission from washing of textiles is 225-525 tonnes NP_{equ} to wastewater in Europe.

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[7] For comparison purposes, the average concentration detected in textiles where NPE has been used is around 125 mg/kg (0.125 kg/t) based on data from Eurofins, referred to above, whilst the typical quantity of NPE used is assumed to be around 20 kg/t (the OECD (2004) emission scenario document suggests a figure of 20g surfactant per kg of textile). Although these figures are subject to uncertainty, they suggest that around 0.6% of the NPE used could remain on the textile after treatment.

[8] The OECD (2004) emission scenario document quotes 5-20 years for tents.

[9] The COHIBA Project identifies the sources and inputs of 11 hazardous substances to the Baltic Sea area, including NP/NPEs. http://www.cohiba-project.net/.

Table 4.3 Emissions from washing of textiles containing NPEs (COHIBA, 2012a)

Scenario	Detail	Source
Scenario I	In Stockholm the estimate emission of NP to wastewater from washing of textiles was 1200 kg/year in 2007. The release was calculated assuming that NP is released every time a household washes imported textiles and gives an emission factor of 1.5g/inhabitant and year to wastewater.	Prevodnik et al., 2008
Scenario II	Estimated emission of NP and NPE to wastewater from washing of imported textiles was 700-1600 kg NP equivalents/year in Stockholm (795,000 inhabitants), giving emission factor range of 0.9 – 2.1g/inhabitant. Estimate based on the assumption that all textiles could have a content of NPE with an average concentration of 514mg/kg textile and NPE is released every time a household washes imported textiles (emission factor 100% to waste water).	Manssön et al., 2008
Scenario III	50% of targeted t-shirts contained 1.7-271 mg NPE/kg textile (fabric) with average of 64 mg/kg and median 17 mg/kg). The estimation is based on the assumption that 50% of imported textiles could have a content of NPE (Seppänen 2010) with an average concentration of 514 mg/kg textile and NPE is released every time a household washes imported textiles containing NPE (emission factor 100% to waste water, Pettersson & Holmström 2011b). The applied range of emission factor was 0.45-1.05 g/inhabitant and year.	Seppänen, 2010

How much NPE is released during the washing process may differ between different types of textiles and NPE content in the fabric when purchased. However, no conclusions can currently be drawn on what kind of fabric releases the most NPE when washed. The studies referenced in Table 4.3 appear to have adopted a linear release model. However, simulated laundry studies with cotton suggest that 99.9% of NPE is washed out after two washes[10]. As NPE is a non-ionic surfactant it is easily dissolved in water. Therefore, it is likely that most NPE will be washed out after repeated washing, regardless of the type of textile (Månsson et al. 2008). There may be articles or products that are not washed as often or maybe ever, however these are assumed to represent a minority of textiles.

For the purposes of this study, import data from Comext[11] has been used to estimate total emissions from washing (see Table 4.4). See Appendix C for more detailed data.

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[10] Consultation with Anglo-Welsh Environment Agency, March 2012. Study has not been published to date.

[11] Comext is the Eurostat reference database for external trade. It contains both recent and historical data from the European Union Member States and a significant number of third countries covering the value of exports and imports of products.

Table 4.4 Total NPE emissions from washing of imported textiles, 2010

Products	Quantity	Note
Imported fabrics (CN Codes 50-59) ('semi-finished'), tonnes	4,109,750	Comext data, 2010. In practice, not all of these semi-finished fabrics will necessarily be subject to washing in either further finishing processes or during usage (for example, CODE 59041000 Linoleum) and therefore may not contribute to NPE release. However, there is insufficient information to determine what proportion of fabrics will be washed. There is a reasonable expectation that some of the imported fabrics products may contain NPEs and would be washed during additional textile processing or product lifetime which would therefore lead to releases (e.g. CODE 51052900 Wool combed). Therefore, 4.1 million tonnes is taken as an upper estimate.
Imported apparel (CN Codes 60-65) ('finished'), tonnes	6,037,000	Comext data, 2010
Total imports, tonnes	**10,146,750**	
Concentration of NPEs	50% at 125 mg/kg 5% at 750 mg/kg	Assumed values based on consultation
Emission factor to wastewater (WW)	0.85	EU RAR (2002). Note that some releases will be direct to surface water (see below)
Total emissions of NPE from washing of imported textiles, tonnes	**862.5 (345t NP$_{eq}$)**	This represents an upper estimate as in practice not all of the semi-finished textiles will be washed. As an example, if 50% of the semi-finished textiles are assumed to be washed subsequently this would result in total emissions of 687t NPE (275 NP$_{eq}$).

Note:

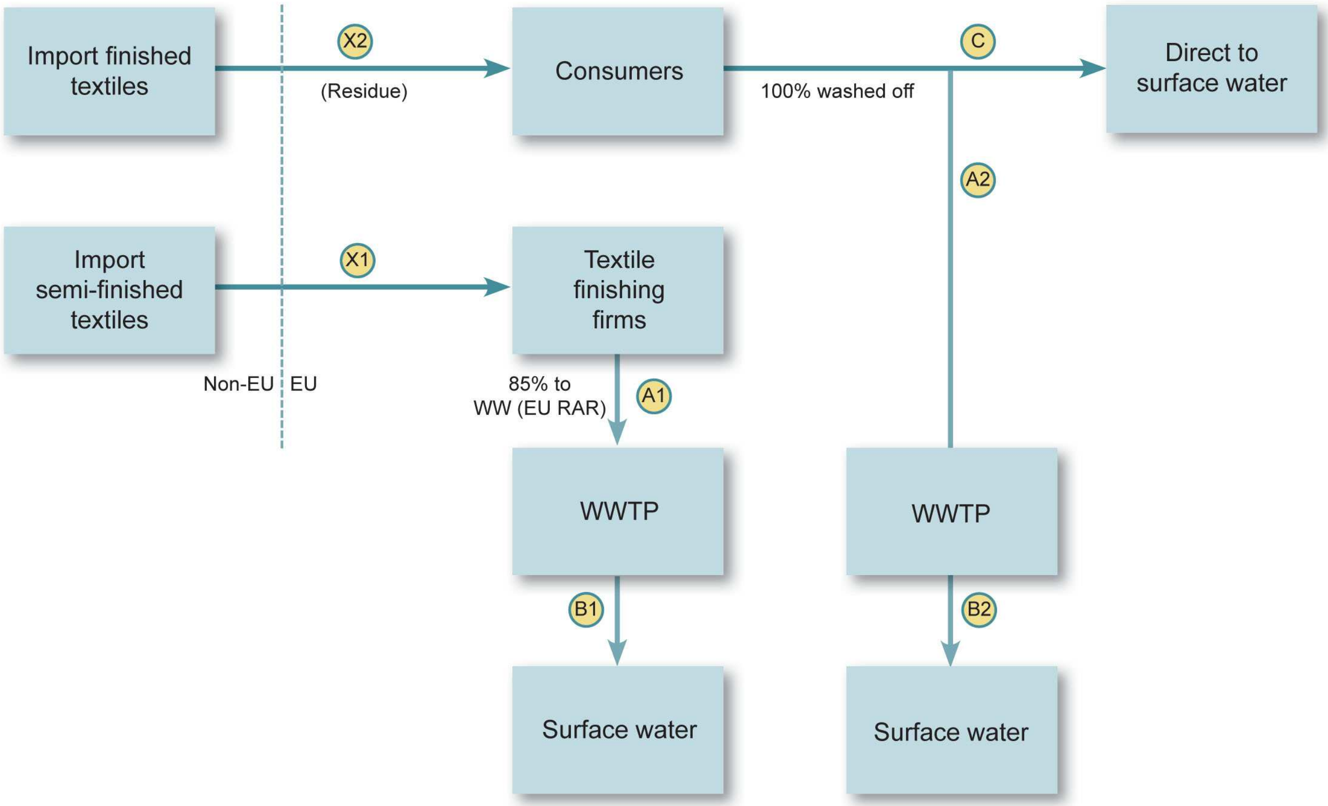
'Imported fabrics' refers to fabrics that are imported that will undergo further processing to produce finished goods (e.g. semi-finished). 'Imported apparel' refers to finishing article of clothing ready to be placed on the market (e.g. finished).

NP$_{eq}$ values have been calculated assuming an NP:NPE ratio of 2:5.

In total, therefore, approximately 860t of NPEs are estimated to be washed out of imported fabrics and clothing apparel each year. Assuming NPE with 8 EO units (where the NP:NPE ratio is 2:5) this is estimated as 345t NP$_{eq}$ annually.

The figure below provides a schematic of the main routes of release of NPEs to water. This has been used as the basis of estimating quantities of NPEs released to the environment, as an input to the abatement cost curve calculations. This only concerns textiles imported to the EU (releases from EU-produced textiles were considered in the previous section).

Figure 4.2 Flow diagram of NPEs from imported textiles



Two sources of releases are considered in turn: releases from imported semi-finished textiles and releases from imported finished textiles.

In developing the abatement cost curves, all releases are treated as NPE. It is recognised that there will be some degradation to other substances (notably NP) but these are not considered separately in the cost curves.

The abatement measures that can be applied to address releases of NPEs from textiles can apply at different stages of the supply chain. A restriction on imported textiles will eliminate both the quantity present in textiles and the quantity released from WWTP to waste water, whereas measures to abate releases from WWTP will reduce (but not eliminate) those releases but not directly affect the quantity present in imported textiles.

Semi-finished textiles

Where semi-finished textiles are imported into the EU (4.11 million tonnes from Table 4.4), it is assumed that these will be treated and most likely washed by EU-based textile firms, during the process of being turned into finished apparel. The quantity of NPE present in these textiles (X1 in the figure above) is assumed to be fully removed from the textiles during this process.

As was assumed in the EU risk assessment (European Commission, 1999), the emission factor to wastewater from the textile processing industry (A1) is assumed to be 0.85, meaning that 85% of the NPEs present in imported semi-finished textiles enter the WWTP. As set out in the OECD's emission scenario document on the textile finishing industry (OECD, 2004), the textile finishing industry is characterised by many small and medium-sized enterprises which discharge predominantly to municipal wastewater plants. As a practical working assumption, it is assumed that all such companies discharge to municipal wastewater treatment plant.

Of the quantity of NPEs entering the WWTP, the EU RAR (2002) provides the following worst-case assumption during anaerobic wastewater treatment would be (based on % weight):

- 19.5% removed as nonylphenol in the sludge;
- 45% mineralised or highly degraded; and
- 35.5% released in effluent.

The quantity released as effluent is all treated as NPE, whereas in practice, the effluent will contain partially degraded products, including NPs[12]. Again, this is a practical working assumption for the development of abatement cost curves.

Finished textiles

For finished textiles, it is assumed that textile apparel is supplied to consumers without any further treatment at industrial installations in the EU. The releases of NPEs are therefore assumed to derive from washing of the textiles at a large number of diffuse sources (i.e. households).

Of the amount of NPE present in imported finished textiles (X2), 100% is assumed to be washed off to wastewater within households (or public launderettes) within a few washes, certainly within the first year of use. Not all households in the EU are connected to WWTPs and so releases to the environment are divided as follows:

- Releases direct to surface water (C); and
- Releases to municipal WWTP (A2) of which 35.5% is assumed to be released from the WWTP (B2) as was the case with textile finishing companies that discharge to municipal WWTP.

The percentage of households that are connected to municipal WWTP is estimated to be 78% based on Eurostat data of population connected to UWWT plants and demographic data.

Summary of estimated releases for development of cost curves

The table below summarises the assumed releases of NPEs from each of the main sources of textiles of interest for the current analysis, based on the discussion above. It should be noted that these releases have been derived solely

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[12] Of the 35.5% released in effluent, 25% was assumed to be released as NP1EO/NP2EO/NPnEC, 8% as NPnEO (n>3) and 2.5% as nonylphenol in effluent 2.5%.

for the purposes of development of abatement cost curves within the current project and should not be interpreted as being official estimates for use in risk assessments or other analyses.

As shown in the table below, in total 750t NPE are estimated to be released to WWTPs per year. In NP_{eq} this is calculated as 300t per year (based on NPE with 8 EO units where NP/NPE ratio is 2:5). This is consistent with the range of 225-525t NP_{eq} per year emitted to WWTPs estimated by COHIBA (2012a).

Table 4.5 Summary of NPE releases to the environment used in cost calculations

Emission source	Releases to WWTP under baseline, t per year	Release to surface water under baseline, t per year	Direct to surface water, t per year	Total to surface water, t per year
Imported semi-finished textiles	400	140	113	253
Imported finished textiles	349	122		122
Total imported	750	262	113	375
Technical textiles produced in the EU	Not applicable (no wastewater releases)		12	12
Overall total	**750**	**262**	**125**	**387**

4.2 Current and Planned Abatement Measures

Relevant measures (voluntary and mandatory) that are already in place are discussed in Section 3.5.

Furthermore, a number of apparel brands (Adidas Group, C&A, H&M, Li Ning, NIKE, Inc. and PUMA) have responded by published a joint agreement to achieve zero discharge of hazardous chemicals by 2020 across their supply chains[13]. As part of this, APEs/NPEs have been added to the priority list. It has been agreed that:

- "By the end of 2011, [the companies] will communicate to all suppliers the requirement to source preparations that are APE/NPE free. (Note: [The companies] believe conversion for detergents/scouring/degreasing could yield a reduction of up to 50% of APE/NPE in apparel and footwear supply chains.)
- In early 2012, [the companies] will initiate a project with chemical suppliers to identify a 'positive list' of APEO/NPE-free detergents.

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[13] "Joint Roadmap: Toward Zero Discharge of Hazardous Chemicals". Published November 18, 2011. See: http://www.roadmaptozero.com/

- In 2012, [the companies] will conduct a follow-up study at a selection of facilities that have converted to APEO/NPE-free detergents to evaluate if there are remaining sources including non-intentional uses etc. of these chemicals. (This study recognizes challenges including a lack of transparency into chemical ingredients and poor quality material safety data sheets (MSDS)).”

Consultation with the European Outdoor Group (EOG)[14] indicates that other large European clothing brands are also making efforts to limit the presence of NPEs in their products. These companies have added NPEs to their restricted substance lists (RSLs) and requested upstream suppliers to provide evidence that they do not use NPEs in production.

However, it appears that the companies stipulate varying restriction limits in their RSLs. For instance, Adidas requires that the sum of NP, OP, NPEO should be less than 1000 ppm[15] whereas most of the companies on the EOG panel have a limit of 100 mg/kg (100 ppm) in line with the target value defined by Greenpeace (2011).

Furthermore, it is of note that Sweden is currently considering the preparation of a restriction dossier in accordance with Annex XV of REACH.

4.3 Possible Future Abatement Measures

4.3.1 Range of possible measures available

Based on the data presented above, it is evident that the major historical use of NPE in textiles has been phased out in Europe. However, there remains some use in applications using closed processes by a limited number of European companies. However, NPE is still widely used in textile processing in countries outside the EU. The NPE content in imported articles is to some extent controlled today as a result of some importers’ and major brands’ own restriction of NPE in articles (e.g. Joint Roadmap, 2011), but the evidence indicates that the implemented measures are not sufficient (e.g. COHIBA, 2011a).

Based on the information reviewed above, a number of potential risk management options (RMOs) are possible:

1. A ban on all remaining use of NP/NPE in textiles in the EU;

2. A ban on the placing of the market of imported textile articles containing NP/NPE;

   This possible restriction could, for example, ban the placing on the market of any imported textile articles containing NP/NPE. In practice, it may not be feasible to have a total ban as there may be cross-contamination issues that are very difficult to avoid. For enforcement purposes, the restriction would probably need to contain a restriction limit so that the enforcement authorities can set up an

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[14] Specifically with the sustainability working group advisory panel (consisting of five brands: Fjäll Räven, Vaude, Gore, Salomon and VF).

[15] See: http://www.adidas-group.com/en/sustainability/assets/Guidelines/A01_Sept_2011.pdf

efficient supervision mechanism. A possible limit could, for example, be based on the current limit of detection, meaning that NPE should not be present in concentration higher or equal to 1 mg/kg. This is further discussed in Appendix D.

3. The implementation of end-of-pipe measures particularly at WWTPs.

Appendix D of this report provides details on the data sources and assumptions used in assessing the potential for each of the measures to reduce use of the substance and the associated costs.

4.3.2 Measures assessed

Table 4.3 provides a summary of the key data on each of the measures assessed. The table presents costs per emission to surface water (SW) and waste water (WW) avoided. Further details are included in the supplementary spreadsheet.

Figure 4.3 presents the cost curve, showing cumulative costs. Figure 4.4 presents the marginal costs.

Table 4.6 Summary of cost curve data

Measure	Label	Total annualised cost (€k)	Emission to WWTP avoided (t NPE)	Emission to SW avoided (t NPE)	Unit abatement cost (€k/t to WWTP)	Unit abatement cost (€k/t to SW)
Ban on EU use in textiles	BBAU1	942	N/A	12	N/A	79
Ban on presence in imported textiles	BBAU2	785,960	750	375	1,049	2,095
WWTP: AC filtration	BBAU3	12,026,403	N/A	111	N/A	108,730
WWTP: Membrane filtration	BBAU4	39,165,056	N/A	152	N/A	258,276
WWTP: Oxidative techniques	BBAU5	68,852,644	N/A	161	N/A	428,827

Notes: All data on costs and emission reductions are at an EU level. All costs have been calculated in line with ECHA Guidance on Compliance Costs[16].

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[16] Addendum to the Guidance on Socio-economic Analysis – Restrictions: Calculation of compliance costs. Available at http://echa.europa.eu/documents/10162/17087/appendix1-calculation__compliance_costs_case_restrictions_en.pdf.

Figure 4.3 Cumulative cost curve for NPEs in textiles (note secondary axis)

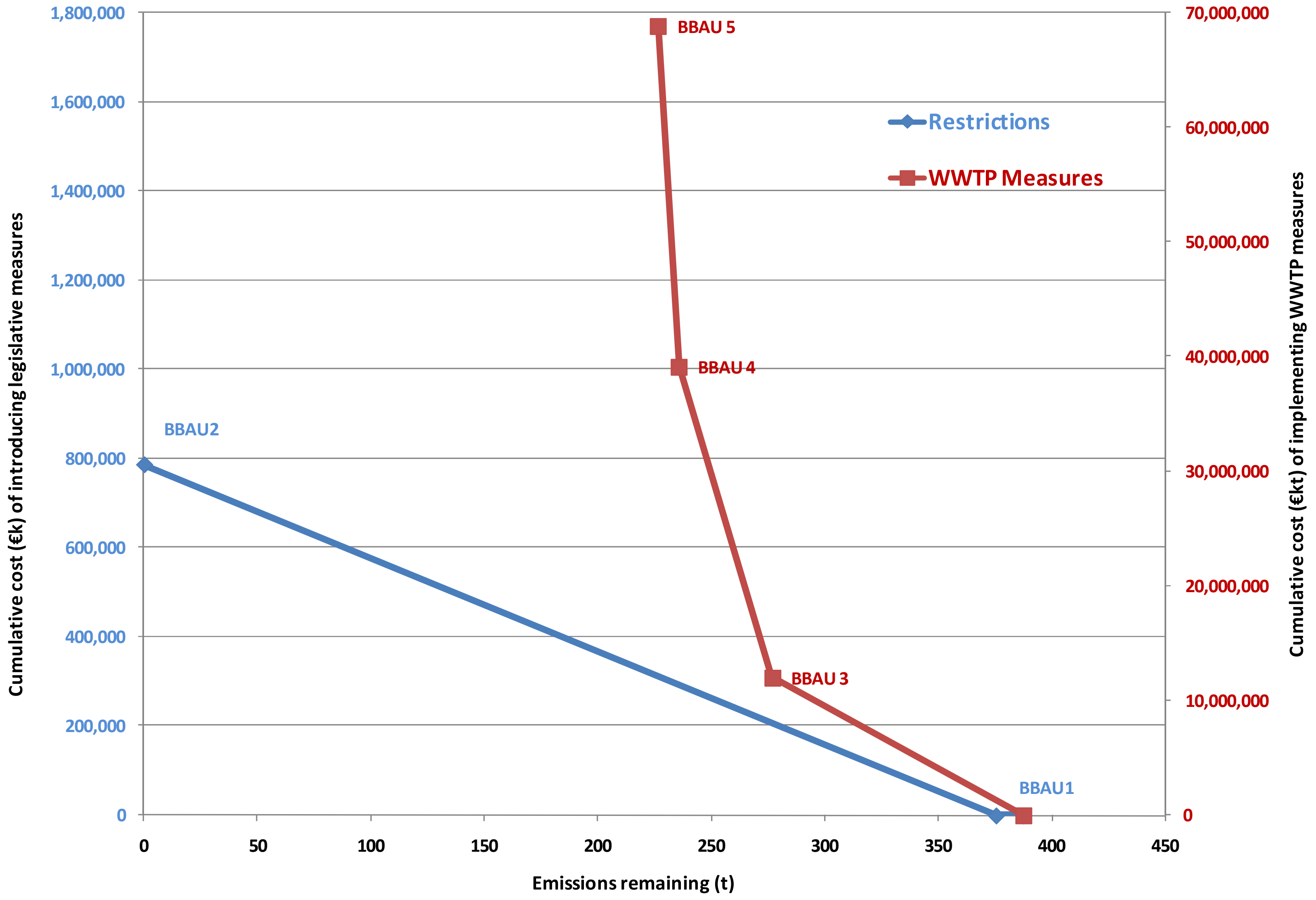
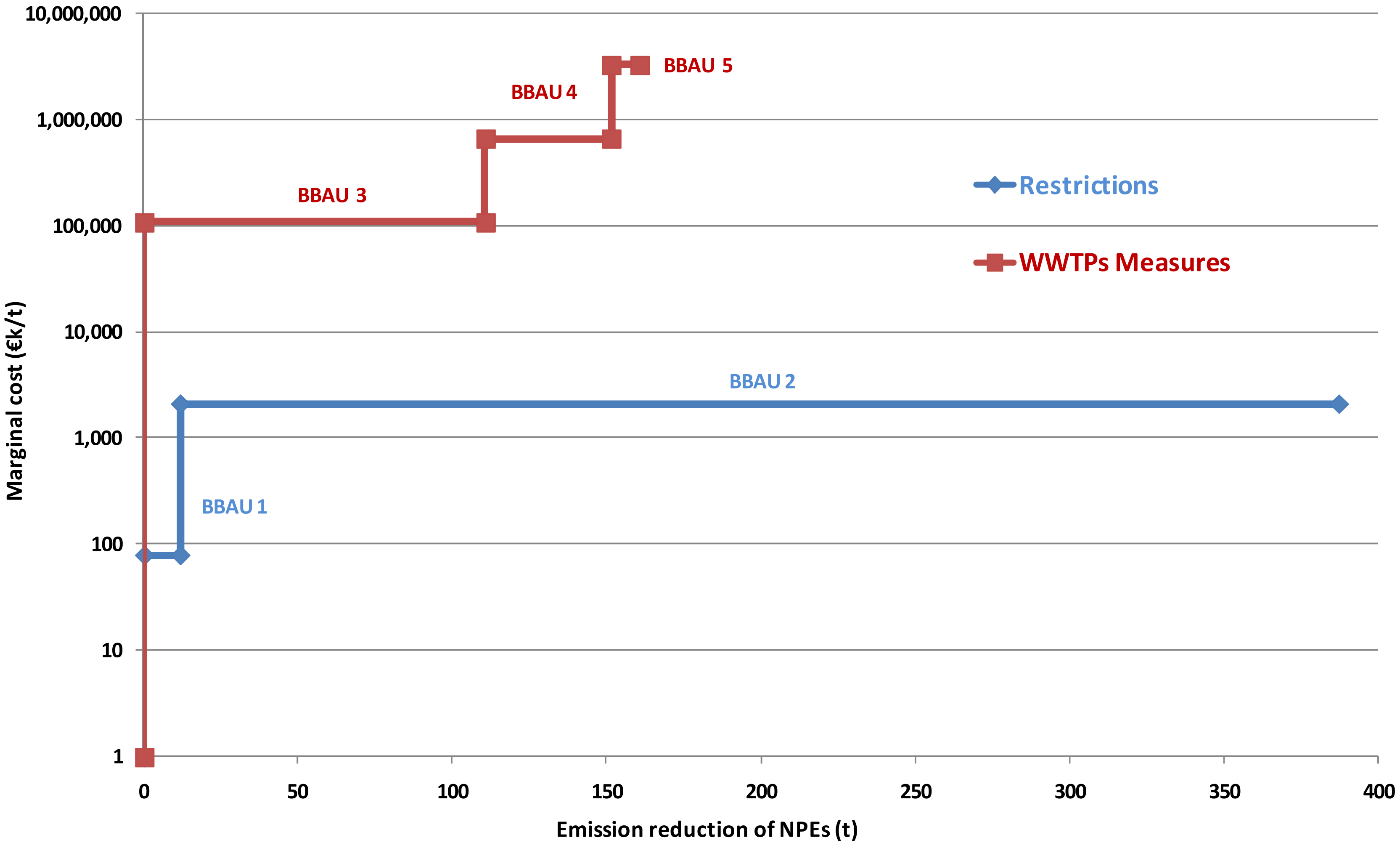


Figure 4.4 Marginal cost curve for NPEs in textiles (note logarithmic scale used)



5. Conclusions

The cost curves developed illustrate the relative costs, in €/t, of reducing emissions of NPE from textiles to surface water. The cost curves illustrate that either legislative measures, in the form of a restriction, could be introduced which would eliminate 100% of NPE emissions to surface water (estimated at close to 400t per year). Alternatively, tertiary WWTP measures, applied at a large number of WWTP across Europe, could be implemented which would eliminate around 42% (about 160t) of NPE emissions to surface water. Whilst the underlying assumptions are subject to a significant degree of uncertainty, the legislative measures (restrictions) are found to be far more cost-effective than the WWTP measures. Specifically, a ban on NPE in imported textiles (above a minimal threshold) is likely to be by far the most cost-effective way in which to eliminate the majority of emissions (97%). The total cumulative cost of possible restrictions is estimated to be around €800m, whilst to achieve the maximum reduction identified for WWTP, the total cost could be around €70bn. However, the costs for advanced WWTP measures do not take account of the co-benefits of removing other pollutants at the same time. These have not been quantified for the purposes of the current study but they are likely to be significant.

The costs of restrictions are based on several key assumptions and the calculated results are highly sensitive to these assumptions. Firstly, it is assumed that substitution costs to textile producers located outside the EU would be passed on to downstream European importers. It is not clear whether this would happen in practice as responses from consultation on this issue were mixed.

Secondly, the measures assume 100% replacement of NPEs with alcohol ethoxylates (AE). This is a simplistic analysis as, in practice, it is likely that a range of alternatives would be used, depending on the specific fabric and final function. Consultation with industry suggests that for companies using NPEs in polymeric dispersion for coating ‘technical textiles’ in Europe it may not be a simple case of substituting NPE with AE (although this represents only a small fraction of the total use, emissions and costs calculated in this report). More sophisticated cost curves would require further data collection and analysis in order to understand the likely uptake of different alternatives.

Thirdly, assumptions have been made regarding the likely response of industry to a proposed restriction on the placing of the market of imported textiles containing NP/NPE. It has been assumed that 50% of importers would conduct additional spot testing of textiles (the remainder are assumed to rely on communication from suppliers or to already undertake spot testing of NPEs in the baseline scenario). A testing frequency of 0.05% of textiles in the first year following the implementation of the restriction is assumed, with that frequency decreasing to 0.005% in subsequent years. These costs would in practice depend on the scope and definition of a restriction, as well as on enforcement requirements, such as whether companies would need to demonstrate compliance by testing or whether supply chain communication alone might be considered sufficient.

The assumptions regarding the potential to achieve reductions in emissions to surface water through tertiary WWTP measures are also subject to significant uncertainty. The underlying data on the amount of NPEs present in imported textiles (and the associated releases to the environment) are not based on extensive monitoring of imported textiles and the picture of current and potential future releases from existing WWTP are also based on relatively generic assumptions from existing literature sources.

What is clear, however, is that the potential to reduce emissions through a restriction is likely to be significantly greater than through additional measures at WWTP, and the costs are likely to be significantly lower. The drafting of any such restriction, and the activities associated with enforcement, will have a significant effect on how companies achieve and demonstrate compliance, and hence on the total costs.

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Appendix A
List of Organisations Consulted

The table below lists all the organisations that were contacted, indicating those that provided information that was used in this report, as well as those that provided a completed questionnaire.

Company	Information provided?
A.I.S.E.	
AEDT - European Association of Fashion Retailers	
AEPSAT	
Akzo Nobel (Sweden)	
BASF AG (Germany)	
British Apparel & Textile Confederation	✓
Centexbel	✓
CEPAD	✓
CESIO	
CIA (Chemicals Industries Association)	
CIRFS	
Clariant	
COHIBA Project	✓
Concordia Textiles (Waregem)	✓
Detic	
Dow Chemicals	
DyStar	✓
Eureau	✓
EURATEX	✓
Eurofins Scientific	✓
Fedustria (federation of Flemish textile companies)	✓
"Huntsman Textile Effects	✓
International Wool Textile Organisation (IWTO)	
LTMA, Lancashire Textiles Manufacturers Association	
M&S (ex-employee)	
Masureel Veredeling (Wevelgem)	✓
Oeke-Tex Association	✓
PCC Synteza SA, Kedzierezyn-Kozle (Poland)	

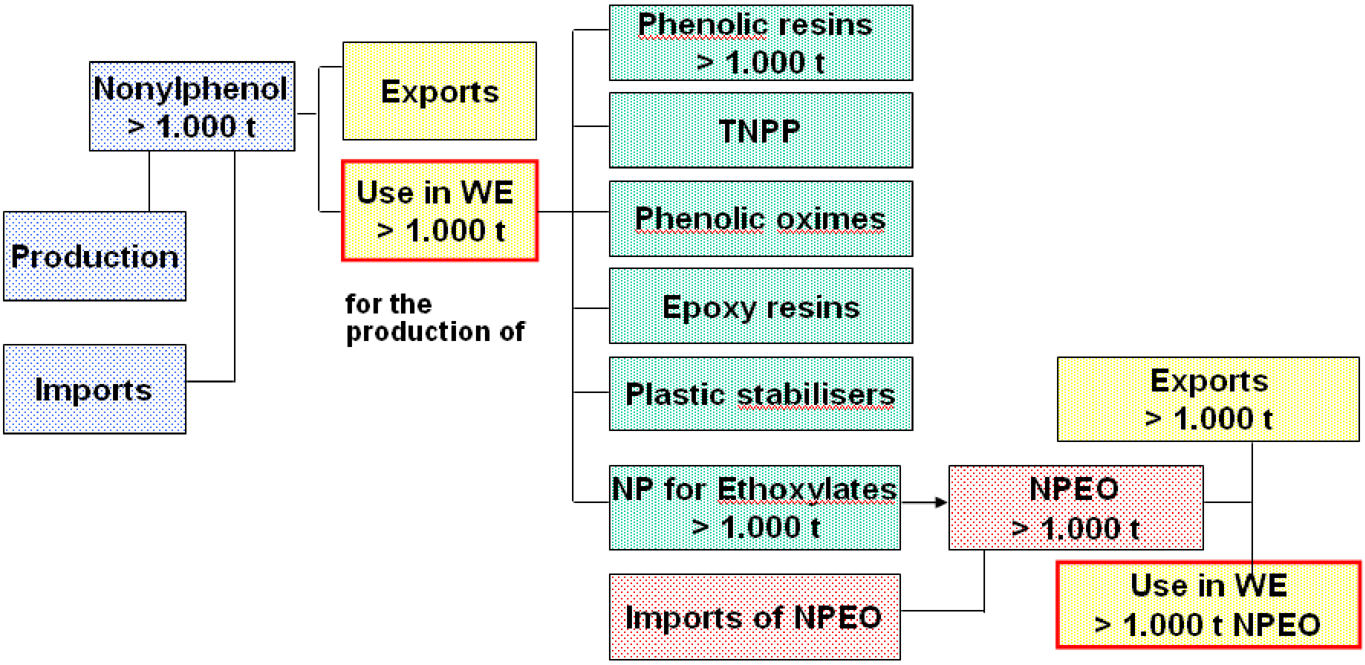
Company	Information provided?
Rhodia (part of Solvay Group)	
Sasol Germany/UK	✓*
Shirley Technologies	✓
SI Group Inc. (switz)	
Stepan UK	
TEGEWA	✓*
Testex	
Textilimportprerna	✓
Textile Institute	✓
Textil-Bekleidung	✓
Union des Industries Textiles de France	
Uniqema Uk (now Croda)	

Notes:

* Questionnaire completed and returned.

Appendix B
Flow Diagram for Nonylphenols Provided by Industry

Figure C.1 p-Nonylphenol: Mass Flow in Western Europe in 2010



Notes:

1) This figure was provided by CEPAD.

2) Western Europe includes Switzerland and Norway.

B2

Appendix C
EU Statistics on Import of Textiles and Clothing Articles

Comext provides import and export data on textiles by 4 and 8-digit CN (combined nomenclature) codes. The EU's import statistics are reported per exporting country. In the table below, China, India, Bangladesh and Vietnam are reported separately and the other countries are aggregated under "Rest of World".

Table D.1 Summary of EU's import of textiles and clothing articles, 2010, tonnes

Country of origin/CN Code	China	India	Bangladesh	Vietnam	Rest of World	TOTAL
Textiles						
5001 - 5007	6,144	1,195	-	174	550	8,063
5101 – 5113	34,351	8,394	2	1	164,910	173,306
5201 – 5212	94,071	157,752	2,938	737	677,553	838,980
5301 – 5313	20,306	72,736	42,030	526	100,885	216,177
5401 - 5408	235,427	52,555	30	2,196	498,017	552,797
5501 - 5516	142,844	116,063	229	27,853	729,770	873,915
5601 - 5609	81,947	7,287	181	823	206,749	215,040
5701 - 5705	70,980	117,729	1,673	323	156,269	275,993
5801 – 5811	40,270	3,462	100	426	29,985	33,973
5901 - 5911	83,224	6,287	56	7,625	104,128	118,096
Sub-total	**809,563**	**543,458**	**47,238**	**40,683**	**2,668,816**	**4,109,757**
Clothing articles/products						
6001 - 6006	107,285	2,257	35	1,817	106,624	218,018
6101 - 6117	1,091,595	135,249	437,637	30,711	687,897	2,383,089
6201 - 6217	1,126,468	106,232	167,182	56,681	496,224	1,952,787
6301 - 6304 / 6306 - 6309	677,894	114,654	45,722	22,323	493,981	1,354,574
64041100	36,907	129	517	14,122	9,046	60,721
95030041	61,590	172	3	995	5,576	68,337
Sub-total	**3,101,739**	**358,694**	**651,096**	**126,649**	**1,799,347**	**6,037,526**
Note: Full descriptions of CN codes can be found in the supplementary Excel File.						

Source: Comext

C2

Appendix D
Data for Cost Curves for NP/NPEs from Textiles

Characterisation of possible abatement measures

As discussed in Section 4.3.1, three potential risk management options (RMOs) have been identified:

1. A ban on the placing of the market of imported textiles containing NP/NPE;
2. A ban on all remaining use of NP/NPE in textiles in the EU;
3. The implementation of end-of-pipe measures particularly at WWTPs. Within this, three possible abatement measures have been identified.

RMO 1: Ban on the placing of the market of imported textiles containing NP/NPE

The Risk Reduction Strategy (RPA, 1999) considered a number of controls to reduce risks posed by NP/NPEs to both human health and the environment through potential amendments to existing legislation. Since then, the use of NP and NPEs has been restricted under Directive 2003/53/EC (REACH Regulation, Annex XVII (46)). NP/NPE shall not be placed on the market or used as a substance or constituent of mixtures in concentrations equal or higher than 0.1% by mass in a number of applications including textile processing. However, the major emission source of interest for the current study (textiles) originates from products and articles imported from outside EU, hence, reduction from this source depends on either restrictions for import of products containing NP/NPEs or on restrictions for use of NP/NPEs in such products and articles also in non-EU countries.

In this RMO, placing on the market of any textiles (fabrics and clothing apparel) would be banned if the NPE content is above a certain level. This could, for example, be based on the limit of detection (LoD) of current analytical testing methods for NPEs in textiles[17]. Eurofins states that the LoD for textile samples is 0.2 mg/kg[18], whilst Greenpeace (2011) notes a LoD of 1 mg/kg. However, there is currently no standardised testing method (e.g. ISO or EN standards) in place to determine NPEs in textiles[19]. Furthermore, specific substances would need to be defined.

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[17] Determination of APEOs involves extraction with organic solvent and quantification by liquid chromatography and mass spectrocscopy (see Wang et al., 2008).

[18] See http://www.eurofins.com/media/17648/apeo%20in%20textiles%20-%20en.pdf

[19] There are two standards available at the moment relating to alkyphenols:
1. EN ISO 18857-2:2011: Water quality - Determination of selected alkylphenols - Part 2: Gas chromatographic-mass spectrometric determination of alkylphenols, their ethoxylates and bisphenol A in non-filtered samples following solid-phase extraction and derivatisation (ISO 18857-2:2009). This method is only applicable for NP1EO, NP2EO, OP1EO and OP2EO. The higher ethoxylates can only be measured with LC/MS.
2. CEN/TS 16182:2012: Sludge treated biowaste and soil - Determination of nonylphenols (NP) and nonylphenol-mono- and diethoxylates using gas chromatography with mass selective detection (GC-MS). This Technical Specification specifies a method for the determination of nonylphenols (NP), nonylphenol-monoethoxylates (NP1EO) and

It has been assumed for the purposes of this study that compliance with such a restriction limit would require substitution of 100% of the NPEs being used in imported fabrics and clothing apparel (112 tonnes). Substitution of NPEs by alternatives seems to be technically feasible, particularly given that the EU textile industry has mainly moved away from these substances. A number of available alternative surfactants are available to replace NPE in textile processing. The main alternatives to NPE can be divided into three groups:

1. Alcohol ethoxylates (AE): These are identified as the most likely alternative to NP/NPE in textiles (KEMI, 2011a). AEs are complex industrial products and are not a naturally occurring. In the last 20 years there has been a rapid growth in using AEs in laundry products and they are also the largest group by volume of the surfactants produced worldwide (ToxEcology, 2002). For example, one third of all surfactants produced in Japan are AEs and in the US production is even higher (AIST, 2009). AEs have many desirable properties for being an effective surfactant: low foaming characteristics, resistant to water hardness, good result in cleaning synthetic fibres and rapid biodegradation. It is possible that AEs can be used in lower quantities (concentrations) than NPE in certain uses which may lower costs of textile production (HERA, 2009).

2. Glucose-based surfactants: These are nonionic surfactants and the group includes: alkylpolyglucosides, glucamides, glucamine oxides and alkylglucosamides. These surfactants are readily biodegradable in standard laboratory tests. The glucose-based surfactants and their degradation products have relatively low toxicity compared to NP/NPE. There is neither concern for bioaccumulation of the parent surfactants or the degradation products (KEMI, 2011a).

3. There are also other surfactants on the market for specialized applications. For example, in textile dying, amine ethoxylates may be used. It is also common that mixtures of different nonionic and anionic surfactants or amphoteric surfactants could be used to replace NPE (ToxEcology, 2002). For example, a linear alcohol ethoxylate and an alkyl glycoside in combination can be used to replace NPE in some uses (US EPA DfE, 2011).

The suitability of these surfactants for different textile applications is not clear. It is understood that AEs (particularly, C12-C15 with ethoxylation between 3-7 EO units) are the most likely alternatives to replace NPEs as surfactants in textile processing (KEMI, 2011a).

The cost of surfactants fluctuates with the price of raw materials. In recent years, the price of alternatives has been estimated at between 20-40% more expensive that NPE (ToxEcology, 2002). It is possible that as demand for AEs increases the market price may decrease, but perhaps not down to the level of NPE (APERC, 2002). Increases in process efficiency may also be possible. In one case, the cost of scouring 15,000 tonnes of raw wool was found to be reduced by 24% when using AEs due to increases in process efficiency that reduces the use of detergent (ToxEcology, 2002). However, potential for efficiency gains will be dependent on the local facility and therefore this has not been addressed in the development of the cost curves.

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nonylphenol-diethoxylates (NP2EO) in sludge using GC-MS. For sludge a limit of detection of 0.1 mg/kg and for soil and treated bio-waste 0.02 mg/kg (expressed as dry matter) can be achieved. Lower limits of detection may be achieved by concentrating the extract by solvent evaporation.

According to industry, a CEN standard is currently being developed that will be specific for APEOs.

Consultation with industry during the current study found that the average price for NPEs is about €2/kg. Typical linear AE prices (e.g. C12/C14 with 2 EO units) are in the region of €2.0 - 2.20/kg. These prices have been used in the development of the cost curves. Costs to non-EU textile producers are assumed to be passed on to EU importers[20].

Additional costs to European importers of fabrics and clothing apparel may result from the means by which compliance is checked by companies and demonstrated to the authorities. Given the complexity of the textile industry (see Kogg, 2008), it is not entirely clear how importers will be burdened by compliance. Efforts have been made to consult with some of the large European brands and retailers who import significant quantities of textiles.

APE analysis is carried out by independent testing laboratories (such as Eurofins, Centexbel and TÜV Rheinland) and some large brands have in-house testing facilities. Consultation with testing laboratories suggests that tests currently cost in the region of €200/test.[21]

According to market actors who have been contacted, some of the larger retailers have already required that their suppliers not intentionally use NPEs in any products being supplied. As discussed in Section 4.2, a number of companies have recently extended their restricted substance lists (RSLs) to include NPEs (e.g. H&M, 2011; Adidas, 2011). These RSLs are subsequently sent to suppliers globally. Documentation demonstrating that the NPE is below a specific limit is requested and spot testing is conducted either in the companies' own laboratories or by commercial test laboratories. Most of the RSLs stipulate that the concentration should be below 100 mg/kg.

Furthermore, some companies are selective in material sourcing and choose to purchase 'approved' fabrics from accredited suppliers (such as the Bluesign® approved fabrics and the OEKO-TEX® Standard). The Bluesign criteria limits NPE concentration to 100mg/kg. The OEKO-TEX® Standard 100 certification requires a NP limit of 100ppm and total nonylphenol(1-9) ethoxylate limit of 1000 ppm.

Consequently it is recognised that certain importers have already established procedures for ensuring elimination of NPEs from their products e.g. RSLs, spot tests, certificates from upstream suppliers guaranteeing no intentional use of NPEs). However, it is important to note that global supply chains can be very complicated with many stages and sub-contractors involved. Non-compliance may occur as a result of unintentional cross-contamination rather than intentional usage. Therefore, it may not be sufficient for importers to request suppliers to stop using NPEs. Kogg (2008) provides the example of H&M which generally does not nominate fabric suppliers, meaning that when a garment is produced at a factory which is not vertically integrated, H&M will have no contractual relationship with the mill which produces the fabric, nor will they require the garment maker to provide any information about where the fabric is produced. In effect this means that H&M only has direct contractual control over the first tier of their supply chain. What goes on beyond that tier can only be controlled by checking the final product and/or through extraordinary measures.

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[20] It is worth noting that views from industry were mixed on this matter. Some stakeholders were of the opinion that costs would be passed on to downstream users whilst others considered it likely that the textile processing companies changing their production systems would simply absorb any cost changes. However, this element is a very small component of the overall textile price.

[21] It has been indicated that savings could be made in the instance of multiple screenings with negative results.

Depending on the restriction limit to be imposed, there may be the need for more frequent spot checks even from trusted manufacturers of articles of clothing.

Furthermore, no information is available from smaller importers who are unlikely to be as proactive in screening for the presence of hazardous chemicals and are unlikely to have the same power to influence their suppliers as some of the multinational companies. It is considered likely that these importers are not likely to currently check for the presence of NPEs in textiles.

The extent of the costs of compliance control are to a large extent dictated by the assumptions on testing frequency (see Table E.1). For the purposes of the abatement cost curves, a testing frequency of 5 per 10,000 articles is assumed in the first year following implementation of restriction, with that frequency decreasing to 0.005% in subsequent years. In reality, some importers may conduct checks more frequently that this to ensure compliance, whilst others may rely on documentation from upstream suppliers.

Table E.1 Relative price impact on articles due to costs of compliance control

Test frequency	Relative impact on the price of articles in %	
	Average price of articles: €15	Average price of articles: €100
1 per 100 articles	13%	2%
1 per 1,000 articles	1.33%	0.2%
1 per 10,000 articles	0.13%	0.02%

Note: €200 per test assumed.

RMO 2: A ban on all remaining use of NP/NPE in textiles in the EU

To eliminate the remaining 5,000t of NPE still used by EU companies in textile processing, a ban could be introduced. Companies would have to substitute NPE with an alternative. Consultation with industry suggests that for companies using NPEs in polymeric dispersion for coating ‘technical textiles’ in Europe it may not be a simple case of substituting NPE with AE. As described in Section 4.1.1, in this use, the NPE acts as a binder. Reformulation of the product may take up to six months (including time for health and safety checks and product trials) and the resulting formulation may be more costly or complex. However, consultation with industry suggested that the coatings confer specific technical functions to the textiles and this cannot necessarily be replicated by a reformulated product. In the absence of better information, substitution with AE is assumed and reformulation costs are estimated based on a single employee working full-time for six months.

RMO 3: End-of-pipe measures

Environmental releases arise from the washing of imported fabrics and clothing apparel containing NPE, as described in Section 4.1.2 in the main part of this report. The introduction of additional measures at waste water treatment plants (WWTPs) are therefore considered here as a set of viable options to abate emissions of NP/NPEs to water.

Nonylphenol removal in conventional WWTPs (mechanical, biological and chemical treatment and improved nitrogen removal) has proved to be efficient. Since NP has a low solubility, high sorption potential and low volatilisation potential, degradation and sorption in sewage sludge are the main mechanisms involved in the removal of NPs from waste waters (COHIBA, 2011).

According to Fauser et al. (2001) NPE parent oligomers are efficiently eliminated during biological treatment. The degradation products predominantly comprise nonylphenol (NP), nonylphenol-monoethoxylate (NPME), nonylphenoldiethoxylate (NPDE) and nonylphenoxy carboxylic acids (NPEC). The abundance of the particular metabolite is dependent on the treatment conditions and influence of physicochemical processes (Ahel et al., 1994).

NPE removal from WWTPs under the baseline have been laid out in the main report. However, if tertiary waste water treatment is required, this can impose significant costs. The following advances measures have been considered:

- Membrane filtration: Membrane filtration includes Nanofiltration (NF) or Reverse Osmosis (RO). Effectiveness of NF filters removing NP from MWWTP effluents is found to vary from 70% to 100% (COHIBA, 2011). In the case of RO the effectiveness is generally above 98% (Feenstra et al., 2009).

- Ozone oxidation: According to SOCOPSE the chemical oxidation treatment efficiency for NP removal is 90%. The costs of ozone treatment depend on the quality of the water (organic load) and contact time for oxidation.

- Activated carbon: Effectiveness of AC filters at WWTP depends on the concentration range of pollutants, technical parameters and matrix. At well maintained MWWTP reduction rates for NP of 50 to 99% can be observed. However, far lower reduction efficiencies (25%) have been observed (COHIBA, 2011). AC has large cross substance effects e.g. elimination of TBT, PFOS, Cd, Hg and other pollutants.

Capital and operational costs for these three measures are illustrated in Table E.2. Of the three technologies that will be assessed, the total costs (capital and operating) are highest for ozone oxidation and lowest for activated carbon[22].

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[22] In practice, costs will vary significantly amongst installations and according to technologies and suppliers used. In addition, wide ranges of estimated costs are presented in the literature, leading to additional uncertainties.

Table E.2 Costs of techniques for abatement of NP/NPE emissions

Technology	Reduction	Capital costs (€k)	Operational costs (€k/year)	Total annualised cost (€m)	COHIBA (2011) results (for comparison) (€/kg emitted NP/NPE)
Activated carbon	25-99%	80	225	12,026[1]	12,000 – 19,000,000
Ozone oxidation	90%	1,520	225	68,853	93,000 – 4,200,000
Membrane filtration	70-100%	768	225	39,165	120,000 – 12,000,000

Note: This figure compares well with the estimate provided by EUREAU (during consultation) that costs are estimated to be in the region of €20-30/capita. Total EU-27 population in 2010 was estimated at 501m, resulting in a per capita cost of €24.

Nonylphenols are one of the 33 priority substances in the field of water policy listed in the Annex II of the Environmental Quality Standards Directive (Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008). In 2011, the EC published a proposal amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy. The proposal includes 48 substances.

The installation of the tertiary end-of-pipe techniques described above at industrial and municipal wastewater treatment plants across the EU would result in the removal of other pollutants present in the wastewater flow apart from NP/NPEs. Whilst the co-benefits of WWTP measures are well understood there is currently no agreement on how to account for this quantitatively. This methodological issue was discussed extensively during the Entec (2011) study for DG ENV "Technical Support for the Impact Assessment of the Review of Priority Substances under WFD". Possible options included a multi-criteria analysis (MCA) however no conclusions could be drawn on the best way to do this. The final report concluded that "costs for advanced sewage treatment to meet an EQS do not take account of the co-benefits of removing other substances at the same time….However, without a site-specific assessment of downstream impacts of WWTP discharges it is not possible to determine the co-benefits which might be realised." Therefore, these have not been quantified for the purposes of the current study but they are likely to be significant.

Instead the table below presents available information on the ranges of removal efficiencies of other priority and priority hazardous substances listed in the EQS Directive and amendment proposal by the three WWTP approaches considered here. Please note that the information summarised in the table is not exhaustive and is presented for illustrative purposes.

Table E.3 Removal efficiencies of priority substances by three tertiary WWTP techniques

CAS Number	Substance	Status	Reported removal efficiency (in per cent)[1]		
			Activated carbon	Ozone oxidation	Membrane filtration
15972-60-8	Alachlor	EQS Directive			Solar: ~40%
120-12-7	Anthracene	EQS Directive		>70%	90% (MBNDC)
1912-24-9	Atrazine	EQS Directive		9%	<10% (MBNDC) Solar: 70%
71-43-2	Benzene	EQS Directive			92-98% (MBNDC)
	Brominated diphenylethers	EQS Directive	90%		
7440-43-9	Cadmium and its compounds	EQS Directive		>90%	
470-90-6	Chlorfenvinphos	EQS Directive			Solar: 70%
107-06-2	1,2-dichloroethane	EQS Directive		71%	
117-81-7	Di(2-ethylhexyl)phthalate (DEHP)	EQS Directive			96% (MBNDC)
330-54-1	Diuron	EQS Directive			<10% (MBNDC) Solar: 70%
206-44-0	Fluoranthene	EQS Directive			83-98% (MBNDC)
608-73-1	Hexachlorocyclohexane	EQS Directive			60% (conventional)
34123-59-6	Isoproturon	EQS Directive		25% <10% (MBNDC)	Solar: 70%
7439-92-1	Lead and its compounds	EQS Directive			78% (MBNDC)
7439-97-6	Mercury and its compounds	EQS Directive		>90%	
91-20-3	Naphthalene	EQS Directive			95-96% (MBNDC)
7440-02-0	Nickel and its compounds	EQS Directive			29% (MBNDC)
87-86-5	Pentachlorophenol	EQS Directive		99%	10-50% (MBNDC)
	Polyaromatic hydrocarbons (PAH)	EQS Directive			10-90% (MBNDC)
122-34-9	Simazine	EQS Directive		95%	<10% (MBNDC)
	Tributyltin compounds	EQS Directive			10-90% (MBNDC)
12002-48-1	Trichlorobenzenes	EQS Directive		95%	<10% (MBNDC)
1582-09-8	Trifluralin	EQS Directive		99%	

Table E.3 (continued) Removal efficiencies of priority substances by three tertiary WWTP techniques

CAS Number	Substance	Status	Reported removal efficiency (in per cent)[1]		
			Activated carbon	Ozone oxidation	Membrane filtration
115-32-2	Dicofol	Proposal	>85% [2]	100% (when combined with preoxidation by ozone)	80-85%
1763-23-1	Perfluorooctane sulfonic acid and its derivatives (PFOS)	Proposal	99% (historic pollution)		
	Dioxins and dioxin-like compounds	Proposal	>90% [3] (historic pollution)		~70% (historic pollution)
52315-07-8	Cypermethrin	Proposal	~98%		UV: ~11%
62-73-7	Dichlorvos	Proposal	na (insecticide: agricultural use)	na (insecticide: agricultural use)	na (insecticide: agricultural use)
	Hexabromocyclododecanes HBCDD	Proposal	99%		UV: 3%
76-44-8 / 1024-57-3	Heptachlor and heptachlor epoxide	Proposal	>90% [2]		70-90%
886-50-0	Terbutryn	Proposal	~99%		UV: ~24%
57-63-6	17alpha-ethinylestradiol	Proposal	~98% [4]		UV: ~25% [5]
50-28-2	17beta-estradiol	Proposal	96-99% [4]		~25% [5]
15307-79-6	Diclofenac [9]	Proposal	80-99%	22-92%	>95% (ozonation); UV ~59%

Notes:

1) ScorePP (2008) and ScorePP (2009)

2) Ormad (2008)

3) US EPA (2010)

4) Felebuegu et al (2006)

5) Defra/Water Industry EDC demonstration Programme data

6) Knappe (2008)

7) MBNDC = Mechanical, Biological, nitrifying/denitrifying, chemical treatment

Data for incorporation into cost curves

The table below outlines the assumptions and data used in developing specific measures for the cost curve for NP/NPE in textiles.

Table E.4 Summary of measures for inclusion in cost curves

Measure	Details of key elements of measure
Ban on the placing of the market of imported textile articles containing NP/NPE (BBAU2)	*One-off costs to EU fabric/apparel importers:* Includes: Communication costs to communicate changes to suppliers and downstream supply chain (estimated as 1 FTE working full-time for 1 month at an hourly price of €18.75[23]) which gives €2,813; 50% of companies are assumed to increase spot testing of textiles, estimated as €200 per test (based on consultation with Oeko-Tex and Centexbel) and 0.05% sampling frequency in the first year following implementation of restriction. [The remainder of companies are assumed to rely on communication from suppliers or are already undertaking spot testing of NPEs]. Total one-off costs are estimated as €67,732 per company. Equivalent annual one-off costs (€) are estimated in the region of €437m. *Recurring annual costs to EU fabric/apparel importers:* Operational costs considered include: testing of textiles for NPE content, estimated as €200 per test and 0.005% sampling frequency and costs of substituting NPE used in textile processing. It is assumed that all NPE is replaced with alcohol ethoxylates (AEs) with average price of €2.10 (consultation with a textile company which compares well with EcoToxecology 2002 report). It is assumed that costs of substituting NPEs with AEs in textile processing outside of the EU are passed on to textiles importers in the EU. Annual ongoing costs are estimated to be in the region of €349m. *Emissions reduced:* Emissions to SW from imported textiles avoided: 375t of NPE.

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[23] Eurostat (2011) average wage for worker in manufacturing sector.

Measure	Details of key elements of measure
	Main uncertainties and limitations • Data on one-off costs are based on limited information and are subject to uncertainty. • Testing frequency for demonstration/achieving compliance is highly uncertain. • Assumed that all NPEs will be replaced by AEs – this is an over-simplification of what may happen in practise.
Ban on all textile products containing NPE in Europe (BBAU 1)	*One-off costs to EU producers:* Includes: R&D costs to reformulate NPE-containing product to use in textile processing (estimated as 1 FTE working full-time for 6 months at an hourly price of €18.75[24] this is based on consultation with a textile company) which gives €16,875; communication costs to communicate changes to suppliers and downstream supply chain (estimated as 1 FTE working full-time for 1 month at an hourly price of €18.75[25]) which gives €2,813. Total one-off costs are estimated as €19,688 per company. Equivalent annual one-off costs (€) are estimated in the region of €0.44 million.
	Recurring annual costs to EU producers: Operational costs considered include: replacing NPEs used in textile processing with alcohol ethoxylates (AEs) with average price of €2.10. Annual ongoing costs are estimated to be in the region of €0.5 million.
	Emissions reduced: Emissions to SW from textiles avoided: 12t of NPE.

Table E.4 (continued) Summary of measures for inclusion in cost curves

Measure	Details of key elements of measure
	Main uncertainties and limitations • Data on one-off costs are based on limited information and are subject to uncertainty. • Assumed that all NPEs will be replaced by AEs – this is an over-simplification of what may happen in practise.
Advanced WWTP tertiary measures (BBAU 3 – 5)	*One-off costs:* One-off costs are based on the European Commission's (2009) draft BREF document on common waste water and waste gas treatment/management. Prices have been adjusted for inflation to 2011 prices. • Activated carbon (BBAU 3): Total one-off costs are estimated as €0.6m per WWTP for activated carbon. A lifetime of 10 years is assumed. Equivalent annual one-off costs (€) are estimated in the region of €80k. • Membrane filtration (BBAU 4): Total one-off costs are estimated as €6m per WWTP for activated carbon. A lifetime of 10 years is assumed. Equivalent annual one-off costs (€) are estimated in the region of €0.8m. • Oxidative techniques (BBAU 5): Total one-off costs are estimated as €12m per WWTP for activated carbon. A lifetime of 10 years is assumed. Equivalent annual one-off costs (€) are estimated in the region of €1.5m.

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[24] Eurostat (2011) average wage for worker in manufacturing sector.

[25] Eurostat (2011) average wage for worker in manufacturing sector.

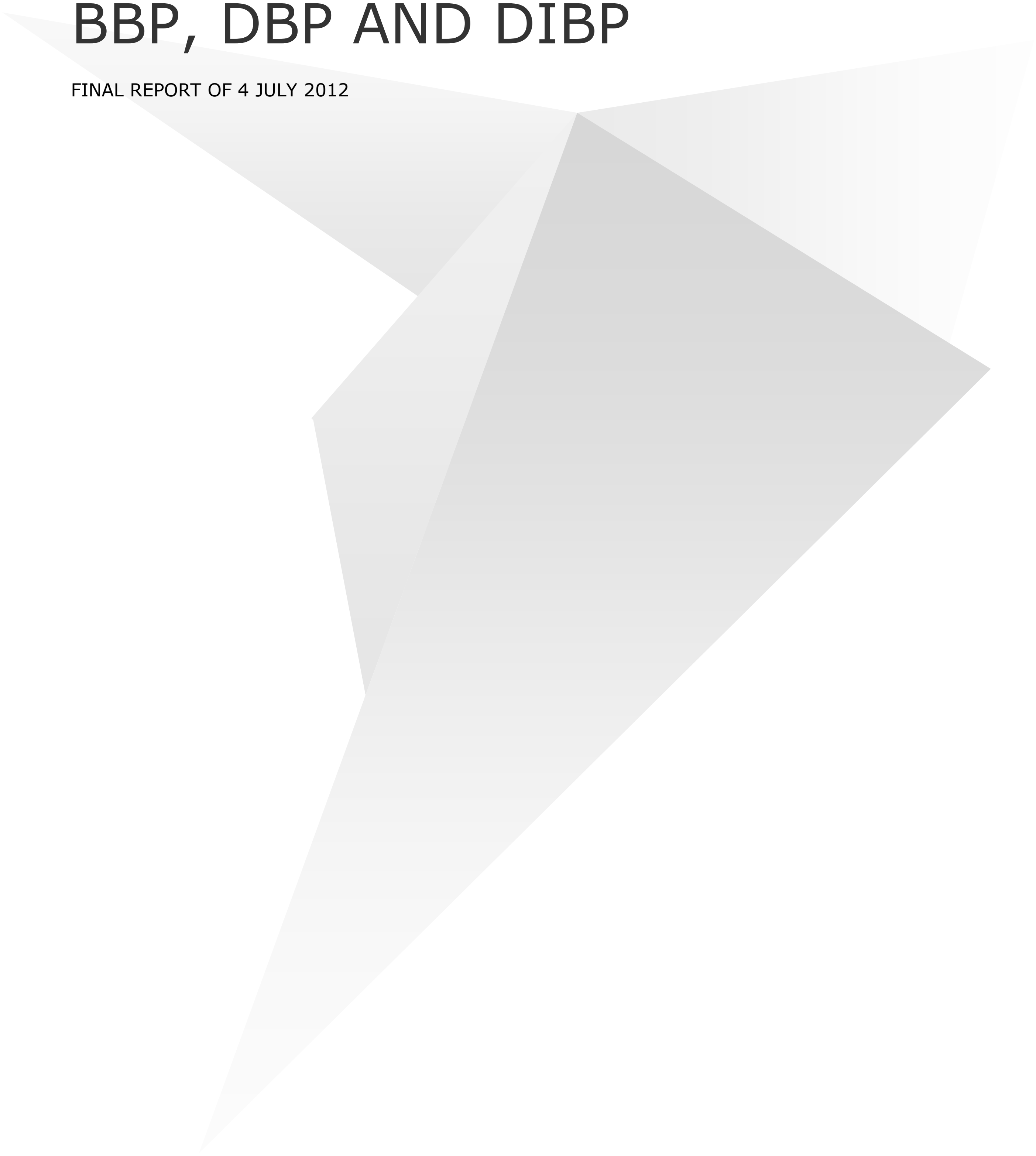
Measure	Details of key elements of measure
	Recurring annual costs: Ongoing costs are based on the European Commission's (2009) draft BREF document on common waste water and waste gas treatment/management. Prices have been adjusted for inflation to 2011 prices. Ongoing costs are estimated as €225,014 per WWTP for each measure. A lifetime of 10 years is assumed. *Emissions reduced:* • Activated carbon: Removal efficiency between 25% and 99% (mid-point of 62% used in analysis). Total emission reduction = 111t. • Membrane filtration: Removal efficiency between 70% and 100% (mid-point of 85% used in analysis). Total emission reduction = 152t. • Oxidative techniques: Removal efficiency of 90%. Total emission reduction = 161t. *Specific measures taken forward:* • All three measures (BBAU 3 BBAU 4, BBAU 5) are included in the cost curve data. *Main uncertainties and limitations* • Data on one-off costs are based on limited information and are subject to uncertainty.

F1

EUROPEAN CHEMICALS AGENCY, ECHA

ABATEMENT COST CURVES FOR THE FOUR PHTAHALATES DEHP, BBP, DBP AND DIBP

FINAL REPORT OF 4 JULY 2012



COWI

CONTENTS

Preface

The present report has been prepared by COWI A/S for the European Chemicals Agency, ECHA under the contract ECHA/2011/140.

The study has been supervised by Jukka Peltola and Kalle Kivelä, ECHA.

The work group has consisted of Carsten Lassen (project manager), Jakob Maag and Michael Munk Sørensen, COWI.

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1 Summary

This report provides the results of a test of a method developed as part of the pilot study “Abatement cost curves for substances of concern”. Based on the method, cost curves for the four phthalates DEHP, BBP, DBP and DIBP have been developed. The four phthalates were chosen by ECHA for the study based on the substances’ potential relevance for risk management activities under REACH.

The study involved collection of data from the literature and from stakeholders on the experience with replacing the four phthalates. The data collected includes data on one-off costs of the replacement and changes in operating costs. Emission factors used for estimating the emission reduction and data on the breakdown of the total EU consumption of the four substances by applications areas were derived from previous studies.

The estimated total costs of reducing the emissions are summarised in the table below. Besides the least-costs alternative scenario, a scenario is developed for DEHP where it is assumed that the four phthalates are replaced by non-phthalate alternatives.

Table 1.1 Total potential emission and consumption reduction as compared to the 2011 level and estimated costs, best estimate

Substance	Total emission reduction (t/y)	Total consumption reduction (t/y)	Total costs (€m/year)
DEHP	4,729	119,800	41.5
BBP	96	4,000	0.9
DBP/DIBP	443	8,940	3.0

The cost curves illustrate the cost-efficiency of reduction of the emission of the four phthalates by application area. The study shows that the costs curves and ranking of the cost-efficiency of the measures are nearly 100% determined by the applied emission factors for the different application areas (the emission factors cover formulation, processing and service life). The four phthalates are for all application areas used as plasticisers, and the differences in the costs of substitution of one tonne

of the phthalates for the different application areas are small compared to the large differences in the life-cycle emission factors.

No information was received indicating that the remaining uses of the substances represent application areas for which it is particularly difficult to replace the four phthalates.

The study describes the key limitations and uncertainties and discusses the applicability of the method for the four phthalates. Key uncertainties are associated with:

- Life-cycle emission factors and differences in potential environmental and human health effects of the emissions to the different compartments.

- BBP, DBP and DIBP are produced by very few companies, and the information on the consumption by application area is consequently considered confidential. Furthermore, the substances have been phased out by most users and it is not possible to estimate the consumption of the substances on the basis of data on the production volume of final articles.

- Fluctuations in differences in price between the four phthalates and the main alternatives. Furthermore, many alternatives are manufactured by one manufacturer only and prices are considered confidential.

- One-off costs of substitution varies among users, however the available information indicates that the one-off costs in general are small or insignificant.

- Applicability of alternatives. For the main least-costs alternatives much experience has been build up. For DEHP, experience with non-phthalate alternatives for some application areas is very limited.

The overall conclusion of the test is that the methodology for these substances provides limited new information for prioritisation of measures, and furthermore the results are highly uncertain.

On the basis of the test, it is suggested that the costs curve methodology would be most informative for prioritisation processes for substances which meet a number of specific criteria mentioned in the report, for example with more diverse cost effectiveness for different measures. The report provides some examples of substances for which the method may inform prioritization processes better.

2 Introduction

2.1 Background and objectives

The objective of this study is to help ECHA in their efforts to establish a capability to assess the abatement costs of reducing the use or consequent emissions of hazardous substances.

During 2010, ECHA has been collaborating in a pilot study concerning "Abatement cost curves for substances of concern" with UK Environment Agency, UK Health and Safety Executive as well as RIVM (Corden *et al.*, 2011). This study explored how marginal or average abatement costs of measures could be used to assess the cost-effectiveness of phasing out the use of a chemical substance or reduce any consequent emissions from its lifecycle. The abatement costs include costs relating to i) using alternative chemicals, ii) applying alternative techniques, or iii) applying means of reducing emissions (e.g. end-of-pipe or process techniques) arising from chemicals use.

The present study is a continuation of this line of work.

The four phthalates DEHP, BBP, DBP and DIBP have been chosen by ECHA for the study based on their potential relevance for risk management activities under REACH. The study is intended to assist ECHA in assessing the cost-effectiveness of phasing out the use of the four phthalates or reduce any consequent emissions by the application of different risk management measures.

Current activities concerning the four phthalates

The phthalates DEHP, DBP, BBP and DIBP are all classified as toxic to reproduction category 1B according to the CLP Regulation (Regulation (EC) No 1272/2008 on the classification, labelling and packaging of substances and mixtures).Furthermore, entry 51 of Annex XVII to REACH includes the restrictions on the placing on the market and use of DEHP, DBP and BBP in toys and childcare articles.

Different management options for the four phthalates are currently under review:

- The European Commission has requested ECHA to review the available new scientific information for these phthalates and to evaluate whether there is evidence that would justify a re-examination of the existing restriction of DEHP, DBP and BBP in toys and childcare articles.

- Denmark has submitted an Annex XV restriction report proposing restriction of the placing on the market and use of certain articles containing DEHP, BBP, DBP and DiBP. In the report, Denmark suggests a ban for the placing on the market of articles intended for indoor use and articles that may come into direct contact with the skin or mucous membranes, containing one or more of these four phthalates in a concentration greater than 0.1 % by weight of any plasticised material. The six-month public consultation on the restriction report was concluded on 16 March 2012.

- All four phthalates are included in the Annex XIV to the REACH Regulation and are substances subject to authorisation. The latest application date (for authorisation) is 21 August 2013 and the sunset date 21 February 2015.

2.2 Applied methodology

The methodology proposed in the pilot study specifies a range of possible stages to generate costs curves for chemicals of concern (Corden *et al.*, 2011). The approach selected for the different stages of this study is summarised in Table 2.1 and further discussed below the table (stage numbering in accordance with the pilot study).

Table 2.1 Approach selected for this study

Stage	Approach selected for this study
1. Set boundaries of the cost curves	
1.1 Select substance(s) to be assessed	The four phthalates DEHP, BBP, DBP and DIBP were selected for the analysis. Due to limited information on the use of DIBP and the fact that DBP and DIBP are used for many of the same application, these two substances are addressed together.
1.2 Determine the geographical scale	EU27
1.3 Determine the economic boundary of the analysis	The study focuses on substitution of the phthalates in the production of mixtures and articles. In the first step the manufacturers of the mixtures and articles would be affected, but the costs of substitution of the phthalates may be transferred down the supply chain.
1.4 Determine the reference year(s) for the analysis	2011
2. Quantification of current and predicted future emissions	
2.1 Identify relevant lifecycle stages and uses of the substance	Data on life lifecycle stages and the use of the substances and consumption by application areas is mainly based on three reports prepared for ECHA in 2009 (COWI *et al.,* 2009,a,b,c) supplemented by data obtained directly from stakeholders as part of this study.

Stage	Approach selected for this study
2.2 Quantify current environmental releases	The quantification of environmental releases is mainly based on application-specific emission factors from COWI *et al.* (2009,a,b,c). The quantified emissions include emissions to air, soil and water (including waste water) from the life cycle stages: formulation, processing, and service life. Possible emissions from the waste disposal are not included due to lack of data which can allocate such emissions to the individual applications. The background reports (COWI *et al.,* 2009,a,b,c) estimated that the emissions from the waste disposal in the short-term is small compared to the emissions from other life cycle stages and e.g. for DEHP it account for less than 1% of the total emissions. The long-term releases from landfills may in fact be higher than indicated if all releases occurring until the DEHP is ultimately degraded are considered, but no data on the long-term fate of DEHP in landfills have been available. In the present assessment disposal of the substance to landfills are not considered a release to the environment. However, other system boundaries may be used where the disposal is considered a release to the environment which results in higher estimates for the releases to the environment (e.g. Socopse, 2009).
3. Existing and planned abatement techniques (the business as usual scenario)	
3.1 Identify and characterise existing abatement techniques and those already planned under current policy (business-as-usual scenario	The study has focused on substitution of the substances which will result in emission reduction from all life-cycle stages. The cost estimates are based on replacement with alternatives which are already in use and have a significant market share. It is for the cost curves assumed that those manufacturers still using the four phthalates would continue to do so under the current policy. In fact a decreasing trend in the use of the substances is seen and this trend may continue the coming years. As a consequence of planned authorisation procedure for the four substances, the consumption of the substances may further decrease the coming years. The level of substitution of the substances is described for each of the substances in the substance specific chapters.
3.2 Estimate baseline (annual) environmental releases in reference year	Annual baseline environmental releases by application area in reference year are calculated from the life-cycle emission factors and the estimated consumption of the phthalates for the different application areas. Releases to air, soil and water (incl. waste water), both diffusive and abrasive releases, are added into one figure.
4. Identify and characterise possible future abatement measures	
4.1 Identify possible future measures to reduce emissions	The study focuses on substitution of the substances (as result of a restriction); other measures for the further reduction of emissions from the different life-cycle stages have not been assessed. The study lists the different alternative substances which may be used for replacement of the phthalates for each of the application areas. For all applications viable alternatives are today available on the market.

Stage	**Approach selected for this study**
4.2 Characterise possible future measures	For all substances it is estimated that the maximum feasible uptake of alternatives for each application area is 100%. The emission abatement potential for all life cycle stages is estimated to be 100% of the current emission. One-off costs (such as capital equipment, product reformulation), operating costs, and economic lifetime of measure (e.g. technical lifetime, expected amortisation period) is described on the basis of information obtained from manufacturers of the phthalates and alternatives as well as downstream users. Costs are financial costs and the price level is 2011. The discount rate for the calculations is based on EU Impact Assessment Guidance value and it is 4% (EC, 2009).
5. Development of cost curves	
5.1 Develop spreadsheet (or other) model	The spreadsheet developed as part of the pilot study (Cordon *et al.*, 2011) has been used as starting point for the further development of spreadsheet.
5.2 Estimate maximum feasible emission reduction for each measure	It has been estimated that a 100% reduction of emissions is feasible for all assessed measures.
5.3 Estimate equivalent annual cost of each measure	The total annual costs of the measures have been estimated as the sum of the total equivalent annualised one-off costs and the total annual operating cost (including costs of raw materials).
5.4 Initial ranking based on cost-effectiveness	The measures have been ranked on the basis of cost-effectiveness expressed as the total costs of reducing the releases of the substances by one tonne. As a pilot trial the measures have furthermore been ranked on the basis of the cost-effectiveness expressed as the total costs of reducing the consumption of one tonne of the substances.
5.5 Determine interactions of measures	The assessed measures do not interact.
5.6 Calculate total emissions abated and total cost for each measure in order of expected preferential uptake	It has been assumed that the preferential uptake of each measure is 100%
5.7 Present results in order of preferential (most cost-effective) uptake	The data have been presented by two cost curves: one showing the accumulated costs of reducing the remaining emission and a second showing the marginal cost of single measures for emission reduction. In both curves the measures are ranked by their cost-effectiveness.

Emission abatement scenario

The used scenarios basically assess the effect of restricting the use of the substances for all applications and thereby reducing the emissions. As alternative plasticisers are available for all applications, and in fact are widely implemented within the EU, it is estimated that a 100% substitution with other plasticisers is realistic using known techniques.

The scoping of the study, within the limits of the resources available, should not be considered as a rejection of other measures that may be relevant for reduction of the emission of the phthalates concerned.

In the context of potential future emissions abatement under REACH, it may be relevant to take into account reductions in emissions and/or use of the substances that could be achieved through other legislative controls. First of all it may be relevant to take into account the cost-effectiveness of measures that may be introduced as part of the conditions for authorisation.

Furthermore, measures to ensure safe disposal and increase recycling of PVC might be taken through European waste legislation. Similarly, the IPPC Directive could be amended, for instance, to extend the scope of relevant BREFs to include considerations specific to the four phthalates.

The analysis uses to the extent possible life-cycle emission factors as an indication of the total environmental releases. The emission factor includes emission from formulation, processing and service life. The quantity (or activity rate), which is multiplied with this factor, is the total quantity of the substance used for formulation and processing in the EU, regardless of whether the final articles are exported or used in the EU. The total quantities of the substance manufactured in the EU, and the total quantities in the marketed articles (which depend of the import/export in articles) would differ from this quantity.

Costs elements considered

The costs elements which have been considered for substitution of the phthalates are the following:

- Operating costs:
  - Change in effective costs of alternative plasticiser (price of alternative and substitution factor describing the ratio of needed original plasticiser to the alternative to obtain the same flexibility in the produced material);
  - Change in costs of other raw materials (e.g. need for use of other pigments or fillers or changes in quantities of resin used);
  - Changes in cost of monitoring and control (e.g. costs of monitoring and control of the substance in the working environment, costs of monitoring of environmental releases, etc);
  - Other changes in operating cost (e.g. higher energy consumption, higher manpower costs);
  - Any changes in product quality that might result in changes in price of product.
- One-off costs:
  - Costs of research and development (R&D);
  - Investment in new equipment or physical alterations of production lines;
  - Investment in more production lines of same type due to lower throughput;

- Costs of product redesign.

The questionnaire has also included a question concerning the price of the final material/mixture compared to a material/mixture with the concerned phthalate. The price difference of the final material/mixture may be considered an aggregated indication of all costs of substitution.

It has been common in assessments of the feasibility of replacement of the phthalates to estimate the incremental costs of materials by multiplying the incremental price of the alternative plasticisers with the substitution factor. The substitution factor indicates the amount of alternative plasticiser (on a weight basis) to be added to the polymer (mostly PVC) in order to obtain the same plasticising properties. The plasticiser most often takes up about 20-40% of the material, and the price of the plasticiser is a very significant part of the total price of the material part. However, increasing the amount of plasticiser implies in many cases that less PVC (or other) resin is used in order to obtain the same volume of final plastic or mixture. Due to data limitations it has in general not been possible to take changes in the consumption of other raw materials into account.

Product quality

Replacement of the plasticiser may affect the quality of the material and thereby the quality of the final article. As consequence, the product may last for shorter time and this may imply extra costs to the users of the product. Many products of PVC last for many years e.g. flooring or cables sheaths and the time the plasticised PVC can retain its properties is determining for the lifetime of the articles. Many new plasticisers have been introduced relatively recently and it us uncertain to what extent the plasticisers may result in a shorter lifetime of the material. Due to lack of information, possible effects on the lifetime of the articles hve not been taken into consideration in the assessment. Manufacturers will generally seek to maintain the durability of the product when substituting the plasticiser.

Substitution scenarios

For DEHP two substitution scenarios have been considered: the main scenario with least cost alternatives (orthophthalates and terephthalates), and a theoretic scenario with least costs non-phthalate alternatives. The main alternatives to DEHP, DINP and DIDP have recently been evaluated by ECHA (ECHA, 2012). The inclusion of a non-phthalate substitution scenario in the present study should not be considered as compromising the conclusions drawn, that no further risk management measures are needed to reduce the exposure of children and adult to these phthalates (except for a few specific applications where some uncertainties still exist).

2.2.1 Pilot study cost curves for DEHP

The pilot study (Cordon *et al.*, 2011) used DEHP for one of the three case studies. The case study considered, besides substitution of DEHP which is also included in the present study, a number of measures addressing emissions of DEHP from industrial processes and waste water treatment plants (measures underlined below were included in the pilot study):

- Additional advanced (tertiary) water treatment for manufacture of DEHP, including membrane filtration, ozone oxidation and activated carbon.

- Additional waste gas treatment for manufacture of DEHP, including biofiltration, coolant condensation and thermal oxidation.

- Additional advanced (tertiary) water treatment for formulation and processing, including membrane filtration, ozone oxidation and activated carbon.

- Additional waste gas treatment for formulation and processing, including biofiltration, coolant condensation and thermal oxidation.

- Additional advanced (tertiary) water treatment at public waste water treatment plants (WWTP) to treat releases from indoor and outdoor public use, including membrane filtration, ozone oxidation and activated carbon.

Of these additional measures, waste gas treatment for manufacture of DEHP by coolant condensation was the most cost-effective of all measures assessed whereas additional advanced (tertiary) water treatment for formulation and processing by membrane filtration was the least cost-effective of all the measures. The cost-effectiveness of the other two measures was of the same magnitude as the substitution measures.

For the present study it has, in consultation with ECHA, been decided to focus on the substitution measures. Reference is made to Corden *et al.* (2011) and Socopse (2009) for more information on the cost-efficiency of the other measures.

2.2.2 Stakeholder consultation

For the collection of information on one-off costs and incremental operating costs by the use of alternative plasticisers, data have been collected from manufacturers of the four phthalates and the alternatives as well as from trade organisations and downstream users. For the stakeholder consultation two questionnaires were developed: one for manufacturers and one for downstream users. As the four phthalates have already been replaced by other plasticisers by the majority of downstream users, the actual downstream users are difficult to identify. Consequently, manufacturers of the four phthalates were encouraged to forward the questionnaire for downstream users to their customers in order to obtain information from the actual users of the substances.

The questionnaires were sent by e-mail to contact persons in the companies, which have been providing information for previous studies (COWI *et al.*, 2009 a,b,c; Maag *et al.*, 2009). Manufacturers of the substances and alternatives who did not answer were subsequently contacted again in order to identify the right contact person, and the questionnaire was forwarded to this person. In case of no answer, the companies were contacted a third time.

Very limited information has been obtained from the current manufacturers of the four phthalates. The three manufacturers in Eastern Europe of one or more of the four phthalates concerned have all informed that they have no information to provide to the study. Two of the manufacturers inform that they are in the process of collecting information for the authorisation process, but would not have the full overview until June 2012. One manufacturer indicates that the remaining applications of DEHP are for particular purposes, but that they are not in a position to provide detailed information on the uses. The manufacturers have been encouraged to send the questionnaire to customers, but no answers have been obtained from customers.

Of the three manufacturers of DEHP in Western Europe only one has responded and forwarded the questionnaire to a number of actual downstream users, who have responded after receiving the questionnaire from the manufacturer. Information was obtained from all manufacturers of BBP.

Information has been obtained from a number of manufacturers of alternatives as listed in the tables in the chapters for each of the phthalates. The manufacturer of one of the most used alternatives, DINCH, has not answered, and consequently details on the use of this substance are not included in the assessment. Some alternatives are manufactured by several manufacturers, but the alternatives from the manufacturers addressed are considered to be among those for which most experience has been gained. They are also assumed to be among the least-costs alternatives. An ICES blog article from April 2012 describes that the phthalate-free plasticiser market is growing and the article mention a number of manufacturers with new activities regarding non-phthlate plasticisers: Lanxess, BASF, Scandiflex, Dow, Galata, PolyOne and Myriant (ICES, 2012). It has been beyond the scope of the current project to provide a full overview of all alternatives on the market.

Several alternatives (e.g. COMGHA) have significantly higher price than DEHP and are marketed for specific purposes where the customers request plasticisers with an optimised health profile, e.g. for toys, food-contact materials and water beds. For the users requesting these alternatives, the use of the substances have some added values (e.g. used in the marketing) which is considered to compensate for the higher price of the plasticisers. Consequently, the shift to these plasticisers is not solely a consequence of a restriction of the use of DEHP (similar issue for BBP and DBP/DIBP). No new information has been collected for the relatively high-costs alternatives as the study has focused on developing cost curves for least-cost alternatives.

The questionnaire have been forwarded to national trade organisations by the EU-wide trade organisations European Plastic Converters (EuPC) , the European Council of producers and importers of paints, printing inks and artists' colours (CEPE), and the European association of manufacturers of adhesives (FEICA). One answer has been obtained on DBP/DIBP from the British Adhesives and Sealants Association (BASA) summarising information from several down-stream users. Furthermore the European Council for Plasticisers and Intermediates, ECPI, has been contacted for discussing the opportunities of obtaining updated information on manufactured volumes.

3 DEHP

3.1 Use of DEHP

Bis(2-ethylhexyl)phthalate (DEHP) is a general purpose plasticiser that is slower fusing, exhibiting higher viscosity and lower volatility compared to the three other studied phthalates DBP, DIBP and BBP, yet quicker fusing, with lower viscosity and higher volatility than DINP and other general purpose phthalates which are used as substitutes for DEHP. Market information on DEHP by applications area is described in section 3.2.1.

Different properties of the plasticisers are required for the different types of PVC processing (ExxonMobil, 2011):

- Plastisol processing (e.g. spread coating of wall covering, cushioned flooring, bags, and coated fabrics etc) requires plasticisers with low neat viscosity, good gelation properties, low volatility and ensuring plastisol storage stability.

- The calendering process (shower curtain, tablecloth, batch equipment, tiles etc) requires plasticisers that have low volatility, good processability (good solvators for PVC, not too viscous) and good resistance to extraction.

- The extrusion process requires very permanent plasticisers, good solvators for PVC, not too viscous and that the plasticiser can be processed and fused at a reasonable temperature (180ºC - 200ºC).

In order to obtain the desired properties, secondary plasticisers such as BBP, DIBP and DBP are used together with DEHP for some types of processing. DBP, DIBP is (was) in PVC used for their viscosity reducing properties as processing aid for PVC plastisols and compounds typically in quantities of 5 to 10 wt% due to their higher polarity. BBP is (was) in PVC used primarily as a fast fusing secondary plasticiser for foamed plastisols e.g. used in flooring (ExxonMobil, 2011). When replacing DEHP for these applications, the general purpose plasticisers would typically be used together with secondary plasticisers so the combined plasticiser system provides the same properties as in the DEHP/secondary plasticiser system.

3.1.1 Price information

The main cost element for the substitution of DEHP is (as shown later in this chapter) the incremental price of the alternative plasticiser and obtaining better pricing information has been a focus area for the data collection.

The main alternatives to DEHP in Europe have so far mainly been the phthalates DINP, DIDP and DPHP (di(2-propylheptyl) phthalate.

Information from ICIS Pricing

Prices of DEHP, DINP, DIDP and DPHP) have been obtained from the website of ICIS Pricing (www.icis.com). The website does not provide information on prices of non-phthalate alternatives.

Public available data from ICIS' website indicates for 9 September 2011 a price of 1,470-1,520 €/tonne for DEHP and 1,670-1,770 €/tonne for DINP and DIDP, corresponding to a price difference of 13-16% (ICIS, 2011a). Four weeks before the prices were about 30 €/tonne higher, but the price difference was the same. The price of DPHP in September 2011 is indicated at €1,690-1,790 €/tonne.

The prices 18 July 2011 were at 1,520–1,570 €/tonne for DEHP, 1,750–1,800 €/tonne for DPHP and at 1,700–1,800 €/tonne for DINP and DIDP (12-15% higher). (ICIS 2011b).

Data provided by one of the manufacturers makes reference to ICIS_LOR publication Friday 3/2/2012 and prices of 1,370-1,410 €/tonne for DEHP and 1,540-1,640 €/tonne for DINP (12-16% higher). The fact that the manufacturer uses these data from ICIS as reference indicates that they are regarded as a good indication of the actual bulk prices.

The data at ICIS pricing indicates that the prices of the two substances fluctuates, but on the large scale more or less follows each other. This is supported by some of the reports on ICIS pricing e.g. from 26 July 2010: "*Meanwhile, prices for both products have already skyrocketed since the beginning of the year; both have climbed 66% above the average levels of January to €1,650-1,700/tonne FD (free delivered) NWE (northwest Europe) for DINP and to €1,525-1,575/tonne FD NWE for DOP*" (ICIS, 2010). (DOP is identical with DEHP).

The price differences, however, have changed even the prices of DINP and DEHP on the large scale follow its other. Using average prices, the difference has increased from 7% in March 2010 to 15% in September 2011.

Changes in prices are both driven by changes in price of feedstock, and differences in the demand/supply. Examples of both mechanisms from ICIS:

› *"In addition to environmental pressure, the European DOP market has faced increasing challenges to access affordable propylene supplies and securing 2-*

ethylhexanol (2-EH) availability. For instance, DINP producers have been able to lower prices because butane-based isononanol (INA), from which DINP is produced, has lower conversion costs than propylene-based 2-EH, the feedstock for DOP. Back integration in the European DOP market has therefore become a critical competitive advantage" (ICIS, 2011c).

- *"The European DINP market is described as balanced to long. Weakening demand, due to year-end destocking and economic uncertainty, has given buyers stronger bargaining power in price negotiations."* (ICIS, 2012)

Regarding the DEHP market, ICIS reports: "*Although European DOP capacity is estimated at 200,000 tonnes/year, consumption has been reduced year on year to just above 100,000 tonnes/year, according to market sources."(ICIS, 2011c)*

And furthermore January 2012: *"Regulatory pressure coupled with depressed economic conditions have shrunk European demand for DOP, forcing producers to run their facilities at reduced rates of 50-60% in an effort to stabilise the market, prevent further price decreases and ease pressure on margins, which are almost at production costs levels".* (ICIS, 2012)

A price of DEHP of 1,500 €/tonne will be used for the cost curves as representing an average for 2011.

Information from manufacturers for this study

Effective prices of alternatives as compared with the price of DEHP are indicated in the Table 3.1 below.

According to Lanxess, the substitution factors may typically vary by less than ±5%. The factor varies with the specific processing conditions, but it is not possible to indicate some general differences between the different processing types (e.g. plastisol processing vs. calendering). The prices of the alternatives are considered confidential by the company. Maag et al. (2009) indicates a price of +75% for ASE, but do not provide prices for the other alternatives.

ExxonMobil indicates that the substitution factor is also depending on the concentration of the phthalates in the material.

As noted in section 2.2 , increasing the amount of plasticiser implies in many cases that less PVC resin is used in order to obtain the same volume of final PVC plastic or ixture. It has been noted by ExxonMobil that the indicated effective price differences do not fully reflect the total differences in raw material consumption for the production of a given volume of plasticised PVC. However, data have not been available for estimating the decrease in costs of other raw materials.

The price of Hexamoll DINCH is indicated to +50% by a downstream user. This is higher than the +30%indicated by TURI (2006) on the basis of experience from the USA. The latter is older but may have a broader basis.

Table 3.1 Price of alternatives as compared with DEHP for use in PVC

DEHP. Market price, September 2011: 1,470-1,520 €/tonne (1,500 €/tonne used for calculations) 1)

Alternative	**CAS No**	**Price compared to DEHP**	**Substitution factor, %**	**Effective price compared to DEHP**	**Source of information**
DINP (Jayflex™ DINP)	68515-48-0	+13-16%	up to 106 2)	+13-20%	ExxonMobil, manufacturer of alternative / ICIS pricing
DIDP (Jayflex™ DIDP)	68515-49-1	+13-16%	up to 110 2)	+13-24%	-"-
DINP	68515-48-0	+5%	107	+12%	DSU, extrusion and injection moulding PVC
DINP	68515-48-0	+15%	106	+18%	DSU, extrusion PVC
DIDP	68515-49-1	+5%	110	+16%	-"-
Hexamoll® DINCH Di-isononyl-cyclohexane-1,2-dicarboxylate,	166412-78-8	+50%	107	+ 61%	-"-
DEHT, DOTP Di(2-ethylhexyl) terephthalate	6422-86-2	+10%	107	+18%	-"-
DEHT, DOTP 1,4- Di(2-ethylhexyl) terephthalate	6422-86-2	+15%	100-103	+15-18%	Eastman, manufacturer of alternative
Citroflex® A-4 Acetyl Tributyl Citrate,	77-90-7	+50-100%	100	+50-100%	Vertellus, manufacturer of alternative
Citroflex® n-Butyryltri-n-hexyl citrate	82469-79-2	+>50-100%	not indicated	+>50-100%	Vertellus, manufacturer of alternative
Mesamoll® (ASE) Sulfonic acids, C10 – C18-alkane, phenylesters,	70775-94-9	not indicated [+75% 4)]	not indicated	not indicated	Lanxess, manufacturer of alternative
Unimoll AGF® Multi-constituente substance - mixture of acylated glycerides,	mixture	not indicated	not indicated	not indicated	-"-
DOA Di-2-ethylhexyl adipate, Adimoll® DO	103-23-1	3)	95	3)	-"-
ODS n-Octyl n-decyl succinate mixture, Uniplex® LXS TP ODS)	mixture	3)	100	3)	-"-
BEHS Benzyl-2ethylhexyl succinate mixture, Uniplex® LXS TP BEHS	mixture	3)	95	3)	-"-

1) Prices provided by ICIC Website with reference to 9 September 2011.

2) The substitution factor depends on the concentration of phthalates in the material. The 106% and 110% represent the typical situation e.g. in cable, film and sheet, but it may be less for some applications.

3) Price reported, but considered confidential.

4) Price difference indicated by Maag et al., 2009.

3.1.2 Experience with substitution

The experience with substitution of DEHP by product group, as reported by the manufacturers of the alternatives, is shown in Table 3.2.

Table 3.2 Experience with substitution of DEHP by product group as reported by the manufacturers; see definition of scores used in notes

Application	**DINP**	**DIDP**	**DEHT/ DOTP 2)**	**Citroflex ® A-4**	**ASE**	**DOA**	**ODS**
	ExxonMobil		**Eastman**	**Vertellus**	**Lanxess**		
Calendering of film, sheet and coated products 1)	1	1		3	2	2	
Calendering of flooring and roofing 1)	1	1			4		4
Extrusion of hose and profile 1)	1	1		3	2	2	
Extrusion of wire and cable	1	1	3		2	2	
Extrusion of miscellaneous products from compounds	1	1		2	2	2	
Injection moulding of footwear and miscellaneous	1	1			?	2	
Slush/rotational moulding 1)	1				?		
Spread coating of flooring 1)	1				2		
Spread coating of coated fabric, wall covering, coil coating, etc. 1)	1	1	1		2	2	4
Car undercoating 1)	1	1			2		4
Non-PVC polymer applications (acrylics)	1		2		?	2	
Adhesives/sealant (e.g. PU), rubber	1		2	2	2	1	
Lacquers and paint			2		2	2	
Printing ink			1	2	2	1	

Notation used: 1) main alternative on market; 2) Significant market experience, 3) Some examples of full scale experience, 4) Pilot/lab scale experience

1) According to ExxonMobil, DEHP is no longer used in most of those end-uses but has been replaced by high phthalates (DINP and DIDP). However this may not be true when considering the use of DEHP in Eastern Europe.

2) The manufacturer Eastman has indicated for this study a relatively small number of applications where they have experience in substituting DEHT for DEHP. According to the company, DEHT has more typically been used for substitution of DINP and DEHT can technically replace both DEHP and DINP in all flexible PVC products.

Ortho-phthalate alternatives

It is well known that DINP, DIDP and DPHP in Europe have substituted for DEHP for most applications. DINP, DIDP and DPHP are by manufacturers indicated as the main alternatives for all PVC applications whereas other alternatives are indicated for lacquer and paints. The primary application areas of the two substances differ somewhat, but as the effective price is nearly the same it will have a very limited effect on the cost curves. Due to DIDP's and DPHP's properties of volatility resistance, heat stability and electric insulation, they are typically used as a plasticiser for electrical cords, synthetic leather for car interiors, and PVC flooring. DINP can be blended into a paste (so-called "plastisol"), which makes it particularly fitted for coating (such as tarpaulins, synthetic leather, flooring, wall covering, etc.) and rotational moulding (such as some toys and sporting articles) applications (COWI *et al.*, 2012).

ExxonMobil, manufacturer of DINP and DIDP, states that these phthalates can be used as alternatives for most DEHP applications with similar cost structure and with technical advantages. The need for process adjustment, product redesign or changes in product quality by application area when substituting DINP or DIDP for DEHP are summarised in Table 3.3.

The manufacturer does not indicate the typical costs of R&D, but indicates that market development and capacity investment have shown, for Europe, that the phthalates DINP, DIDP and DPHP can advantageously replace DEHP.

The marked shift from DEHP to the three alternative phthalates is clearly indicated by the market volume data shown in Figure 3.1. The main drivers for the shift have been the classification of DEHP as toxic to reproduction introduced in 2001and the technical advantages of the three alternatives. As shown in the figure, in 2001 the three alternative phthalates already had a higher market share than DEHP.

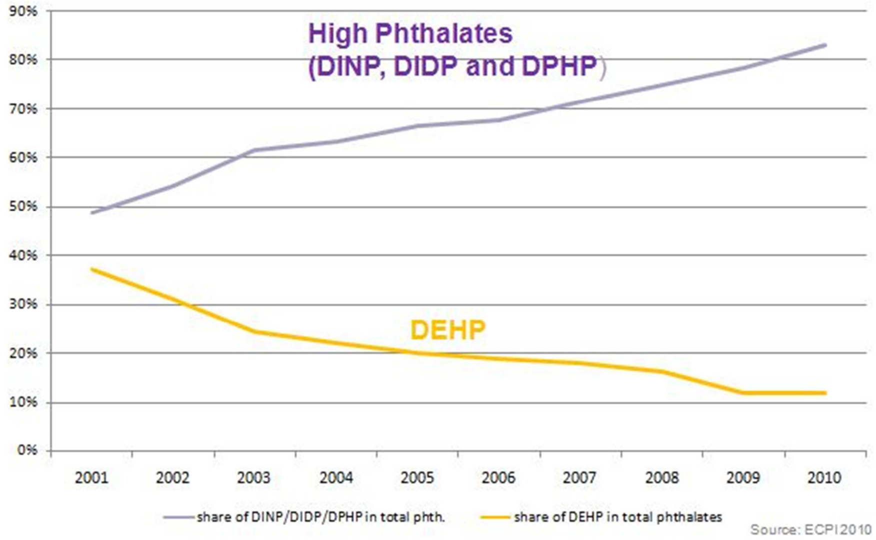

Figure 3.1 Share of DEHP and DINP/DIDP/DPHP of total phthalates sale in Europe (ECPI, 2010)

According the ExxonMobil (2012), the advantages of DINP can be summarised as:

- Better resistance to aging, increasing product life;
- Easier plastisol coating, spraying and dipping;
- Higher permanency and improved resistance for cold outdoor conditions;
- Compatibility with secondary plasticisers for further cost savings;
- Low volatility for reduced process emissions and improved working conditions;
- Lower density and lower energy consumption with higher extrusion outputs compared to DEHP.

In spite of these advantages, data from downstream users indicate that the overall price of the final material as compared with materials with DEHP is +3-5% (Table 3.4). No specific data on the advantages of better resistance to ageing (and thereby increased product life) that allows for an adjustment of the cost estimates have been available. Furthermore, the methodology used, which focus on the short-term impacts on the manufacturers would not take long-term costs born by the consumers into account.

Table 3.3 Important process adjustment, product redesign or changes in product quality when using DINP or DIDP as alternative to DEHP (ExxonMobil)

Application	Product name of main alternative	Important process adjustment	Any product redesign required	Any changes in product quality
Calendering of film, sheet and coated products	DINP,DIDP	No	No - Small formulation adjustments only	Yes, quality improved (better aging, lower process volatility, increased permanency) 1)
Calendering of flooring, roofing, wall covering	DINP,DIDP			-"-
Calendering of film, sheet and coated products	DINP,DIDP			-"-
Extrusion of hose and profile	DINP,DIDP			-"-
Extrusion of wire and cable	DINP,DIDP			Yes, quality improved (better aging, increased permanency 1)
Extrusion of miscellaneous products from compounds	DINP,DIDP			Similar performance, slightly lower abrasion resistance.
Injection moulding of footwear and miscellaneous	DINP,DIDP			Similar performance, slightly lower abrasion resistance.
Slush/rotational moulding	DINP			Similar quality, lower process volatility.
Spread coating of flooring	DINP,DIDP			Yes, quality improved (extraction resistance, lower process volatility, dimensional stability (less curling).
Spread coating of coated fabric, wall covering, coil coating, etc.	DINP,DIDP			Yes, quality improved (better aging, better cold properties, lower process volatility).
Car undercoating	DINP,DIDP			Yes, quality improved (lower process volatility, extraction resistance).
Non-PVC polymer applications	DINP,DIDP			Yes, quality improved (better aging).
Adhesives/sealant, rubber	DINP,DIDP			-"-
Lacquers and paint	DINP,DIDP			-"-
Printing ink	DINP,DIDP			-"-

1) Comments from manufacturer: quality and performance relate to flexible vinyl articles made with PVC resin and DINP / DIDP. The improvements in quality with DINP and DIDP also result in lower potential for exposure, as DINP and DIDP bind tightly within the PVC matrix (Van der Waal's forces and dipole-dipole interactions).

Non-ortho phthalate alternatives

A number of non-ortho phthalate alternatives are marketed for the same applications as DEHP and may be used as alternatives to DEHP for remaining uses.

DEHT, which is a tere-phthalate has in particular in North America been one of the main alternatives to DEHP. DEHT is often referred to as a non-phthalate alternative, but more correctly it is a non-ortho phthalate alternative. The manufacturer Eastman has indicated for this study a relatively small number of applications where they have experience in substituting DEHT for DEHP. According to the company, DEHT has more typically been used for substitution of DINP and DEHT can technically replace both DEHP and DINP in all flexible PVC products.

Eastman indicates that DEHT is a drop-in alternative for DEHP for most applications (indicated in Table 3.2) and no significant costs of R&D and process changes are foreseen. A downstream user indicates the extra price of the DEHT (DOTP) as 10% (Table 3.4).

Lanxess indicates that they believe that ASE and DOA can replace DEHP without any changes to the existing equipment. Additional costs may be incurred by minor one-off reformulating work, the costs of this is indicated as “insignificant” by the manufacturer. The company has indicated that the main part of the R&D will take place by the manufacturer of the alternatives in order to ensure that the plasticiser blend has the desired properties.

Costs of Research and Development

Several manufacturers of alternatives have indicated that a significant part of the R&D is done by the plasticiser manufacturers testing the properties of the plasticisers for the different applications. Some adjustment is, however typically necessary and this is typically done in cooperation between the manufacturer and the downstream user.

A compounder (right column in Table 3.4), who prepares compounds for many downstream users, indicates that the R&D costs is not an essential cost element neither for the compounder nor for the users of the compounds. The R&D is undertaken in cooperation between the compounder who test the compounds in the laboratory and the user of the compounds who test it in the production process and change the process parameters. New compounds have to be developed continuously and much experience with the use of different plasticisers has been build up over the years.

The costs of shifting to the non-phthalate plasticisers may more likely imply higher costs of research and development as well as adjustment/adaptation of manufacturing equipment and conditions due to less experience with the plasticisers. The costs may decrease over the years as result of a larger market for the alternatives.

One of the manufacturers of plasticisers has indicated that R&D has been particularly resource consuming for replacing DEHP (and co-plasticisers) in multilayer flooring manufactured from plastisols where the flooring consist of 4-5 different layers formed on top of each other in a continuous process at high speed. This is a particularly complex application. It has been indicated that the R&D for one product typically has taken several years of two employees of the floor manufacturers working in cooperation with the manufacturer of the plasticiser. The costs are not indicated, but may well be in the order of several 100,000 €. This information is somehow contradictory to the information that DINP or DEHP may replace DEHP without significant process changes, but the reason may be the high complexity of this type of products. According to the manufacturer of plasticisers all producers of flooring in Western Europe today have changed to alternative plasticisers.

Data obtained from five downstream users are summarised in Table 3.4 and Table 3.5.

The downstream users cover extrusion, injection moulding, calendering and compounding of flexible PVC. The indication of price and substitution factors for replacement of DEHP with DINP or DOTP (DEHT) is, in accordance with the information obtained from the manufacturers of the alternatives, resulting in an effective price of the alternatives of +15% to +24% of the price of DEHP. One DSU indicates costs of R&D while another indicates some costs of separation of process lines (details were not given.

The overall price of the manufactured PVC material is indicated to be in the range of +3% to +10%. Considering that the plasticiser in the examples takes up some 30-50% of the PVC material, the extra price of the material is mainly a consequence of the extra price of the plasticiser, whereas other costs contribute insignificantly to the extra prices of the material. This is in accordance with the information obtained from manufacturers of the alternative plasticisers, which in general indicates that the costs of R&D and changes in equipment are insignificant.

For the cost curves, as described in section 3.2.2, it has been assumed that the downstream users on average would have some costs of R&D and costs of equipment (some would have costs and some not), but the estimates indicates, in accordance with the information provided by manufactures, that the annualised one-off costs are small compared with the incremental operating costs caused by the higher effective price of the alternatives. More detailed data on the one-off costs would consequently not influence the total cost estimates significantly.

One downstream user with experience in replacing the plasticised PVC materials with a thermoplastic elastomer compounds TPE –S and TPE –O has answered that the price of the alternative material was 50% higher than the price of PVC plasticised with DEHP. Alternative materials with similar properties as PVC plasticised with DEHP have not been further investigated as it is for the major application area not considered the least cost alternative.

Maag *et al.* (2010) provides a few examples of replacement of DEHP, DBP, BBP, DINP, DIDP and DnOP in toys and childcare articles. One company identified after some testing three potential replacements for DINP: DEHT, ATBC and DINCH. These could be blended in a variety of combinations to achieve softened PVC that performed to the required standards of safety and reliability. These blends could be used in many cases as one-to-one replacements for DINP so major changes to designs and tooling were not necessary. The costs of the R&D are not indicated. According to another Danish manufacturer of toys, the ban of certain phthalates in toys has resulted in an increase in prises of approximately 10-20% because the alternative substances generally are more expensive (Maag *et al.* 2011).

Table 3.4 Experience of downstream users (DSU) regarding alternatives to DEHP

	DSU, extrusion and injection moulding	**DSU, extrusion**	**DSU, extrusion and calendering**	**DSU, compounding**
Type of material	Soft PVC with 30-50% DEHP	Soft PVC with 35% (30-40%) DEHP	Soft PVC with 27% (0-30%) DEHP	Soft PVC with 40% DEHP (30-50%) and co-plasticiser: DBP, DDP, DIDP, 5% (3-7%)
Alternatives tested	DOTP,DINC, DINP, DIDP	DINP	DPMP, DINP, DIDP, DOTP	DINP, DIDP, DPHP, DEHT, DINCH, ASE, ESBO, Benzoates (underlined used today)
Substitution factor	DOTP=DINCH= DINP = 107% DIDP = 110%	106%	103-105%	105%
Price of alternative compared to DEHP	+5% for DIDP and DINP; +10% for DOTP and +50% for DINCH (plasticiser price) but it depends also from the volume bought	Cheapest alternative is +10-15% more expensive	+ 6-10%	+15% (varies, nearly the same as DEHP for DINP, slightly more for DIDP and DEHT and more for DINCH)
Any process changes required	Increase between 5 and 10% the time to get PVC dry	Extrusion speed 10% faster with alternative	Changes in mixing process (less gelation)	Changes in process temperature (higher gelling temperature for DINCH)
Costs of research and development (R&D) for development of useful article/mixture	4,000 € per mixture	No extra costs	No	Not able to give exacts costs (but not an essential cost element)
Costs of new equipment	None	No extra equipment	25,000 €, separation of production lines	No
Any extra operating costs of the substitution	Cost of production for longer time with higher energy demand: about 10 €/tonne of compound for France	No extra costs	No	Higher energy consumption (due to higher gelling temperature for DINCH). Not able to inform on exact costs
Changes in cost of monitoring and control	None	No extra costs	Additional tests, costs not estimated yet	No
Any changes in product quality	No possibilities to produce very low hardness which requires high plasticiser content	No change	Changes in formulation required	No
Any product redesign required	None except low hardness	No	yes (not specified)	No

	DSU, extrusion and injection moulding	**DSU, extrusion**	**DSU, extrusion and calendering**	**DSU, compounding**
Overall price of final material/mixture compared to material/mixture with the substances concerned	Up to +10%	+5% 1)	+3-5%	+10% (DINP, DIDP, DPHP, DEHT) slightly more with DINCH and significantly more with other non-phthalates

1) The respondent notes that alternative plasticisers have been considered/tested but substitution is not considered feasible, as the use of an alternative (DINP, DINCH, DOTP, etc…) in articles made in Europe is not competitive against articles made outside Europe with DEHP on the European market. The suppression of DEHP in EUROPE will not allow anymore the European articles producers to export articles to countries where DEHP will keep produced and used because of the price difference.

Table 3.5 One downstream user's experience with alternative materials

	DSU, extrusion and injection moulding, PVC
Type of material	Soft PVC with 30-50% DEHP
Name of alternative material	Thermoplastic elastomer compounds TPE –S and TPE –O , Trade name Tefabloc
Chemicals of concern in the material, chemical name and CAS number	No data
Costs of research and development for development of useful article/mixture	20,000 € for simple grades
Any product redesign required	Yes : product specification has to be changed and process as well
Price of alternative material as compared to the price of the phthalate containing material, in %	50% higher
Any changes in product quality (e.g. changes in product life-time or constraints in the use of the products)	Function is met but specification has to be changed
Changes in cost of monitoring and control	None
Other costs of the substitution	Change all processing machine for production of the compound and also some changes in the extrusion line and injection machines at the customers who are producing the articles
Overall price of final material/mixture compared to material/mixture with the substances	Variable

3.1.3 Summary on substitution

The results obtained for DINP and DIDP is in accordance with the information which has been available from other studies, and indicate that the effective extra prices of the plasticisers are in the range of +12-23%. DINP or DIDP have been used as substitutes for DEHP in all types of PVC applications.

No data have been obtained indicating particular difficulties in replacing DEHP for specific applications, and the manufacturers of DEHP have not been in the position to provide information on such applications. One downstream user indicates that DEHP has not been replaced because of the extra costs of using DINP/DIDP, but do not indicate any significant costs beyond the extra price of the plasticiser.

A likely explanation is that the remaining uses do not represent applications where it is particularly difficult to replace DEHP, but rather that DEHP is still used by some producers of formulations and articles of PVC, because their customers do not request the use of alternatives.

For most applications, significant market experience exists in the use of non-phthalate alternatives. For the following applications the information on experience with the use of the non-phthalate alternatives collected from manufacturers for this study is however limited: Slush/rotational moulding and calendering of flooring and roofing. Furthermore, detailed information has not been obtained from the producer of the main alternative, DINCH. For the non-polymer applications (adhesives/sealant, lacquers and paint, and printing ink) non-phthalate plasticisers seem to be main alternatives. The effective price of the least-costs non-phthalate alternatives seems to be higher than the price of the phthalate (ortho and tere) alternatives.

One answer from a downstream user which has replaced flexible PVC with alternative plastic material, confirms the general view that material replacement in general would not be the least-cost solution. However, the alternative materials may be competitive with PVC with non-phthalate alternatives.

In the case of medical equipment, there might be additional costs to comply with regulatory procedures for introducing new materials in medical devices. Medical equipment is included as a subgroup of several of the application areas covered by the study. DEHP is specified by the European Pharmacopoeia as the plasticiser for blood bags. One-off costs of such conversions have not been further investigated and are not reflected in the cost curves.

3.2 Costs curves

3.2.1 Consumption and emission of DEHP

The most recent estimate on the use of DEHP by application areas is presented by COWI *et al.* (2009a) in a report prepared for ECHA. The data concerns 2007.

For that study, data on use of DEHP (as well as BBP and DBP) were requested from all manufacturers of the substance. All seven manufacturers provided data on total manufacture and sale of DEHP, but only three DEHP producers provided data of their sales distribution by process type. The data confirmed the continued usage of DEHP in most of the processes and end-uses mentioned in the EU Risk Assessment Report

(RAR) (ECB, 2008). The dataset does not explicitly confirm the continued use of DEHP for "calendering of film sheet and coated products", "spread coating of flooring", "car undercoating, and "production of ceramics". The first two applications have been large and there are no other indications of their cessation. The applications for cars and ceramics have constituted smaller consumption, and no other information has indicated either their cessation, or their continuation. This is however a very small sample and it does not rule out the continued use of DEHP for these applications. The producer data were not sufficiently representative to conclude that the usage pattern had changed significantly since the RAR inventory (representing 1997), and did not significantly contradict the usage pattern indicated in the RAR. Therefore, the distribution percentages presented in the RAR were used for the calculation of consumption by category by COWI *et al.* (2009), whereas the total was updated on the basis of the information received from all manufacturers.

With the experience that it would be difficult to update the breakdown of the total consumption by application areas without having answers from nearly all manufacturers, no attempt was done in this study to obtain a detailed split. Furthermore, for DBP, BBP and DIBP in any case very specific data would be confidential and could not be presented in this report.

Some information from manufacturers indicate that DEHP may not today be used for same applications, but most information is based in knowledge of the Western European market, whereas very limited information is available on the market for DEHP in Eastern Europe, where DEHP is produced by 2-3 manufacturers.

According to a major manufacturer of phthalates none of the four phthalates are today used in the manufacturing of flooring in Western Europe, but it is known that it is still used in Russia and it is uncertain whether it is still used in Eastern Europe.

For the cost curves, consequently the spilt between applications areas presented in COWI *et al.* (2009a) is used as the best estimate whereas the total is updated on the basis of information from ECPI and the statistics.

According to ECPI (2010), DEHP accounted for about 12% of the phthalates sales in Europe in 2010 corresponding to approx. 120,000 tonnes. According to the statistics from Eurostat (Annex 1), the total consumption of dibutyl and dioctyl orthophthalates in 2010 can be estimated at 147,506 tonnes. If about 10,000-15,000 tonnes are DBP, DIBP (see discussion later) and a small amount is DIOP (diisooctyl phthalate) then about 130,000 tonnes would be DEHP. Assuming a minimal decrease from 2010 to 2011, the total in 2011 is here estimated at 120,000 tonnes.

The number of sites is not know, but it is roughly assumed that the average consumption per site resemble the average presented in the RAR for DEHP (ECB, 2008). No data are available to indicate whether it today could be higher (in case of a general trend toward larger sites) or lower (in case DEHP is mainly used in the smaller sites).

The distribution mainly follows the distribution used in COWI *et al.* (2009a) and the EU RAR which provide data on volumes used for the different processes, however it has for this purpose been considered adequate to group flooring and wallcovering (both used indoors) in one category and roofing (used outdoors) in a separate category irrespective that both groups include articles produced using different processes. The number of sites is roughly estimated using the average for all sites producing flooring wallcovering and roofing (irrespective of the processes). As roofing may be manufactured in smaller quantities at some of the sites also manufacturing flooring and wallcovering (and other articles), the actual number of sites is presumably higher than the number indicated.

Life-cycle emission factors for the total emission to air, soil and water (including wastewater) for each application area is derived from COWI *et al.* (2009a) and multiplied with the total tonnage in 2011. The highest emissions are in general associated with the outdoor uses. Many of the application areas included both indoor and outdoor uses and for those applications the emissions are mainly linked to the outdoor uses, e.g. cables left in the ground (incl. in “Extrusion of wire and cable and misc. products”) and abrasive releases from shoos (included in “Moulding, dip coating”).

The distribution between the different processes and application areas should not be considered to represent the actual situation in 2011, but should be considered the best available distribution scenario for this modelling purpose.

Table 3.6 Scenario tonnage, number of sites and emission of DEHP by application area in 2011

Process/application area	Tonnage	Number of sites	Total emission	
	t/y, 2011	2011	t/y	in % of consumption
Calendering of film/sheet and coated products	18,700	19	69	0.4%
Flooring and wallcovering (calendered and spread coated)	18,700	10	143	0.8%
Extrusion of hoses and profiles	14,800	21	52	0.4%
Extrusion of wire and cable and misc. products	29,700	23	1,600	5.4%
Roofing (calendered and coil coated)	1,500	1	312	20.8%
Spread coating of coated fabric, wall covering, other coil coating, etc.	19,000	29	1,076	5.7%
Car undercoating	1,700	2	196	11.5%
Moulding and dip coating	11,900	12	885	7.4%
Adhesives/sealant	3,000	3	207	6.9%
Other non-polymer use	800	1	189	23.6%
Total (rounded)	120,000	121	4,700	

3.2.2 Assumptions used for the cost curves

Assumptions regarding the cost elements (operating costs and one-off costs) for the DEHP cost curve are summarised in Table 3.7. It is assumed that DEHP is replaced by DINP and DIDP for all PVC applications and by DEHT for the non-PVC applications.

DOA is indicated by the manufacturer as “main alternative on market” for adhesives and sealant and printing ink, and indicated as “significant market experience” for lacquers and paint (Table 3.2), but no specific price data for DOA is available.

The prices of alternatives and substitution factors are based on the information in Table 3.1. As noted in the previous sections it is difficult to quantify any process advantages of using DINP and DIDP. The information from down-stream users on the price of the final PVC material with DINP/DIDP as compared to the material with DEHP indicates that the higher price of the plasticiser result in higher price of the final material and is not counterbalanced by savings due to any better performance of DINP/DIDP.

For most applications both DINP and DEHT can be used as alternatives with nearly the same costs and both are consequently indicated in the table.

All responding market actors indicate that R&D and new equipment are not significant cost elements in the replacement of DEHP with the alternatives concerned. It is notable that none of the manufacturers of DEHP has pointed at particular difficulties in replacing DEHP for particular applications.

However, some of the downstream users, as well as downstream users of DBP/DIBP (Table 5.3) indicate that there might be some small costs of R&D and equipment change. Based on information from a manufacturer of DINP, the costs of R&D for replacement of DEHP (and secondary plasticisers) in the manufacturing of multi-layer flooring have been particularly high. In general it is difficult for down-stream users to precisely quantify the costs of R&D specifically for DEHP replacement, because the development is done simultaneously with other changes in the processes and in the design of the products. The one-off costs for many users of the DEHP will be negligible because the manufacturers of the alternatives have already have much experience in using the alternatives for similar applications, whereas for others it may be necessary to make more tests and changes of process parameters to find the right solution.

One downstream user indicates some costs of separating production lines, probably running one line with DEHP and another with an alternative. In the current scenario it is assumed that all DEHP is replaced, and consequently it is assumed that no extra equipment will be needed.

The costs of R&D indicated in Table 3.7 are very rough estimates of a possible average level. They are generally higher than indicated by any of the downstream users, because the answers concern individual articles and it is has been difficult to interpret the answers in terms of total costs for the entire production site.

All measures are considered to be readily applicable as the alternatives are available on the market. For some non-phthalate alternatives the actual production volumes would be small if all DEHP should be replaced by these alternatives, but for the alternatives considered in this model, no shortages in production volumes are expected.

A summary of the cost curve data for DEHP using least costs alternatives (ortho-phthalates for most applications) is shown in Table 3.8.

Table 3.7 Assumptions regarding cost elements for DEHP cost curve using least costs alternatives (tere- and orthophthalates for all applications)

	DEHP market price in 2011: 1,500 €/tonne							
Process/application area	**Alternative**	**Operating cost elements**			**One-off cost elements, €/site**			
		Price of alternative compared to DEHP	**Substitution factor, %**	**Other operating costs**	**R&D**	**New equip-ment**	**Other one-off costs**	**Amortisation period**
Calendering of film/sheet and coated products	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Flooring and wallcovering (calendered and spread coated)	DINP; DEHT	+15%	106	no significant	150,000	no significant	no significant	5
Extrusion of hoses and profiles	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Extrusion of wire and cable and misc. products	DIDP	+15%	110	no significant	30,000	no significant	no significant	5
Roofing (calendered and coil coated)	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Spread coating of coated fabric, wall covering, other coil coating, etc.	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Car undercoating	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Moulding and dip coating	DINP; DEHT	+15%	106	no significant	30,000	no significant	no significant	5
Adhesives/sealant	DEHT	+15%	103	no significant	50,000	no significant	no significant	5
Other non-polymer use	DEHT	+15%	103	no significant	50,000	no significant	no significant	5

Table 3.8 Summary of cost curve data for DEHP using least costs alternatives

Application area	Total annual single measure costs (€k/y)	Total operating cost (€k/y)	Total one-off costs (€k/y)	Single measure emission reduction (t/y)	Single-measure cost-effectiveness (€k/t); reduced emission	Single measure use reduction (t/y)
Calendering of film/sheet and coated products	6,271	6,143	128	69	90.88	18,700
Flooring and wallcovering (calendered and spread coated)	6,480	6,143	337	143	45.31	18,700
Extrusion of hoses and profiles	5,003	4,862	142	52	96.22	14,800
Extrusion of wire and cable and misc. products	11,961	11,806	155	1,600	7.48	29,700
Roofing (calendered and coil coated)	499	493	7	312	1.60	1,500
Spread coating of coated fabric, wall covering, other coil coating, etc.	6,437	6,242	195	1,076	5.98	19,000
Car undercoating	572	558	13	196	2.92	1,700
Moulding and dip coating	3,990	3,909	81	885	4.51	11,900
Adhesives/sealant	864	830	34	207	4.17	3,000
Other non-polymer use	233	221	11	189	1.23	800
Total	42,310	41,207	1,103	4,729		119,800

3.2.3 Costs curves for least-cost alternatives

The cost curve for the least cost alternatives is presented graphically in tree formats:

- A curve based on <u>emission</u> abated and marginal cost of single measures for emission reduction (Figure 3.2);
- A curve based on the <u>emission</u> remaining and the accumulated costs of reducing the emission (Figure 3.3);
- A curve based on the <u>consumption</u> remaining and the accumulated costs of reducing the consumption (Figure 3.4).

The cost curve focuses on replacement of DEHP and excludes various measures to reduce the emission from the different life-cycle stages.

The cost-effectiveness of reducing the emissions is totally determined by the life-cycle-emission factors, due to the fact that the costs of reducing the consumption of DEHP for the various applications are more or less the same as DEHP can for most applications be replaced by the same alternatives. To illustrate this, a cost curve showing the remaining consumption of DEHP and the cumulative costs of reducing the consumption using least costs alternatives is presented in Figure 3.4. The curve is approximately a straight line.

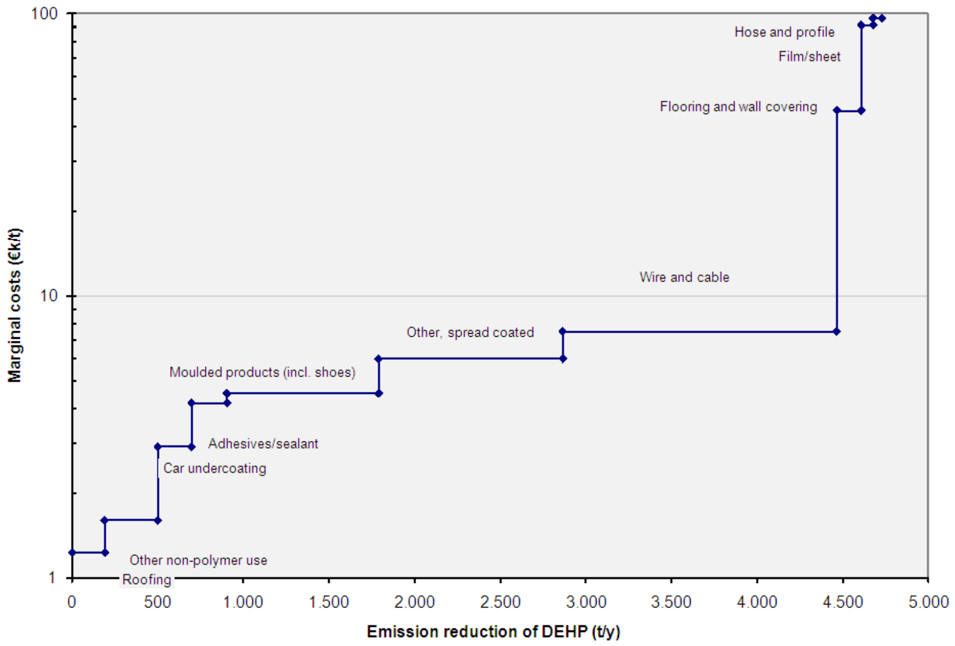

Figure 3.2 Marginal costs of single measures for emission reduction for DEHP using least costs alternatives. Note logarithmic scale

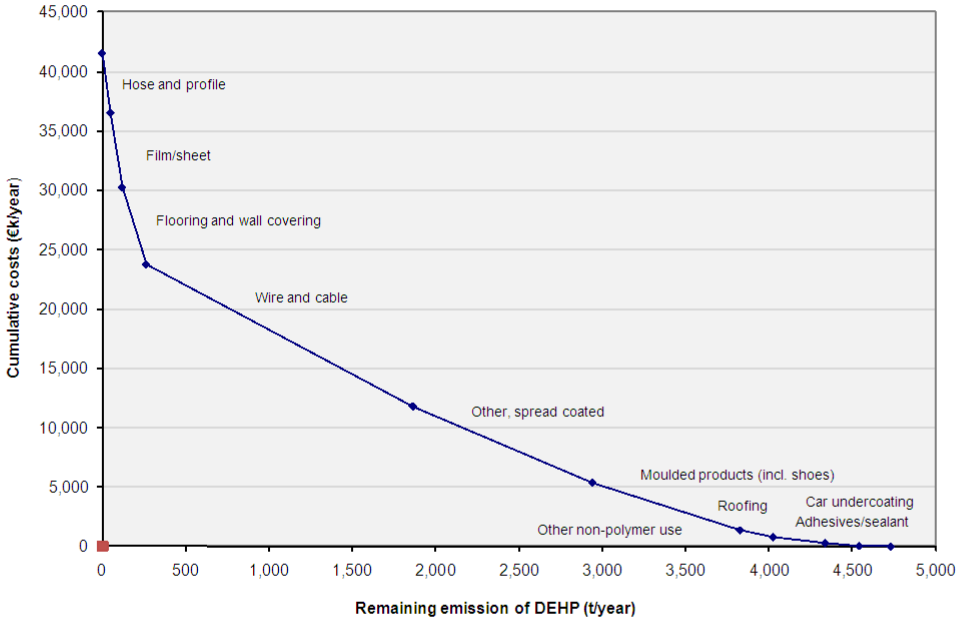

Figure 3.3 *Remaining emissions of DEHP and cumulative costs of reducing the emission using least costs alternatives*

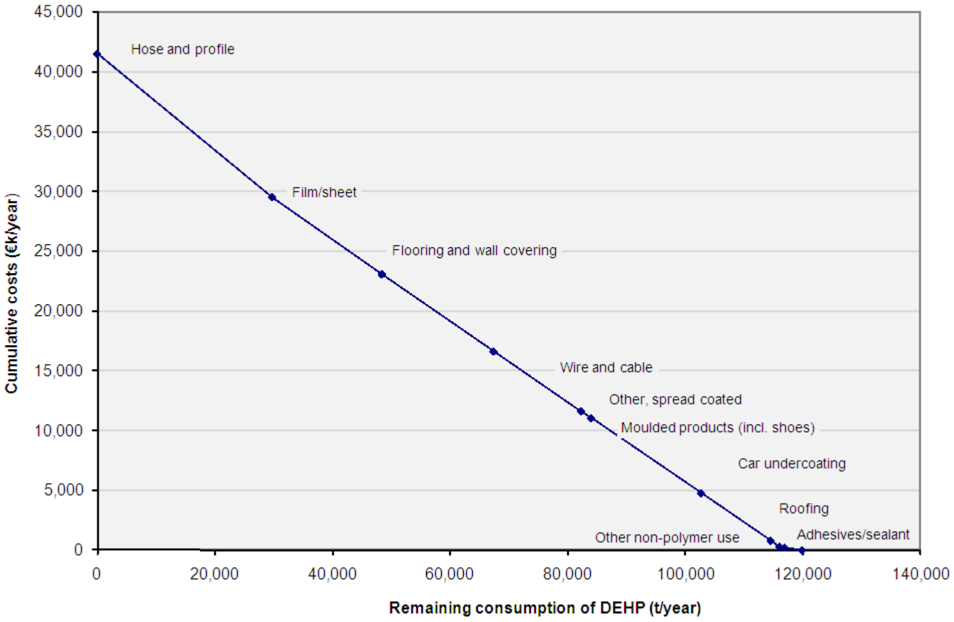

Figure 3.4 *Remaining consumption of DEHP and cumulative costs of reducing the consumption using least costs alternatives*

3.2.4 Costs curves for non-phthalate alternatives scenario

In order to illustrate the differences between different scenarios, a scenario where DEHP is replaced by non-phthalate alternative for all application areas has been developed.

The scenario is purely theoretic, as it is not likely that non-phthalate alternatives would be used as alternatives for the majority of the remaining uses of DEHP. The non-phthalate alternatives have until today mainly been used for sensitive applications with high risk of human exposure such as toys and sports products, food contact materials, water beds and medical applications.

For several of the major applications of PVC, very limited experience with the use of non-phthalate alternatives exist and experience is at pilot/lab scale level (see Table 3.2).

Detailed information on experience have not been obtained for the high volume alternative Hexamoll® DINCH. The manufacturer has recently decided to raise the manufacturing capacity from 100,000 tonnes/year to 200,000 tonnes/year, expected to start up in 2013 (ICIS, 2011d), and the substance is probably the non-phthalate alternative with the highest production volume. The substance is according to the technical data sheets among other applications used for film and sheets (e.g. shower curtains), hoses, sealants, gaskets and shoes. As mentioned, the substance is today mainly used for sensitive applications, but it could in principle also be used for other applications of e.g. other film and sheets and hoses.

According to the information in Table 3.2 significant market experience exists for ASE and DOA for many applications.

The information on prices of alternatives compared with DEHP is relatively uncertain. However, some information exists on the relative costs of materials produced using some of the main alternatives.

In an eco-efficiency analysis, BASF (manufacturer of DINCH) compares various non-ortho phthalate plasticizers for use in PVC applications in Germany (BASF, 2011). The result of the eco-efficiency analysis is beyond the scope of this cost-curve study, but the normalised costs of the different materials is considered to indicated actual costs differences on the German marked. In the analysis, the normalised costs of materials plasticised with ASE and DEHT is indicated to be at the same level in toy balls, while the normalised costs of materials with DINCH and ATBC is indicated as approximately 5% and 10% higher, respectively than the costs of material with DEHT or ASE (the prices are not compared to the price of DEHP). For medical tubing and garden hoses, the costs of material with ASE are in between the costs of materials with DEHT and DINCH. The analysis indicated the differences in the price of the final materials. Assuming that the material contains approximately 30% plasticiser, the difference in material costs would roughly correspond to a relative price of DINCH and ATBC of +15% and +30% compared to the price of DEHT. The price of ASE would on average be about 10% higher than the price of DEHT.

Several sources indicate that the effective price of DEHT is approximately +18% compared to the price of DEHP, and it will consequently be assumed that the effective price of ASE is about 25%. This is in contradiction to information in Maag *et al.* (2009) indicating the price of ASE as compared to DEHP of +75%. The eco-efficiency analysis of BASF is considered to provide better data at this point, and an effective price of ASE in between the price of DEHT and DINCH will be assumed for this study.

If the price of DINCH is 15% higher, the effective price would be about 25% which is close to the +30% previously reported for the US market. The 30% is used as a realistic value.

An analysis of the applicability of the various plasticisers for different applications would require a thorough analysis of factors that will affect the choice of plasticizers for the specific uses. These factors include plasticizer vapour pressure, boiling point, viscosity and diffusion coefficients (ExxonMobil, 2011).

Within the limits of the current study it is assumed that substances that are indicated with “significant market experience” in Table 3.2 may in fact be used for these applications. ASE is indicated as “significant market experience” for a number of applications. ASE is in the literature is indicated as general purpose alternative to DEHP (Maag *et al.*, 2009).

For the remaining PVC applications, “Calendering of flooring and roofing”, “injection moulding of footwear” and “slush rotational moulding” it is assumed that DINCH can be used as alternative. This assumption is in accordance with the analysis provided by ExxonMobil (2011) which indicates that DINCH is a potential alternative for calendering of bags and suitcases and other articles with films, spread coating of wall paper, injection moulding of footwear and rotomoulding of balls for physical exercises. The analysis of ExxonMobil does not indicate any uses of ASE.

Flooring and wires and cables may be application areas where the non-phthalate alternatives may not fully match the phthalate plasticisers and extensively R&D would be needed in order to indentify the right plasticisers and processing conditions.

Due to the more limited experience with the non-phthalate alternatives as compared to the phthalate alternatives it is roughly assumed that the costs of R&D could be twice the costs estimated for the phthalate alternatives. It should be noted that only an expert estimate based on very limited information. Manufacturers of alternatives indicate that the R&D costs are not significant.

It should be noted that for some applications non-PVC alternatives may be competitive with PVC with non-phthalate alternatives, e.g. in wires and cables where non-PVC alternatives have been introduced. It has been beyond the scope of the current study to compare alternative materials with PVC with non-phthalate alternatives. The summary of the cost curve data for DEHP using least non-phthalate alternatives is shown in Table 3.8.

Table 3.9 Assumptions regarding cost elements for DEHP cost curve using scenario with non-phthalate alternatives for all applications

	DEHP market price in 2011: 1,500 €/tonne						
Process/application area	**Alternative**	**Operating cost elements**		**One-off cost elements, €/site**			
		Effective price of alternative compared to DEHP 2)	**Other operating costs**	**R&D**	**New equip-ment**	**Other one-off costs**	**Amortisation period**
Calendering of film/sheet and coated products	ASE	+25%	no significant	60,000	no significant	no significant	5
Flooring and wallcovering (calendered and spread coated)	ASE;DINCH 1)	+30%	no significant	300,000	no significant	no significant	5
Extrusion of hoses and profiles	ASE	+25%	no significant	60,000	no significant	no significant	5
Extrusion of wire and cable and misc. products	ASE	+25%	no significant	60,000	no significant	no significant	5
Roofing (calendered and coil coated)	ASE;DINCH 1)	+30%	no significant	60,000	no significant	no significant	5
Spread coating of coated fabric, wall covering, other coil coating, etc.	ASE	+25%	no significant	60,000	no significant	no significant	5
Car undercoating	ASE	+25%	no significant	60,000	no significant	no significant	5
Moulding and dip coating	DINCH	+30%	no significant	60,000	no significant	no significant	5
Adhesives/sealant	ASE	+25%	no significant	100,000	no significant	no significant	5
Other non-polymer use	ASE	+25%	no significant	100,000	no significant	no significant	5

1) ASE is indicated as a possible alternative for Spread coating of flooring while DINCH is indicated as a possible alternative for calendered products. The highest effective of the alternatives is used for the costs estimations.

2) Due to data limitations only the effective price of alternatives are indicated.

Table 3.10 Summary of cost curve data for DEHP using DEHP cost curve using scenario with non-phthalate alternatives for all applications

Application area	**Total annual single measure costs (€k/y)**	**Total operating cost (€k/y)**	**Total one-off costs (€k/y)**	**Single measure emission reduction (t/y)**	**Single-measure cost-effectiveness (€k/t); reduced emission**	**Single measure use reduction(t/y)**
Calendering of film/sheet and coated products	7,269	7,013	256	69	105.34	18,700
Flooring and wallcovering (calendered and spread coated)	9,089	8,415	674	143	63.56	18,700
Extrusion of hoses and profiles	8,608	8,325	283	52	165.54	14,800
Extrusion of wire and cable and misc. products	12,116	11,806	310	1,600	7.57	29,700
Roofing (calendered and coil coated)	970	968	2	312	3.11	1,500
Spread coating of coated fabric, wall covering, other coil coating, etc.	7,516	7,125	391	1,076	6.98	19,000
Car undercoating	664	638	27	196	3.39	1,700
Moulding and dip coating	5,382	5,355	27	885	6.08	11,900
Adhesives/sealant	742	675	67	207	3.59	3,000
Other non-polymer use	322	300	22	189	1.71	800
Total	52,678	50,618	2,060	4,729		119,800

The cost curve for the non- phthalate alternative scenario is presented graphically in two formats:

- A curve based on emission abated and marginal cost of single measures for emission reduction (Figure 3.5);
- A curve based on the emission remaining and the accumulated costs of reducing the emission (Figure 3.6);

The cost curve focuses on replacement of DEHP with non-phthalate alternatives and excludes various measures to reduce the emission from the different life-cycle stages.

The shape of the curve is more or less the same as the shape of the curve for least-cost alternatives, but the costs per tonne emission reduction is in general higher.

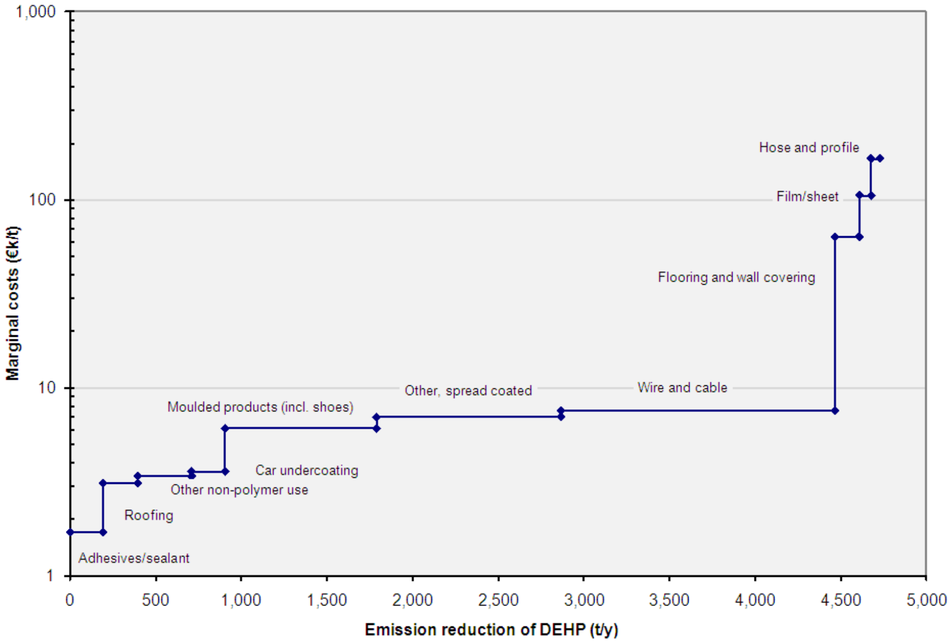

Figure 3.5 *Marginal costs of single measures for emission reduction for DEHP. Scenario with non-phthalate alternatives for all applications. Note logarithmic scale, different from the scale of the corresponding curve for least-cost alternatives.*

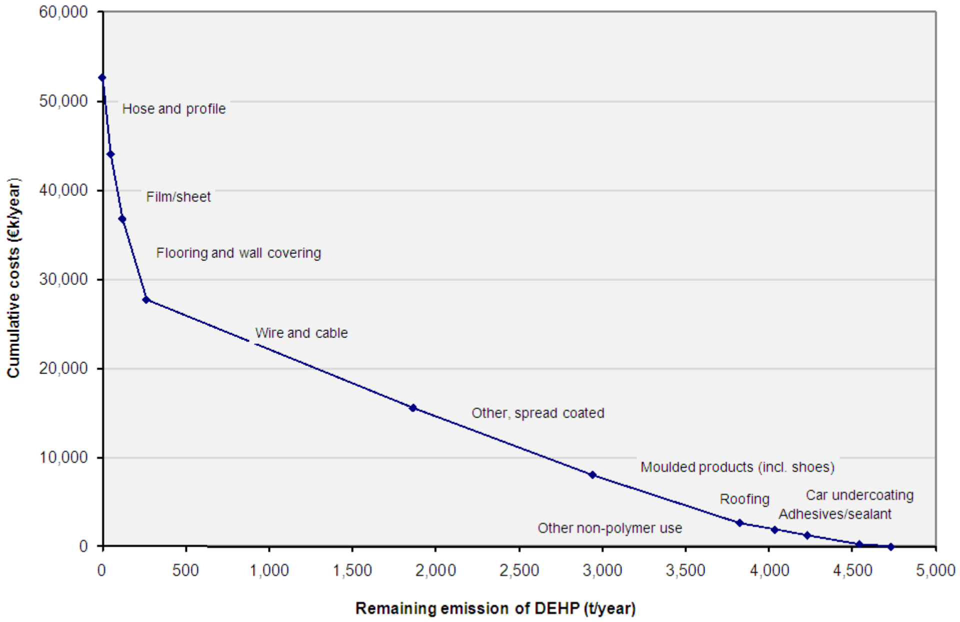

Figure 3.6 *Remaining emissions of DEHP and cumulative costs of reducing the emission. Scenario with non- phthalate alternatives for all applications. Note that the y-axis is different from the axis of the corresponding curve for least-cost alternatives.*

4 BBP

4.1 Use of BBP

Benzylbutylphthalate, BBP is a fast fusing plasticiser (good solvent of the PVC amorphous region), exhibiting lower volatility than DBP or DIBP. It is substantially more volatile than DEHP and exhibits poor low temperature properties. Its high solvency results in poor plastisol shelf life, requiring the need to blend this plasticiser with DEHP or DINP. BBP is (was) used primarily as a fast fusing plasticiser for foamed plastisols and as a plasticiser in polysulfides. It needs to be replaced by other fast fusing plasticisers - for example benzoates or blends of di-benzoates with primary plasticisers like DINP and DIDP (ExxonMobil, 2011).

Market information on BBP by applications area is described in section 4.2.1.

4.1.1 Price information

No information of the price of BBP or alternatives is available from ICIS Pricing.

Prices of alternatives (as compared to the price of BBP) obtained from manufacturers of the alternatives are shown in Table 4.1.

Prices of BBP are considered confidential. In the absence of reported prices it is assumed that the average prices are similar to the prices of DEHP of 1,500 €/tonne for 2011 (for the U.S. market TURI (2006) indicates that the price of BBP is similar to the price of DEHP).

Table 4.1 Price of alternatives as compared with BBP

<table>
<tr><td colspan="6">Assumed average BBP market price in 2011: 1,500 €/tonne</td></tr>
<tr><th>Alternative</th><th>CAS No</th><th>Price compared to BBP</th><th>Substitution factor, %</th><th>Effective price compared to BBP</th><th>Source of information</th></tr>
<tr><td>C7-C9 alkylbenzylphthalate, INBP</td><td>68515-40-2</td><td>+5%</td><td>105</td><td>+10%</td><td>Ferro, manufacturer of alternative</td></tr>
<tr><td>Dipropylene glycol dibenzoate, DGD</td><td>27138-31-4</td><td>+15%</td><td>105</td><td>+21%</td><td>-“-</td></tr>
<tr><td>DINP in blend with isodecyl benzoate (2/3 DINP, 1/3 isodecyl benzoate)</td><td>68515-48-0
120657-54-7</td><td>“same”</td><td>100</td><td>=</td><td>ExxonMobil, manufacturer of alternative</td></tr>
<tr><td>Mixture of DEGD, DGD and TGD 1) , Benzoflex™ 2088</td><td>12055-8
2713831-4
120-56-9</td><td>“N/A due to minimal use of BBP in EMEA” 2)</td><td>100</td><td>not indicated
[equivalent to BBP]3)</td><td>Eastman, manufacturer of alternative</td></tr>
<tr><td>Di-butyl terephthalate, DBT</td><td>1962-75-0</td><td>“N/A due to minimal use of BBP in EMEA” 2)</td><td>“minor reformulation may be required”</td><td>not indicated</td><td>-“-</td></tr>
<tr><td>Diproplyene glycol dibenzoate, Benzoflex™ 9-88, DGD</td><td>27138-31-4</td><td>not indicated</td><td>not indicated</td><td>not indicated
[equivalent to BBP]3</td><td>-“-</td></tr>
<tr><td>Acetyl Tributyl Citrate, Citroflex® A-4</td><td>77-90-7</td><td>+50-100%</td><td>100</td><td>+50-100%</td><td>Vertellus, manufacturer of alternative</td></tr>
<tr><td>Isononyl benzoate, Mesamoll® LP LXS 01 (Blend) ASE/INB</td><td>70775-94-10
Mixture</td><td>4)</td><td>105</td><td>4)</td><td>Lanxess, manufacturer of alternative</td></tr>
<tr><td>Mesamoll TP LXS 51067 (Blend) ASE/GTA</td><td>70775-94-10
102-76-1</td><td>4)</td><td>110</td><td>4)</td><td>-“-</td></tr>
<tr><td>Polypropylene glycol dibenzoate, Uniplex® 400 (PGDB)</td><td>72245-46-6</td><td>not indicated</td><td>not determined</td><td>not determined</td><td>-“-</td></tr>
</table>

1) Reaction mass of diethylene glycol dibenzoate (DEGD), diproplyene glycol dibenzoate (DGD) and triethylene glycol dibenzoate (TGD)

2) The manufacturer states that price difference is not indicated due to minimal us of BBP in EMEA (Europe, Middle East and Africa).

3) In Maag *et al.* (2010) the effective price of Benzoflex™ 2088 is indicated as equivalent to BBP but slightly higher than DEHP.

4) Price reported, but considered confidential.

4.1.2 Experience with substitution

The experience with substitution of BBP by alternative and product group, as reported by the manufacturers of the alternatives, is shown in Table 5.2. The applications indicated as “other” were not included in the questionnaire sent to the manufacturers, but has been added by respondents. Some of the other plasticisers may in fact be used for these applications as well.

BBP is today used by very few companies, which may be the reason why for many of the applications, the alternatives plasticisers are indicated as “Significant market experience” or “Some examples of full scale experience “; in fact the substance may even be the main alternative on market.

ExxonMobil indicates that BBP is not used in calendaring (too volatile/fast fusing) and that BBP is no longer used or in very limited quantities for spread coating of flooring.

Table 4.2 Experience with substitution of BBP by product group

Application	**C7-C9 alkylbenzyl-phthalate, INBP**	**DGD**	**DINP/ isodecyl benzoate**	**Benzoflex™ 2088**	**Benzoflex® 9-88, benzoflex 507**	**Benzoflex® TPU 405**	**Citroflex® A-4**	**ASE/GTA**	**ASE/INB**	**PGDP**
	Ferro		**ExxonMobil**	**Eastman**			**Vertellus**	**Lanxess**		
Flooring	1	Indicated that less experience than with INBP	2	1				3		
Calendered film	2							3		
Spread coated fabric	2			3				3		
Adhesives	3			3					2	
Paint/lacquers	4							2		
Sealants - glass	1			1	1					4
Sealants - construction	1							2		
Coatings and inks 2)					1		3	2		
Other – thermoplastic polyurethane (TPU) applications 1)						2				
Other- Wallcovering 1)								3		
Other - Nail polish 1)							1			

Notation used: 1) Main alternative on market; 2) Significant market experience, 3) Some examples of full scale experience, 4) Pilot/lab scale experience

1) "Other" applications have been added in the individual questionnaire replies by the manufacturers. More of the substances than indicated her may be applicable for these applications.

2) The application area was not included in the questionnaire and data have been derived from Maag *et al.* (2009).

Costs of Research and Development

Scattered information has been provided regarding the possible costs of R&D and changes in equipment.

A manufacturer of two of the alternatives indicates that for all applications no process adjustment or product redesign is needed when substituting C7-C9 alkylbenzylphthalate (INBP) and dipropylene glycol dibenzoate (DGD) for BBP. The manufacturer indicates that there is no major technical constraint when using the alternatives apart from some minor issues with odour and fungus-sensibility of DGD. The manufacturer indicates that costs in addition to extra costs of the plasticiser are minimal.

For the use of the mixture of DINP/isodecyl benzoate in manufacturing of flooring, ExxonMobil indicates that important process modifications are needed as the process would need a fast fusing plastisol. The costs of process chances are not indicated.

Eastman indicates that Benzoflex 2088 or Benzoflex 9-88, are drop-in alternatives to BBP in flooring, spread coated fabric, adhesives, sealants and TPU applications. This means that no process adjustments and product redesign is required, and the alternative has no technical constraints.

Lanxess indicates that process adjustments are necessary when using ASE/GTA, ASE/INB and PGDP but they also note that the effort for reformulation, based on the company’s own experience, will not be significant. Since both ASE/GTA and ASE/INB are offered as formulated blends, they can be processed on the same equipment as BBP. Both products are based on single components with noticeable economies of scale.

No data from downstream users of BBP has been obtained.

4.1.3 Summary on substitution

Manufacturers of both phthalate and non-phthalate alternatives indicates an extra effective price of the plasticiser of 10% and about 20% for phthalate and non-phthalate alternatives, respectively. The alternatives are indicated as drop-in alternatives with no significant costs of research and development, process changes, etc. For flooring, a phthalate/non-phthalate mixture is available at prices comparable to the price of BBP, but major process changes are necessary.

The available information indicates that the extra costs for all applications using least-cost alternatives are due to the extra price of the alternatives.

4.2 Costs curves

4.2.1 Consumption and emission of BBP

The most recent estimate on the use of BBP by application areas is presented by COWI *et al.* (2009c) in a report prepared for ECHA. The data concerns 2007.

For that study, data on use of BBP were requested from the two manufacturers of the substance. Both manufacturers provided data on total manufacture and sale of BBP. Due to confidentiality, the exact manufactured tonnage could not be provided, but the total manufactured volume in 2007 was indicated to be below 18,000 tonnes and of this the consumption for formulation and processing was estimated at about 8,000 tonnes.

Information on the distribution of the BBP supply to the various formulation and processing activities was obtained from the manufacturers of the substance. The data confirmed the continued usage of BBP in most of the processes and end-uses mentioned in the EU Risk Assessment Report (BBP RAR) for BBP (ECB, 2007), but flooring seemed to take up a larger part of the current total in 2007 compared to the data in the BBP RAR.

According to a manufacturer of plasticisers, all producers of flooring in Western Europe today have changed to alternative plasticisers, however less information is available as concern the situation in Eastern Europe.

Considering the confidentiality of the data, no attempt has been done to obtain exact updated data on the consumption of BBP.

According to the Annex XV restriction dossier for DEHP, BBP, DBP and DIBP, a screening of the registration dossiers submitted by producers and importers in 2010 indicates that the aggregated production volume of the BBP, DBP and DIBP in 2009 or 2010 was approximately 20,000 tonnes (ECHA, 2011). Of the 20,000 tonnes produced or imported about 7,000 tonnes was exported. The total consumption of the three substances can consequently be estimated at approximately 13,000 tonnes.

The same dossier estimate the consumption of BBP, DBP and DIBP in 2007 at 8,000, 8,250 and 10,750 tonnes, respectively (ECHA, 2011).

The consumption of the DBP/DIBP in 2006 can on the basis of the statistics be estimated at approximately 32,000 tonnes (Annex 1, Table A1), whereas the consumption figure for 2007 cannot be estimated as the import/export data are confidential. A total consumption of DBP and DIBP in 2007 of 19,000 tonnes as estimated in the Annex XV dossier which, however seems to be underestimating the consumption of the two substances. Under the assumption that the aggregated consumption of DIBP and DBP is more than twice the consumption of BBP, it is here roughly estimated that the total consumption of BBP in 2011 was 4,000 tonnes while the total for BBP/DIBP was 9,000 tonnes.

For the cost curves, consequently the spilt between applications areas presented in COWI *et al.* (2009c) is used as the best estimate whereas the total is updated as described above.

The number of sites is not known, but it is roughly assumed that the average consumption per site resemble the average presented in the BBP RAR (ECB, 2007). No data are available to indicate whether it today could be higher (due to a general trend toward larger sites) or lover (because BBP is mainly used in the smaller sites). It is notable, that the available data indicate that BBP is likely used at 10 sites only across the EU

Life-cycle emission factors for the total emission to air, soil and water (including wastewater) for each application area is derived from COWI *et al.* (2009c) and multiplied with the total tonnage in 2011. Most of the articles are used indoors and compared to the emission factors for DEHP, the emission factors for most applications are relatively small. Sealants are believed to be mainly used for insulating glazing which may be considered an outdoor use, but the sealant in the glazing is not exposed to the weather and the emission from the sealant is considered to resemble releases from indoor uses (COWI *et al.*, 2009c)

The breakdown by application areas should not be considered to represent the actual situation in 2011, but be considered the best available distribution scenario for this modelling purpose.

The highest emission factor is for “other non-polymeric” but the actual applications covered by this product category are confidential.

Table 4.3 Scenario tonnage, number of sites and emission of BBP by application area in 2011

Process	**Tonnage**	**Number of sites**	**Total emission**	
	t/y, 2011	**2011**	**t/y**	**in % of consumption**
Plastisol coating for flooring	1,920	2	64	3.3%
Coating of leather and textiles	400	2	13	3.3%
Calendering of films	280	1	1	0.4%
Processing of hard PVC	320	1	2	0.6%
Processing of sealants	760	1	1	0.1%
Processing of coatings and inks	80	1	6	7.5%
Processing of adhesives	200	1	3	1.5%
Processing of other non-polymeric	40	1	6	15.0%
Total	4,000	10	96	2.4%

4.2.2 Assumptions used for the cost curves

Assumptions regarding the cost elements (operating costs and one-off costs) for the BBP cost curve are summarised in Table 4.4.

The bulk price of BBP has been informed to be 1,600 to 1800 €/tonne in February 2012, and a price of 1,700 €/tonne is used as the best estimate for 2011.

The fact that dibenzoates for many years have been well known and much used competitors to BBP, especially in PVC flooring and in PVA adhesives, indicates a clear potential for substituting the dibenzoates for BBP, from a technical point of view.

The prices of alternatives and substitution factors are based on the information in Table 3.1.

Benzoflex^{TM} 2088 is by the manufacturer indicated as the main alternative for flooring and sealants for glass and benzoflex® 9-88 is indicated as the main alternative in coatings and inks. The manufacturer has not indicated a price as compared to the price of BBP because they consider the consumption of BBP in Europe to be minimal (no price to compare with) and do furthermore not indicate the bulk market price of the plasticisers. In Maag *et al.* (2009c) the effective price of the two plasticisers is indicated as “equivalent to BBP” but not further specified. The effective price of one of the dibenzoates (DGD), supplied by another manufacturer, is reported to be +21% whereas a blend of DINP and isodecyl benzoate is indicated to have an effective price similar to BBP.

For some of the applications in PVC, coating and calendering, the phthalate lNBP has been indicated as main alternative, and this is used in the cost curve even this phthalate may not be the least cost alternative. It is furthermore assumed to be used for two applications where no other information has been obtained.

It is here assumed that the price on the European market of the benzoate and dibenzoate based plasticisers could be +15% the price of BBP and the substitution factor is assumed to be 100%.

One manufacturer of BBP has indicated that no process adjustment or product redesign is needed for all applications. Another manufacturer has indicated that dibenzoates are “drop in” alternatives i.e. R&D and new equipment will not be significant cost elements in the replacement of BBP with the alternatives concerned. Consequently, it is assumed that the one-off costs of substitution are negligible.

As the alternatives already have substituted for the major part of the former use of BBP, much experience exist regarding the performance of the alternatives as replacement plasticisers for BBP. No information indicating that the remaining uses of BBP are applications where it is particularly difficult to replace BBP has been obtained from manufacturers of BBP or manufacturers of alternatives.

All measures are considered to be readily applicable as the alternatives are available on the market and no shortages in production volumes are expected.

A summary of cost curve data for BBP using least costs alternatives is shown in Table 4.5. The available data do not indicate that non-phthalate alternatives are more costly than the phthalate alternatives, and the curve would be quite similar if it included non-phthalate alternatives only.

Table 4.4 Assumptions regarding cost elements for BBP cost curve using least costs alternatives

Assumed BBP market price in 2011: 1,500 €/tonne								
Process/application area	**Alternative**	**Operating cost elements**			**One-off cost elements, €/site**			
		Price of alternative compared to DBP/DIBP	**Substitution factor, %**	**Other operating costs**	**R&D**	**New equip-ment**	**Other one-off costs**	**Amortisation period**
Plastisol coating for flooring	Benzoates and dibenzoates	+10%	100	negligible	negligible	negligible	negligible	-
Coating of leather and textiles	INBP	+5%	105	negligible	negligible	negligible	negligible	-
Calendering of films	INBP	+5%	105	negligible	negligible	negligible	negligible	-
Processing of hard PVC	INBP	+5%	105	negligible	negligible	negligible	negligible	-
Processing of sealants	Benzoates and dibenzoates	+15%	100	negligible	negligible	negligible	negligible	-
Processing of coatings and inks	Benzoates and dibenzoates	+15%	100	negligible	negligible	negligible	negligible	-
Processing of adhesives	Benzoates and dibenzoates	+15%	100	negligible	negligible	negligible	negligible	-
Processing of other non-polymeric	INBP	+5%	100	negligible	negligible	negligible	negligible	-

Table 4.5 Summary of cost curve data for BBP using least costs alternatives

Application area	**Total annual single measure costs (€k/y)**	**Total operating cost (€k/y)**	**Total one-off costs (€k/year)**	**Single measure emission reduction (t/y)**	**Single-measure cost-effectiveness (€k/t); reduced emission**	**Single measure use reduction (t/y)**
Plastisol coating for flooring	432	432	0	64	7	1,920
Coating of leather and textiles	62	62	0	13	5	400
Calendering of films	43	43	0	1	43	280
Processing of hard PVC	49	49	0	2	25	320
Processing of sealants	171	171	0	1	171	760
Processing of coatings and inks	18	18	0	6	3	80
Processing of adhesives	45	45	0	3	15	200
Processing of other non-polymeric	6	6	0	6	1	40
Total	826	826	0	96		4,000

4.2.3 Costs curves

The cost curve is presented graphically in three formats:

- A curve based on emission abated and marginal cost of single measures for emission reduction (Figure 4.2)
- A curve based on the emission remaining and the accumulated costs of reducing the emission (Figure 3.3);
- A curve based on the consumption remaining and the accumulated costs of reducing the consumption (Figure 4.3)

The cost curve focuses on replacement of BBP and excludes various other measures to reduce the emission from the different life-cycle stages.

The cost-effectiveness of reducing the emissions is (like the situation for DEHP) totally determined by the life-cycle emission factors, due to the fact that the costs of reducing the consumption of BBP for the various applications are more or less the same in comparison to the large differences in the emission factors. To illustrate this, a cost curve showing the remaining consumption of BBP and the cumulative costs of reducing the consumption using least costs alternatives is presented in Figure 3.4. The curve is approximately a straight line.

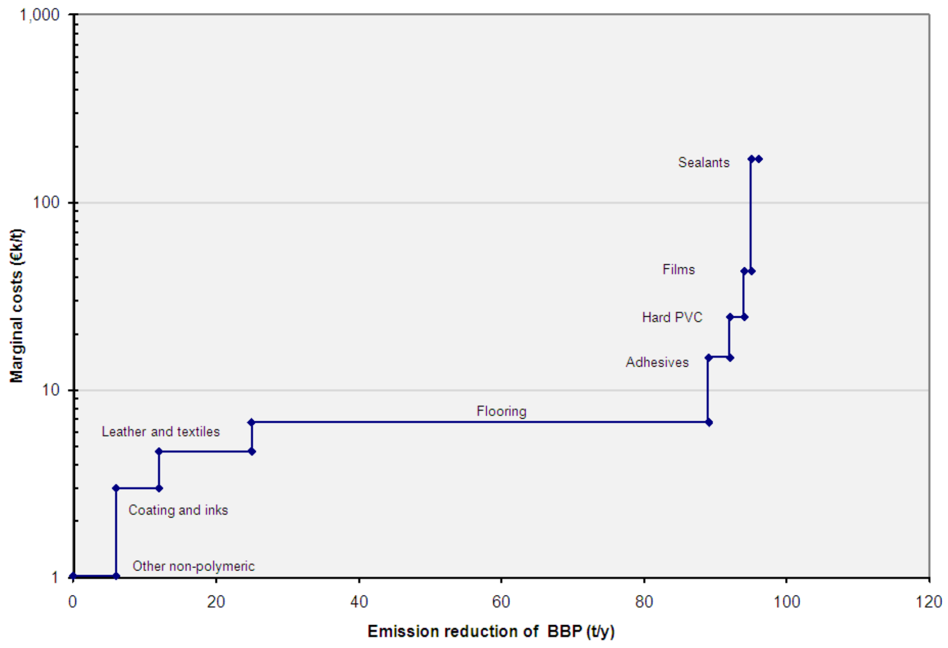

Figure 4.1 Marginal costs of single measures for emission reduction for BBP by using least costs alternatives. Note logarithmic scale.

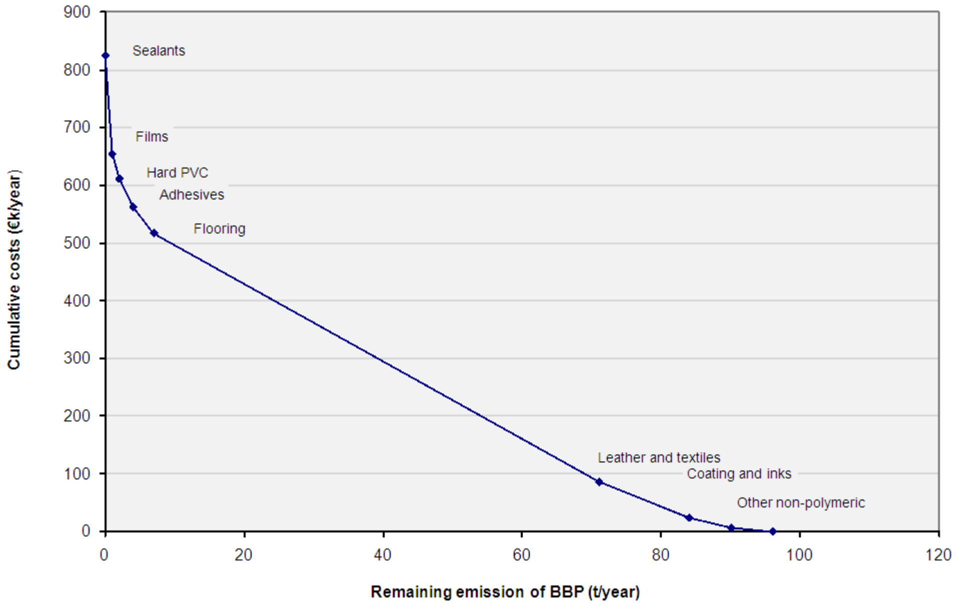

Figure 4.2 Remaining emissions of BBP and cumulative costs of reducing the emission using least costs alternatives.

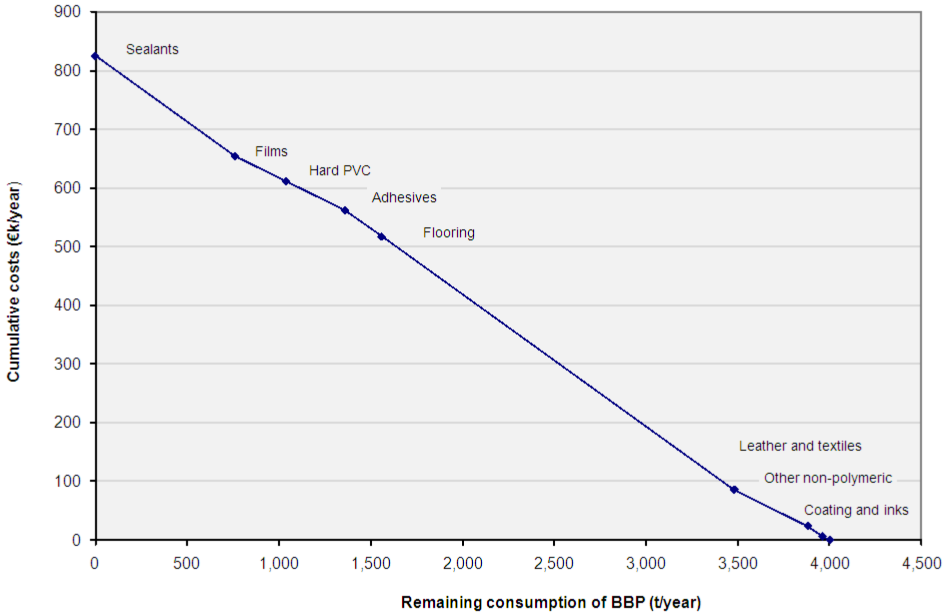

Figure 4.3 Remaining consumption of DBP/DIBP and cumulative costs of reducing the consumption using least costs alternatives

5 DBP and DIBP

5.1 Use of DBP and DIBP

Dibutyl phthalate (DBP) and diisobutyl phthalate (DIBP) exhibit low viscosity and good solvating properties but their extremely high volatility has limited their selection as primary plasticisers for PVC. They are essentially used for their viscosity reducing properties and compatibility with non-PVC (lacquers, printing inks, sealants, adhesives) or as processing aid for PVC (plastisols, compounds) in concentrations of 5 to 10 % w/w due to their higher polarity. (ExxonMobil, 2011).

As processing aid the substances are used as gelling agents which are the agents that reacts fastest with the PVC in the gelation. Gelation is a process in which the plasticiser diffuses into the particles of PVC resin during heating. According to COWI *et al.* (2009b) it was difficult to obtain very specific information on the use of DBP in PVC, but the following applications were mentioned by different sources: floor coverings, automotive uses and garden hoses. Dutch surveys of phthalates and other plasticisers in toys and childcare products demonstrated that 30% of 24 analysed products in 2004 contained DBP (FCPSA, 2008a). The share had decreased to 13% of the products in 2007 and 1% in 2008 (FCPSA, 2008b).

In general, limited specific information is available on the use of DIBP. According to ECHA’s background document for DIBP it may due to similar application properties be used for the same applications as DBP (ECHA, 2009). DIBP may in fact have been promoted as an alternative to DBP. Besides the applications described above the background document mention that DIBP may be used in coatings, e.g. antislip coatings, and in epoxy repair mortars.

5.1.1 Price information

It has not been possible to obtain actual bulk market prices for DBP and DIDP. Market prices are not reported on ICIS Pricing, and manufacturers of DBP and DIBP have not provided any information for this study.

The price of DIBP is indicated to be similar to the price of DBP (COWI *et al.*, 2009 making reference to BASF). Maag *et al.* (2010) indicates that the price of DBP is similar to the price of DEHP. It will consequently here be assumed that the price of both substances in 2011 was 1,500 €/tonne, as estimated for DEHP.

Information on prices and substitution factors of alternatives as compared with DBP and DIBP are shown in Table 5.1.

The benzoates and dibenzoates are indicated to be slightly more costly than DBP/DIBP. 2-ethylhexyl benzoate (EHB) is indicated by a downstream user to be available at an effective price of +10% , isodecyl benzoate at +10-20% and Benzoflex^{TM} 2088 (mixture of dibenzoates) to be “slightly” higher.

The prices of the actual alternatives to DIBP, reported by four manufacturers of adhesives, range from +10-20% for the alternatives with lowest price to more than +80-116% for the alternatives with the highest price. Adhesives are a very diverse group of mixtures with different specific properties and the experience from the downstream users illustrates the difficulties in estimating the most likely costs of substitution.

Table 5.1 Price of alternatives as compared with DBP and DIBP

DBP/DIBP. Assumed market price of DBP and DIBP in 2011: 1,500 €/tonne

Alternative	**CAS No**	**Price compared to DBP/DIBP**	**Substitution factor, %**	**Effective price compared to DBP/DIBP**	**Source of information**
Mixture of DEGD, DGD and TGD (2), Benzoflex™ 2088	12055-8 2713831-4 120-56-9	N/A (1) [slightly higher] (4)	100 "drop in"	[slightly higher] (4)	Eastman, manufacturer of alternatives
Di-butyl terephthalate, DBT	1962-75-0	N/A (1)	"minor reformulation"	N/A	_"_
Isodecyl benzoate	120657-54-7	+10-20%	100	+10-20%	ExxonMobil, manufacturer of alternatives
Acetyl Tributyl Citrate, Citroflex® A-4	77-90-7	+50-100%	100	+50-100%	Vertellus, manufacturer of alternative
Triethyl Citrate, Citroflex® 2	77-93-0	+50-100%	100	+50-100%	Vertellus, manufacturer of alternative
Tributyl Citrate, Citroflex®4	77-94-1	+50-100%	100	+50-100%	Vertellus, manufacturer of alternative
Glycerin triacetate, Triacetin	102-76-1	~150£/t more expensive [~ + 15%] (3)	102-110	+17-27%	British Adhesives and Sealants Association
2-ethylhexyl benzoate	5445-75-7	+10%	100	+10%	_"_
Trimethyl pentanyl diisobutyrate (TXIB)	6846-50-0	>+80%	100-120	> +80-116%	_"_
Citrofol AH II	144-15-0	>+80%	100-120	> +80-116%	_"_
Sulfonic acids, C10 – C18-alkane, phenylesters, Mesamoll® (ASE)	70775-94-10	not indicated	not indicated	not indicated	Lanxess, manufacturer of alternative
Glyceryl triacetate, triacetin (GTA)	102-76-1	(5)	102-110 (6)	(5)	_"_
Adimoll® DB	105-99-7	not indicated	not indicated	not indicated	_"_

1) Respondent indicate that the price difference is not assessed due to minimal use of DBP and DIBP in AMEA (Europe, Middle East and Africa)

2) Reaction mass of diethylene glycol dibenzoate (DEGD), diproplyene glycol dibenzoate (DGD) and triethylene glycol dibenzoate (TGD)

3) The percentage calculated by the authors of this report.

4) COWI *et al.* making reference to Genovique (manufacturer at that time).

5) Price reported, but considered confidential.

6) Indicated by downstream user (Table 5.3)

5.1.2 Experience with substitution

The experience with substitution of DBP and DIBP by main alternative and product group, as reported by the manufacturers of the alternatives, is shown in Table 4.2.

As for BBP, DBP and DIBP is today used by very few companies, which may be the reason that for many of the applications the alternatives plasticisers are indicated as “Significant market experience” or “Some examples of full scale experience “. Actually the substance may in fact be the main plasticiser on market.

One manufacturer indicates in the questionnaire response that DBP and DIBP to their knowledge are not used in vinyl flooring and other PVC applications. This may be true for the Western European market, but may be different for the Eastern European market.

ExxonMobil notes that for other non PVC applications (adhesives, paints,…), formulations are customer specifics as well as the solution implemented. Many alternatives are in play, but typically esters derived from benzoic acid are used (dibenzoates or mono-benzoates). Furthermore, they note that DBP/DIBP can be replaced by a mixture of DINP and isodecyl benzoate in flooring, and by isodecyl benzoate in some other applications.

In the response to the questionnaire, the Association of the European Adhesive & Sealant Industry (FEICA) states that most of the members of the Association have already ceased using the four phthalates years ago and these phthalates are no longer used by the majority of the adhesive and sealant industry. There are, however, a few companies that are still using DBP and DiBP. In this regard, the Association notes that the substances are in the process of being substituted for these uses.

Eastman indicates that DBT can be used as a drop-in alternative without important process adjustment or R&D for all application except paint (no experience with paint).

In order to reduce DIBP in paper and board EU-wide, FEICA in 2008 recommended a voluntary initiative on the reduction of DIBP in adhesive formulation in order to avoid it coming back in to the paper cycle through the recycling process, a potential risk for food contact applications. The companies supplying such materials in Germany have signed a voluntary agreement, and in 2008 they have confirmed a complete phase-out of DIBP in their products. The voluntary agreement has been successfully repeated in several other countries, and recent studies by FEICA have confirmed a significant reduction in the use of DIBP for this application in Europe.

Table 5.2 Experience with substitution of DBP and DIBP by product group

Application	**DBT**	**Benzoflex 2088**	**Isodecyl benzoate**	**Citrates, Citroflex® A4**	**ASE**	**GTA**	**Adimoll DB**
	Eastman	**Eastman**	**ExxonMobil**	**Vertellus**	**Lanxess**		
Polymers formulation and processing	2	1	1)	2	2		
Paints		2	2)	2	2	3	3
Adhesives	2	1	2)	3	2	2	
Grouting agents (sealants and filler)	2		2)	4			
Other, non-polymeric				2	2	3	

Notation used: 1) main alternative on market; 2) Significant market experience, 3) Some examples of full scale experience, 4) Pilot/lab scale experience

1) ExxonMobil answers that to the best of their knowledge DBP and DIBP are no longer used for PVC applications.

2) ExxonMobil answer: many alternatives, but typically esters derived from benzoic acid are used (dibenzoates or mono-benzoates).

Costs of Research and Development

The manufacturer of DBT, Eastman indicates that the substance can be used as a drop-in substitute for DBP and no R&D or important process or product redesign is necessary.

Experience with replacement of DIBP in adhesives by four non-phthalate alternatives have been reported in a questionnaire response by the British Adhesives and Sealants Association (BASA). BASA reports that many plasticiser alternatives to phthalates have been evaluated by members with successful substitutions of DIBP in adhesives being made. Other alternative materials are still being evaluated on the basis of better/more appropriate materials offered by suppliers.

The table below summarises information on experience in substituting DIBP obtained from the British Adhesives & Sealants Association (BASA) through FEICA.

Table 5.3 Experience of members of BASA with the use of alternatives to DIBP in adhesives with a typical content of DIBP of 5-6% (range of 2-15%)

	Triacetin	**2-Ethylhexyl benzoate**	**TXIB / Citrofol AH II**
Price of alternative as compared to the price of the phthalate	~£150/t more expensive plus the extra product added to each formulation	+10%	>80%
Substitution factor	102-110%	100%	100-120
Any process changes required	None	None	Process was changed
Costs of research and development for development of useful article/mixture	£20,000	€30,000	18 months of research (SME status) – real concerns for supply continuity for customers
Costs of new equipment	None	None	Drum stock rather than tank direct delivery. Using more than one plasticiser on plant to achieve requirements
Any extra operating costs of the substitution	None	None	Drum handling costs
Changes in cost of monitoring and control	None	None	Much lower – hazards are minimal.
Any changes in product quality	Marginally; but difficult to assess in a significantly scientific way	None	Products are 'harder' – Shore A hardness has increased from around 40 to around 70
Any product redesign required	Most products had to be 'rebalanced' for the overall formulation to achieve the same adhesive performance. Adhesion was not always the same with the new plasticiser.		YES: Formulations needed other modifications with the replacement plasticisers e.g. for application properties, slump/slip/sag; extrusion rates; curing/drying times etc
Overall price of final material/mixture compared to material/mixture with the substances concerned	Estimated £16,000	2% price increase initially	Approx.: average 18% increase on raw material costs for the product range.

5.1.3 Summary on substitution

Alternatives are available for all applications and it has not been possible to identify applications for which the substitution is particularly difficult. Manufacturers of DBP/DIBP have not provided such data.

The main alternatives are the benzoates and dibenzoates, but many different plasticisers may provide the same properties as DBP/DIBP. The effective price of the least-cost alternatives indicated by both manufacturers and downstream users is approximately +10-20%. Manufacturers of adhesives indicate costs of R&D of about 30,000 € per mixture, but the number of mixtures per site have not been indicated. One manufacturer reports that 18 months of research have been necessary and that the raw material costs for the mixture increased by 18%. The available data indicate that at least in the use for adhesives the costs of R&D have been substantial.

5.2 Costs curves

5.2.1 Consumption and emission of DBP/DIBP

The most recent estimate on the use of DBP by application areas is presented by COWI *et al.* (2009c) in a report prepared for ECHA. The data concerns 2007.

For that study, data on use of DBP were requested from the three manufacturers of the substance. All manufacturers provided data on total manufacture and sale of DBP. Due to confidentiality, the exact manufactured tonnage could not be provided, but the total manufactured volume in 2007 was indicated to be below 10,000 tonnes and the consumption for formulation and processing was estimated at about 8,200 tonnes.

It was in the study not possible to obtain comprehensive quantitative updated information on the use of DBP for the different uses from manufacturers and suppliers and the available information did not allow real estimates of the distribution between the different use areas to be made. It is, however, deemed that the distribution between applications most likely was different from the 1997 distribution used in the EU Risk Assessment Report (RAR) (ECB, 2004), but the updated distribution was highly uncertain.

No data is available on the distribution of the use of DIBP on end uses. Considering that the two substances have similar use profiles it can be assumed that the distribution is similar to DBP, but this is highly uncertain.

Considering the confidentiality of the data, and that the manufacturers for the previous study was not in a position to provide data on the distribution on uses of DBP, no attempt has been done to obtain exact updated data on the consumption and distribution of DBP and DIBP.

The registrations available under “Registered substances” at ECHA’s website indicate the following applications of the two substances (note that information on uses is not indicated for all of the registrations):

- DBP:
    - Solvent in production of maleic anhydride
    - Formulation and use in polymers
    - Formulation and industrial use of DBP as a plasticiser
    - Formulation of DBP in compounds
    - Intermediate for the use in refining catalysts
    - Use as laboratory reagent, analytics, QC
    - DU use as metal working fluid.

- DIBP:
    - Industrial adhesive

- Adhesive for automotive repair
- Use as an intermediate for the use in refining catalysts
- Formulation of DIBP in dry-blends and plastisol formulations
- Polymer processing through compounding, calendering, spread coating, extrusion, injection moulding, low energy manipulations
- Industrial use as intermediate for chemical synthesis.

The registrations confirm that the substances are used for the major application areas indicated in COWI *et al.* (2009b): Polymers formulation and processing and processing of adhesives. The continued use in paints, grouting agents and other non-polymeric is not specifically confirmed, but may be covered by the use category “Formulation and industrial use of DBP as a plasticiser”. The use as intermediate for chemical synthesis was not mentioned in COWI *et al.* (2009b), whereas COWI *et al.* (2009b) mentioned the use as solvent in production of different organic compounds.

According to the Annex XV restriction dossier for DEHP, BBP, DBP and DIBP, a screening of the registration dossiers submitted by producers and importers in 2010 indicates that the aggregated production volume of BBP, DBP and DIBP in 2009 or 2010 was approximately 20,000 tonnes (ECHA, 2011). Of the 20,000 tonnes produced or imported about 7,000 tonnes was exported. The total consumption can consequently be estimated at approximately 13,000 tonnes.

The same dossier estimate the total use of BBP, DBP and DIBP in 2007 at 8,000, 8,250 and 10,750 tonnes, respectively (ECHA, 2011).

The consumption of the DBP/DIBP in 2006 can from the statistics be estimated at approximately 32,000 (Annex 1, Table A1), whereas the consumption figure for 2007 cannot be estimated as the import/export data are confidential. A total consumption of DBP and DIBP in 2007 of 19,000 tonnes is estimated in the Annex XV dossier, however this seems to be underestimating the consumption of the two substances. Under the assumption that the aggregated consumption of DIBP and DBP is more than twice the consumption of BBP, it is roughly estimated that the total consumption of BBP in 2011 was 4,000 tonnes while the total for DBP/DIBP was 9,000 tonnes.

For the cost curves, consequently the split between applications areas presented in COWI *et al.* (2009b) is used as the best estimate whereas the total is updated as described above.

The number of users by application area is not known, and not indicated in the RAR for DBT (ECB, 2004) or COWI *et al.* (2009b). Based on information obtained from the major supplier of DBP in Europe, COWI *et al.* (2009b) estimated that in total 50-100 major primary users of DBP existed in 2009. Furthermore, some minor users, supplied with DBP from 10-20 suppliers of the substance, may exist, but it was not possible to estimate the number of these minor users. If the number of major users from COWI *et al.* (2009b) is used, it can be estimated that the average consumption per user (site) is approximately 110 tonnes. In the absence of more specific information, this average is used for an estimate of the total number of users in 2011. The number of sites for grouting agents and paint is by this method estimated at 1 for each application area, and could as well be zero, as no specific information confirmed the continued use for this application area is obtained. Compared to BBP, the number of users is relatively high.

Life-cycle emission factors for the total emission to air, soil and water (including wastewater) for each application area is derived from COWI *et al.* (2009b) and multiplied with the total tonnage in 2011. The end-product uses of DBP containing polymers are not well known. The DBT RAR (ECB, 2004) assumes in a worst case scenario that all the polymers are used outdoors. The available information of the uses, however, indicates that indoor applications may account for a significant tonnage. In the absence of more specific data, COWI *et al.* (2009b) considered that a 50/50 % split between indoor and outdoor polymer applications is more likely. As consequence of the large share of outdoor applications, the life-cycle emission factors are compared to the factors for BBP relatively high, but it should be noted that the emission factors are extremely uncertain. COWI *et al.* (2009b) indicates that the releases to water and soil from the end-product use of adhesives are considered small and not further estimated, but indicates in the tables (apparently mistakenly) some releases to water and soil. Consequently, only the estimated releases to the air from the use in adhesives are included here.

The distribution between the different processes and application areas should not be considered to represent the actual situation in 2011, but be considered the best available distribution scenario for this modelling purpose.

Table 5.4 Scenario tonnage, number of sites and emission of DBP/DIBP by application area in 2011

Process	**Tonnage**	**Number of sites**	**Total emission**	
	t/y, 2011	**2011**	**t/y**	**in % of consumption**
Polymers formulation and processing	6,350	53	250	3.9%
Processing of paint	170	1	34	19.9%
Processing of adhesives	2,060	17	117	5.7%
Processing of grouting agents	90	1	10	11.3%
Processing of other non-polymeric	270	2	31	11.5%
Total (rounded)	9,000	74	443	4.9%

5.2.2 Assumptions used for the cost curves

Assumptions regarding the cost elements (operating costs and one-off costs) for the DBP/DIBP cost curve are summarised in Table 5.5.

The fact that benzoates and dibenzoates for many years have been a well known and much used competitor to DBP/DIBP, especially in PVC and in adhesives, indicates a clear potential for substituting the dibenzoates for DBP from a technical point of view. On the other hand, the experience from downstream users demonstrates that in reality different alternatives are used by the various producers of adhesives.

The prices of alternatives and substitution factors are based on the information in Table 3.1. Various alternatives are used for each application area and some of the alternatives are actually used even though the price is significantly higher than the price of DBP/DIBP. As the average content of DBP/DIBP is relatively small, the price of the final product is (contrary to the situation for DEHP) less dependent on the price of the plasticiser.

Benzoflex^{TM} 2088 is by the manufacturer indicated as the main alternative for polymers and adhesives. The manufacturer has not indicated a price as compared to the price of DPD/DIBP because they do not know of any application of DBP/DIBP in Europe and do not have any indication of the price. In Maag *et al.* (2009c) the effective price of Benzoflex^{TM} 2088 is indicated as “equivalent to BBP” but not further specified. The effective price of isodecyl benzoate is indicated at +10-20% and that of 2-ethylhexyl benzoate at 10-20% (Table 5.1).

It is here assumed that the price on the European market of the benzoate and dibenzoate based plasticisers could be +15% of the price of DBP/DIBP and the substitution factor is assumed to be 100%.

As the alternatives already have substituted for the major part of the former use of DBP/DIBP much experience exist regarding the performance of the alternatives as replacement plasticisers for DBP/DIBP. No information indicating that the remaining uses of DBP/DIBP are applications where it is particularly difficult to replace the substances has been obtained from manufacturers of DBP/DIBP or from manufacturers of alternatives.

All measures are considered to be readily applicable as the alternatives are available on the market and no shortages in production volumes are expected.

A summary of cost curve data for DBP/DIBP using least costs alternatives is shown in Table 5.6. All identified alternatives are non- phthalate alternatives.

Table 5.5 Input assumptions for DBP/DIBP cost curve using least costs alternatives

Assumed DBP/DIBP market price in 2011: 1,500 €/tonne								
Process/application area	**Alternative**	**Operating cost elements**			**One-off cost elements, €/site**			
		Price of alternative compared to DBP/DIBP	**Substitution factor, %**	**Other operating costs**	**R&D**	**New equip-ment**	**Other one-off costs**	**Amortisation period**
Polymers formulation and processing	Benzoates and dibenzoates	+15%	100	insignificant	50,000	insignificant	insignificant	-
Processing of paint	-"-	+15%	100	insignificant	50,000	insignificant	insignificant	-
Processing of adhesives	-"-	+15%	100	insignificant	100,000	insignificant	insignificant	-
Processing of grouting agents	-"-	+15%	100	insignificant	50,000	insignificant	insignificant	-
Processing of other non-polymeric	-"-	+15%	100	insignificant	50,000	insignificant	insignificant	-

Table 5.6 Summary of cost curve data for DBP/DIBP using least costs alternatives

Application area	Total annual single measure costs (€k/y)	Total operating cost (€k/y)	Total one-off costs (€k/year)	Single measure emission reduction (t/y)	Single-measure cost-effectiveness (€k/t); reduced emission	Single measure use reduction (t/y)
Polymers formulation and processing	2,024	1,429	595	250	8	6,350
Processing of paint	49	38	11	34	1	170
Processing of adhesives	845	464	382	117	7	2,060
Processing of grouting agents	31	20	11	10	3	90
Processing of other non-polymeric	83	61	22	31	3	270
Total (rounded)	3,034	2,012	1,022	443		8,940

5.2.3 Costs curves

The cost curve is presented graphically in three formats:

- A curve based on emission abated and marginal cost of single measures for emission reduction (Figure 5.1);
- A curve based on the emission remaining and the accumulated costs of reducing the emission (Figure 5.2);
- A curve based on the consumption remaining and the accumulated costs of reducing the consumption (Figure 5.3).

The cost curve focuses on replacement of DBP/DIBP and excludes various other measures to reduce the emission from the different life-cycle stages.

The cost-effectiveness of reducing the emissions is (like the situation for DEHP and BBP) totally determined by the life-cycle emission factors, due to the fact that the costs of reducing the consumption of DBP/DIBP for the various applications are more or less the same in comparison to the large differences in the emission factors. To illustrate this, a cost curve showing the remaining consumption of DBP/DIBP and the cumulative costs of reducing the consumption using least costs alternatives is presented in Figure 3.4. The curve is approximately a straight line.

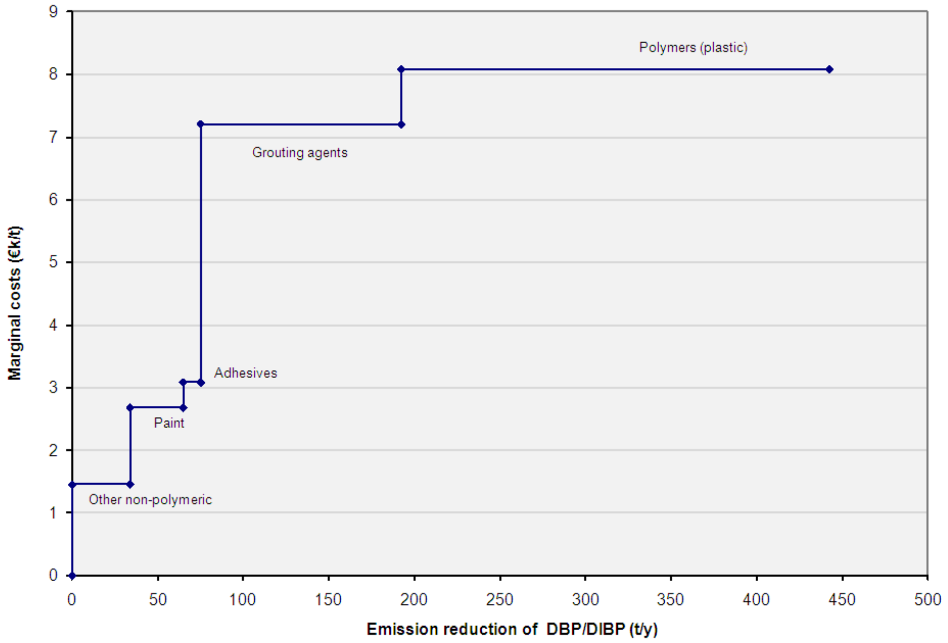

Figure 5.1 *Marginal costs of single measures for emission reduction for DBP/DIBP. Least costs alternatives. Note linear scale*

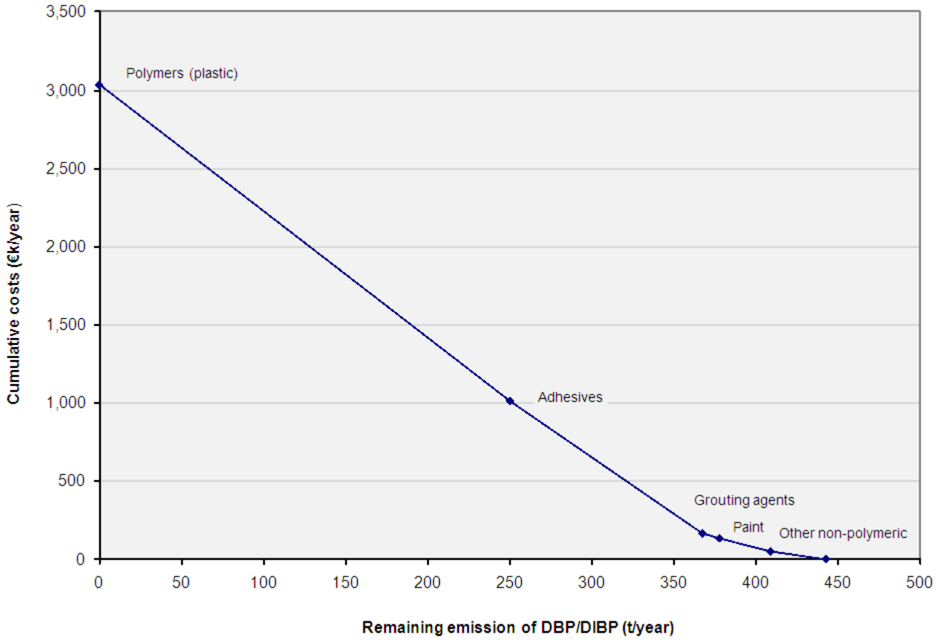

Figure 5.2 *Remaining emissions of DBP/DIBP and cumulative costs of reducing the emission using least costs alternatives.*

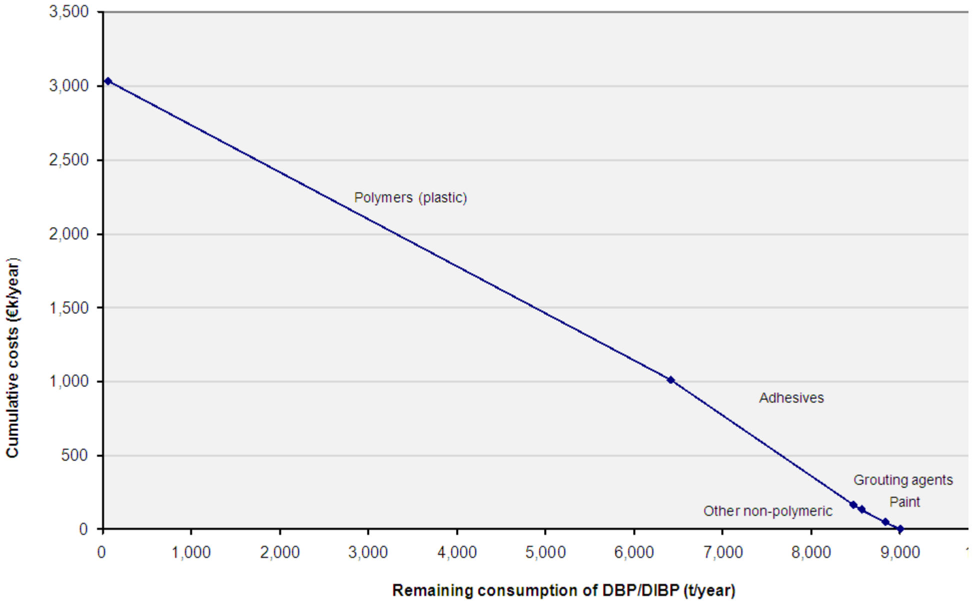

Figure 5.3 *Remaining consumption of DBP/DIBP and cumulative costs of reducing the consumption using least costs alternatives*

6 Conclusion

The project tested a method developed as part of the pilot study "Abatement cost curves for substances of concern" (Corden *et al.*, 2011). This study explored how marginal or average abatement costs of measures could be used to assess the cost-effectiveness of phasing out the use of a chemical substance or reduce any consequent emissions from its lifecycle.

6.1 Key results

The study has developed cost curves to illustrate the costs and relative emission reductions of substituting alternative plasticisers for the phthalates DEHP, BBP, DBP and DIBP. The study has focused on substance substitution and not included other measures for reducing the emissions of the substances.

The total costs of reducing the emissions are summarised in the table below.

Table 6.1 Total potential emission and consumption reduction compared to the 2011 level and estimated costs, best estimate

Substance	Total emission reduction (t/y)	Total consumption reduction (t/y)	Total costs (€m/year)
DEHP	4,729	119,800	41.5
BBP	96	4,000	0.8
DBP/DIBP	443	8,940	3.0

For DEHP and BBP the total costs of the reduction in the use and emissions are estimated to be quite certain when considering the least cost alternatives, whereas for DBP/DIBP the applicability of the least-cost alternatives for the different applications are more uncertain.

For the DEHP and BBP extensive experience exist in replacing the two phthalates with alternative phthalates (both ortho and tere-phthalates) and the uncertainties on the costs of substitution is mainly due to variations in the differences in prices of the alternative phthalates as compared to DEHP and BBP. Market information from ICIS pricing illustrates how the differences between e.g. DEHP and DINP fluctuates. As an example, the price difference increased within one and a half year from 7% in March 2010 to 15% in September 2011. Such fluctuations are inherent uncertainties in the estimations of price and costs differences. The price differences used for the costs curve are considered the average for 2011 and the uncertainty on the costs estimate is deemed to be significantly below ±50%.

The total costs of a scenario where it is assumed that non-phthalate alternatives are used as substitutes for DEHP is estimated at 53 €m/year. The estimate is very uncertain but illustrates that the total costs may be significantly higher if these alternatives are used. For many of the major application areas, very limited (if any) experience in the use of non-phthalate alternatives exist and it is in the current situation not likely that the non-phthalate alternatives would be used as main alternatives to DEHP for these applications.

The study shows that the costs curves and ranking of the cost-efficiency of the measures are nearly 100% determined by the applied life-cycle emission factors for the different application areas. The four phthalates are for all application areas used as plasticisers, and the differences in the costs of substitution of one tonne of the phthalates for the different application areas is small compared to the large differences in the life-cycle emission factors. The ranking of cost efficiency is consequently correlated with the ranking of emission factors for these four substances.

All available information indicates that the main cost element of the substitution is the difference in price between the phthalates and the alternatives. Limited information is available about the one-off costs and other operating costs than the change in costs of plasticiser, but the available information indicates that the contribution from other costs elements in any case is small. The one-off costs vary by company, but apart from information that the costs of R&D have been relatively high for multi-layer flooring because of the complexity of the processing, there is no indication that the one-off costs would be particularly high for some applications. More likely the variation within each application area is higher than the variation between application areas.

For DEHP and BBP alternatives exist which can act as drop-in substitutes for nearly all applications and the costs of substitution of one tonne of the phthalates is largely the same for all application areas. A cost curve showing the costs of reducing the consumption using least costs alternatives is consequently nearly a straight line. For DBP/DIBP the substitution pattern is more diverse with many competing alternatives, and it is more uncertain if the costs of substitution would be the same for all application areas. However, also for the costs of substitution the variation within the application areas would likely be higher than the variation between the application areas.

For a prioritisation of measures the cost curves for these four substances provide limited information beyond what would be available form comparing the life-cycle emission factors.

6.2 Key limitations and uncertainties

As noted in the pilot study (Corden *et al.*, 2011), there are a number of limitations and uncertainties with the method in general. Cost curves provide relatively crude indications of the likely implications of an intervention to reduce emissions. They do not indicate the wider implications of measures, such as affordability or macroeconomic effects, which are typically included in impact assessments and other such analyses. Decision-making on new policies should take into account such factors.

Besides these general limitations the present study had some more specific limitations and uncertainties.

The cost curves were mainly determined by three factors:

- The life-cycle emission factors;
- The consumption by application area;
- The difference in effective price between the four phthalates and the alternatives.

Life-cycle emission factors. The fact that the life-cycle emission factors were the determining factors for the shape of the cost curves and the ranking of the measures is not in itself a limitation of the method. However, it limits how much new information for the prioritisation of the measures the curves provide. The emission factors derived from COWI *et al.* (2009a,b,c) are mainly based on the EU Risk Assessment Report and Emission Scenario Documents (ESC). These factors are in general worst case emission factors and in particularly the emission from processes may likely be lower today. Furthermore the total emissions are represented as one figure aggregating indoor and outdoor emissions to the compartments air, water and soil. Consequently, an emission to the air from indoor use of flooring is compared with abrasive emissions (pieces of material) released to soil from the use of roofing outdoors. In terms of potential human health and environmental impacts the different emissions are not comparable, and aggregating the different emission must be considered to be somewhat problematic. It is indicated in the pilot project that it would be helpful to further prioritise releases of substances to different environmental compartments in terms of their relative importance for environmental effects. It could be feasible to differentiate between these compartments in terms of the overall level of concern. For example, emissions to water might have a greater environmental impact than emissions to land and abatement of these emissions could thus have a greater environmental benefit. As a first step, releases to different environmental media could be given a relative ranking.

With the data available for the current project it is not considered feasible to differentiate between the different compartments. It would require that a ranking system was developed for releases to the different compartments.

Furthermore, the emission factors are quite uncertain and dependent on the methodology used for the estimations e.g. the Risk Assessment Report for DEHP includes abrasive emissions from outdoor use while the reports for DBP and BBP do not (somewhat adjusted in COWI *et al.* (2009a,b,c). An update of all emission factors, would be the most effective way of reducing the uncertainty, however this is estimated to be an extensive task as actual measurements of emissions of the substances are limited.

Consumption by application area. The total consumption in the EU of each of the substances (DBP and DIBP as one aggregated figure) is estimated to be quite certain and the uncertainty of the total volume have a small influence in the total uncertainty of the study. As the four substances, for each of the application areas, only take up a small percentage of the market, estimates on the volume per application area is however very uncertain. The estimates have to be based on information obtained directly from the manufacturers of the substances. It cannot be based on information from downstream users; information from e.g. 5 downstream users within an application area cannot be used to extrapolate the entire market for this area. Due to the limited number of manufacturers, the breakdown by application area for BBP, DBP and DIBP is in any case confidential and only a scenario can be presented. For DEHP, based on the experience from a previous study (COWI *et al.*, 2009a), it was not considered realistic to obtain detailed information on the breakdown from all manufacturers. As the cost efficiency of substituting one tonne of the substances is nearly the same for all application areas, the total costs are not influenced by the uncertainties in the split between the application areas, but the shape of the costs curve is.

Difference in effective price between the four phthalates and the alternatives. Data have been obtained from many manufacturers of alternatives. For DEHP and BBP, the indication of differences in prices for the least costs alternatives are quite certain as the manufacturers of the alternatives have some experience with the uses of the two phthalates and the differences in market prices. For DBP/DIBP several of the manufacturers of alternatives had no knowledge on the market of the two phthalates, because the use of these substances is quite limited in Western Europe. Consequently they were not able to provide an indication of differences. In the scenarios, it is assumed that the users changes to the least-costs alternatives. For DEHP and BBP, the least-costs alternatives are also the main alternatives and this assumption is justified. For DBP/DIBP it is more uncertain whether the least-cost alternatives would in fact be the choice when the remaining uses are replaced. In PVC, DBP/DIBP is used as processing aid in small quantities in many different applications. The substances are used as secondary plasticisers and would typically be replaced together with the primary plasticiser. The costs of the primary plasticiser would be the main determinant for the total incremental costs of the substitution and many factors may influence which plasticisers would be the most feasible alternatives.

One-off costs. Manufacturers of alternatives and the responding downstream users in general indicate that the one-off costs (e.g. research and development and change in equipment) do not influence on the total costs of substitution. Responses from downstream users indicate that even the costs elements differs among the different downstream uses (some indicate R&D, some equipment, etc.). The actual level of R&D is quite uncertain, but it is very certain that these costs have very small influence on the total costs of substitution. The uncertainty on the one-off costs consequently contributes very little to the uncertainty on the total costs.

Limitation in measures included. The result of the study is limited to indicating the costs of reducing the emissions by use of substitution measures. Some other measures (e.g. emission control in production facilities) may be more efficient for reducing some of the emissions of the substances.

Of the total emissions of DEHP to air, soil and water in 2007, releases from manufacture of the substance, formulation and processing was estimated to be about 7%. As mentioned in section 2.2.1 waste gas treatment for manufacture of DEHP by coolant condensation was the most cost-effective of all measures assessed whereas additional advanced (tertiary) water treatment for formulation and processing by membrane filtration was the least cost-effective of all the measures. The cost-effectiveness of the other two measures was of the same magnitude as the substitution measures.

Similarly, some reduction of the emission of BBP and DBP/DIBP may be obtained by emission control in production facilities.

Baseline scenario. The baseline scenario is assumed to be a “business-as-usual” scenario i.e. the consumption continues at the same level as the reference year. In fact, some of the current uses of the substances are in a process of being phased out under the present legislative regime, and the future authorisation procedure would probably result in further reduction of the releases of the substances from production processes. Estimating the potential emission reduction by the authorisation procedure would require a very extensive study and a close cooperation with a large number of users of the substances today.

6.3 Suggestions for further work

For the reasons mentioned above, the methodology provided limited new information for prioritisation of measures for these four phthalates, and furthermore the results are highly uncertain.

The following has influenced the applicability of the method:

- The analysis has been limited to one type of measure: substitution of the use of the substances as plasticisers;

- The function of the phthalates is for all applications the same (used as plasticisers) with relatively small differences between application areas.

- The phthalates have already been phased out by most users. The remaining uses of the phthalates are not for applications where it is particularly difficult to replace the phthalates i.e. the manufacturers and users of the four phthalate have no incentives for providing specific information on the applications.

- Three of the substances (BBP, DBP and DIBP) are produced by a few manufacturers and market volume data are confidential.

- A major part of the releases of the substances is from the use phase, but emission factors are determined with high uncertainty, and the emission volume provides limited information on the potential impact of the emissions.

- The remaining uses of BBP, DBP and DIBP are limited and many manufacturers of alternatives had no knowledge of the market of the three phthalates i.e. they could not provide information on price differences.

It is estimated that fine-tuning the analysis and decreasing the uncertainty would be an extensive task and require contribution from a large number of actual users of the substances. It would furthermore require an update of all emission factors.

It is considered that the application of the costs curve methodology would be most informative for prioritisation processes for substances with some of the following characteristics:

- The function of the substance for the different application areas is different >> in many case the alternatives would be different for the different application areas.

- The substance is mainly used for application areas where there is a significant consumption for all important application areas >> it is relatively easy to identify the actual users.

- A significant part of the emissions can be controlled by other measures than substitution >> the analysis can compare substitution and other measures for abating the emissions.

- The substance is manufactured by more than two companies >> information on total market volume and market volume by application area is not confidential.

- The substance is persistent or the main part of the emissions are to the same compartment >> the emission volumes can be used as a proxy for the potential human health and environmental effects. If later, a ranking system for the releases to the different compartments is developed, it would be of less importance that many different releases are aggregated.

Considering the characteristics above, examples of substances where the methodology probably would provide a better basis for prioritisations could be mercury (and other heavy metals), lead oxide, dimehylacetamide (DMAC) or 1-methyl-2-pyrrolidone. (NMP).

7 Abbreviations and acronyms

ASE	Alkylsulphonic phenyl ester
ATBC	Acetyltributyl citrate
BASA	British Adhesives and Sealants Association
BBP	Butyl benzyl phthalate
BEHS	Benzyl-2ethylhexyl succinate mixture
CEPE	European Council of producers and importers of paints, printing inks and artists’ colours
DBP	Di-n-butyl phthalate
DBT	Di-butyl terephthalate
DEGD	Diethylene glycol dibenzoate
DEHP	Bis(2-ethylhexyl)phthalate
DEHT	Di(2-ethylhexyl) terephthalate (same as DOTP and DEHTP)
DGD	Dipropylene glycol dibenzoate
DIDP	Diisodecyl phthalate
DINCH	Diisononylcyclohexane dicarboxylate
DINP	Diisononyl phthalate
DIOP	Diisooctyl phthalate
DOA	Di-2-ethylhexyl adipate
DOTP	Di(2-ethylhexyl) terephthalate (same as DEHT)
DPHP	Di(2-propylheptyl) phthalate
ECHA	European Chemicals Agency
ECPI	European Council for Plasticisers and Intermediates
EU	European Union
EHB	2-Ethylhexyl benzoate
EMEA	Europe, Middle East and Africa
ESBO	Epoxidized soy bean oil
EuPC	European Plastic Converters
FEICA	Association of the European Adhesive & Sealant Industry
GTA	Glycerol triacetate
INB	Isononyl benzoate
lNBP	C7-C9 alkylbenzylphthalate
IPPC	Integrated Pollution Prevention and Control
ODS	n-Octyl n-decyl succinate mixture
PGDP	Polypropylene glycol dibenzoate
PU	Polyurethane

PVC	Polyvinylchloride
R&D	Research & development
REACH	Registration, Evaluation, Authorisation and Restriction of Chemical substances (Regulation EC 1907/2006).)
RAR	Risk Assessment Report
TGD	Triethylene glycol dibenzoate
TPE	Thermoplastic elastomer

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Annex 1 Statistics

The production statistic (Prodcom) and the external trade statistics of Eurostat include data on the production and external trade of the phthalates. The grouping of the substances under the different CN8 (combined nomenclature) codes in the external trade statistics and the industrial activity codes (NACE codes) in the production statistics have changed over time and the tables presented below have been prepared by combining data from different statistics.

Until 2007, specific codes were used for dibutyl orthophthalates (mainly DBP and DIBP) and dioctyl orthophthalates (mainly DEHP, but a small part may be DIOP (diisooctyl phthalate)). BBP has never been covered by a specific code, but has been included in “Other esters of orthophthalic acid”. In the external trade statics a specific CN8 code is used for “dinonyl and didecyl orthophthalates” (mainly DIDP and DINP), but in the production statistics they are included in “Other esters of orthophthalic acid”.

Table A.1 shows the market data for dibutyl orthophthalates and dioctyl orthophthalates from 2000 to 2007 for EU27. Please note that production data from 2000-2003 represent the production volume in EU15.

From 2007-2011 the dibutyl and dioctyl orthophthalates have been grouped into one code in the production statistics. Table A.2 shows the production and external trade data for the dibutyl and dioctyl orthophthalates in one group and other phthalates in another group. The decreased production and consumption of the other phthalates from 2006 to 2010 may be due to incomplete reporting from some Member States and is not in accordance with the general market data reported by the industry at the website of the European Council for Plasticisers and Intermediates (ECPI).

Table A.1 *Production, import and export of dibutyl and dioctyl opthophalates in EU 27 according to Eurostat databases.*

Substances	Activity	Codes	2000	2001	2002
			tonnes	tonnes	tonnes
Dibutyl orthophthalates (mainly DBP and DIBP)	Production [1]	2414.3413	43,108 [3]	35,128 [3]	48,873 [3]
	Export [2]	2917.3100	6,418	4,457	6,919
	Import [2]	2917.3100	148	115	49
	Consumption[4]		36,839	30,786	42,004
Dioctyl orthophthalates (mainly DEHP)	Production [1]	2414.3415	394,739 [3]	417,335 [3]	371,903 [3]
	Export [2]	2917.3200	75,509	48,631	39,165
	Import [2]	2917.3200	11,170	8,439	3,487
	Consumption [4]		330,399	377,143	336,226

[1] Source: PRODCOM ANNUAL TOTAL [DS-043409] (NACE Rev 1.1); EU27

[2] Source: EU27 Trade Since 1988 By CN8 [DS-016890]; EU27

[3] Source: PRODCOM ANNUAL TOTAL [DS-043409]; NACE 1.1; EU15 (no data for EU 27)

[4] Consumption = production – export + import

Table A.2 Production, import and export of dibutyl and dioctyl opthophthalates and other esters of orthophthalic acid in EU 27 according to Eurostat databases

	Activity	Codes 6)	2000	2001	2002	2003	2004
Dibutyl and dioctyl orthophthalates	Production	2014 3410	437,847 3)	452,463 3)	420,777 3)	390,669 2)	348,
	Export	2917 3100 2917 3200	81,927 4)	53,089 4)	46,083 4)	43,394	69,9
	Import		11,318 4)	8,554 4)	3,536 4)	1,818	4,57
	Consumption 5)		367,238	407,928	378,230	349,093	283,
Other esters of orthophthalic acid	Production	2014 3420	313,105 3)	281,220 3)	318,197 3)	755,835 2)	802,
	Export	2917 3300 2917 3400 2917 3410 2917 3490	33,427 4)	38,971 4)	60,793 4)	97,735	66,2
	Import		2,163 4)	2,114 4)	3,221 4)	2,749	4,33
	Consumption 5)		281,841	244,362	260,625	660,850	740,
	Total consumption		649,079	652,291	638,855	1,009,943	1,02

1) Source if nothing else mentioned PRODCOM ANNUAL SOLD (NACE Rev. 2.) EU 27 [DS-066341] (include external trade statistics for corresponding CN8 codes from the trade statistics)

2) Source: PRODCOM ANNUAL SOLD (NACE REV 1,1); EU27

3) Source: PRODCOM ANNUAL SOLD (NACE REV 1,1); EU15

4) Source: EU27 Trade Since 1988 By CN8 [DS-016890]; EU27

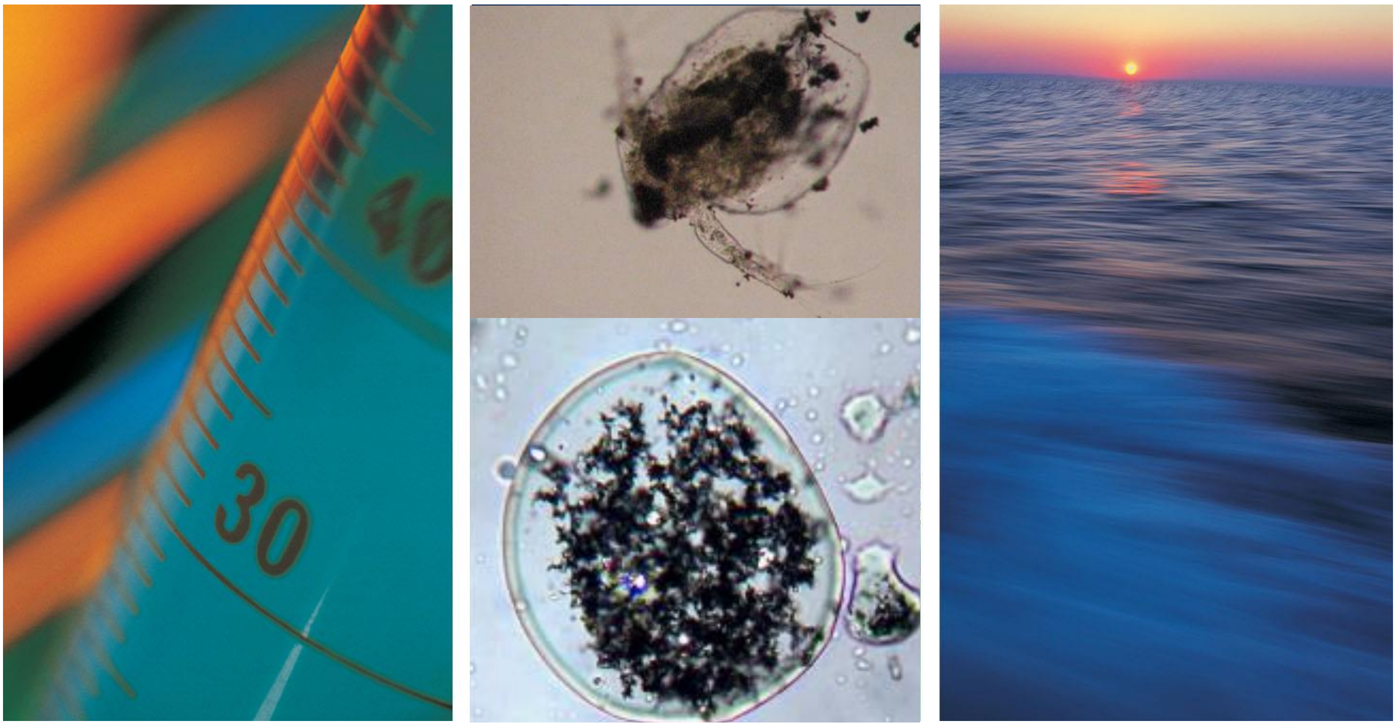
5) Consumption = production – export + import

6) The CN8 codes for the other esters of orthophthalic acid have changed during the period; all the used codes are listed

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

1,4-Dichlorobenzene in toilet blocks and in air fresheners – Final Report



AMEC Environment & Infrastructure UK Limited

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European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Lot 3 (1,4-Dichlorobenzene in toilet blocks and in air fresheners) – Final Report

AMEC Environment & Infrastructure
UK Limited

April 2012

Report for

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Contents

Executive Summary

This report provides the results of work on collection of data on abatement costs of reducing the use of 1,4-diclorobenzene (1,4-DCB) in toilet blocks and air fresheners. The work involved developing cost curves based on information readily available in the literature in combination with consultation with industry. It has been undertaken on behalf of ECHA by AMEC.

Much of the underlying data for the analysis in this report is based on a recent analysis on the socio-economic impacts arising from a proposal for risk reduction measures related to restrictions on 1,4-dichlorobenzene (RPA, 2010). This represents a useful, recent source of information so that much of the effort for the current work was devoted to obtaining any updated information as well as exploring methodological issues in developing cost curves.

The report includes best estimates of the current production volumes of 1,4-DCB-based toilet blocks and air fresheners in Europe. There are two known European manufacturers of 1,4-DCB with an overall production level of approximately 30,000 tonnes/year. However, it is understood that sales to European manufacturers of air fresheners and/or toilet blocks represents a very small part of their total sales with the remainder being used for the production of polyphenylene sulphide (PPS) resin, as an intermediate in the production of other chemicals such as 1,2,4-trichlorobenzene and to a lesser extent as a fumigant for the control of moths and moulds.

Stakeholder consultation found that there are a limited number of European manufacturers of either toilet blocks or air fresheners currently using 1,4-DCB. Following the change in the classification of the substance as a carcinogen (Category 2 according to Regulation 1272/2008) a number of previous manufacturers and suppliers stopped selling products based on 1,4-DCB and switched to 1,4-DCB-free alternatives. These trends suggest a decline in the use of 1,4-DCB in toilet blocks and air fresheners in Europe.

In total, it is estimated that in 2011 around 600 to 650 tonnes/year of 1,4-DCB was used in the production of toilet blocks, of which around 98% was for the professional market and only 2% was for domestic consumption. In terms of air fresheners, best estimates are that 83 tonnes of 1,4-DCB air fresheners are consumed in EU per year for domestic usage, and a further 100 tonnes of 1,4-DCB air fresheners are consumed for professional usage.

ECHA has concluded that a restriction on the placing on the market of 1,4-DCB-based air fresheners and toilet blocks is the appropriate risk management option. Therefore the cost curves developed only include the use of alternative products to replace the use of 1,4-DCB in toilet blocks and air fresheners, rather than measures to reduce (as opposed to eliminate) use or emissions. For both urinal and toilet rim blocks and air fresheners there appear to be a wide range of alternative products readily available on the market which come in a variety of forms. However, there is evidence that these alternatives are not as effective at masking malodours as 1,4-DCB which has a very strong deodorising effect. Additional cleaning or a combination of alternatives may be required to provide the same level of odour control as 1,4-DCB products. However, there is insufficient information to quantify this possible impact in the cost curves.

The cost curves developed illustrate the relative costs, in €/t of use avoided, of substituting 1,4-DCB in different areas of use. The measures assume 100% uptake of the least-cost alternative. In practice, it is likely that a range of different alternatives would be used, depending on users’ preferences. Substitution of air fresheners for professional

usage was found to be the most cost-effective measure (-€18.4k/t) in terms of use, whilst substitution of toilet blocks for professional use was found to be the least cost-effective measure (€7.8k/t).

The measures in the cost curves only include the relative prices of alternatives as compared to 1,4-DCB, and not other costs, such as investments in new equipment (which are assumed to be already reflected in the prices of alternatives) or sunk costs related to the residual value of any capital equipment that is currently used to produce 1,4-DCB-based blocks.

1. Project Understanding

1.1 The Project

The European Chemicals Agency (ECHA) has commissioned a project to provide information on “abatement costs for certain hazardous chemicals” (contract number ECHA/2011/140). The work is being undertaken by AMEC Environment & Infrastructure UK Limited (“AMEC”).

The present report is intended to provide a summary of the data collected on abatement costs of reducing the use of the substance 1,4-dichlorobenzene (EC Number 203-400-5, CAS Number 106-46-7) in toilet blocks and air fresheners.

The data collected is intended to be used for:

- Supporting the Agency in assessing the most appropriate risk management options for the substances addressed;
- Furthering the understanding of the usefulness of data on use/emissions abatement costs in risk management decision-making; and
- Supporting the Agency in the preparation of restriction dossiers.

1.2 Project Context

This project follows on from a 2010 project on “*Abatement cost curves for substances of concern*” conducted by AMEC Environment & Infrastructure UK (previously Entec UK) for the Environment Agency, ECHA, the Health and Safety Executive (HSE) and RIVM. The main aim of that project was to develop a suitable method for estimating abatement costs to reduce emissions of chemicals and to apply and test the method with three selected case study substances. That study provided a first illustration of the benefits of being able to compare unit abatement costs amongst different substances and different uses.

The objective of the present project is to assist ECHA in establishing capability to assess the abatement costs of reducing the use or emissions of hazardous substances. Under this lot (“Lot 3”), abatement costs for 1,4-DCB have been assessed specifically in: air fresheners and toilet blocks used by consumers at home and also in public toilets.

1,4-DCB is not currently included in the Annex XIV (the list of substances subject for authorisation). However, the European Commission has requested ECHA to prepare an Annex XV restriction report for 1,4-DCB. The project has concentrated on assessing abatement costs of shifting to alternatives (i.e. alternative products), as agreed with ECHA (in 2011).

The main outputs of the work, for this substance and for the other substances being assessed under different lots, are expected to be as follows:

- Data on abatement costs of reducing the use or emissions of the chemical. Different applications of the same substance may/will introduce different abatement costs.

- An overview on the functioning of the markets for the substances in question, including information on prices, amounts of the substance on the markets (including import and export), the relative shares of the substance used for different applications, number of actors involved in the business, as well as possible trends in the relevant market.

1.3 Structure of this Report

This report is structured as follows:

- Section 2 of this report provides an overview of the process of data gathering for this study and presents an overview of the data received;

- Section 3 sets out an overview of the general market for 1,4-DCB in air fresheners and toilet blocks;

- Section 4 presents the data analysis and resulting abatement cost curves;

- Section 5 presents conclusions.

The appendices to this report include various other background data.

2. Data Collection

2.1 Overview

The aim of the study is to gather abatement costs data for 1,4-DCB specifically in air fresheners and toilet blocks used by consumers at home and also in public toilets. Significant existing data on this topic already exists. Therefore, in this study, the focus has been on gathering new data from stakeholders in a targeted manner to supplement the existing data.

This chapter briefly summarises the process of data gathering for this study and presents an overview of the data received.

2.2 Review of Existing Data Sources

In 2010, Risk & Policy Analysts (RPA) conducted a study to perform an economic and social analysis of the use of 1,4-DCB in air fresheners and toilet blocks for the European Commission, DG Enterprise and Industry. The study considered a range of policy options in the form of operational conditions and risk management measures. It also considered different policy implementation options ranging from ‘command and control’ interventions to voluntary agreements and economic instruments. The report focussed on the use of 1,4-DCB-based products by consumers at home. Potential restrictions on the use of such products in public toilets were not considered in the impact assessment of options for policy change. However, the report refers to market data and other information relating to the use of the relevant products both at home and by professional users (in Annex 6).

RPA (2010) consulted with a wide range of stakeholders and examined a wide range of available literature. At the inception meeting for this project with ECHA, it was agreed that, given the comprehensiveness of the RPA (2010) study and the short intervening time period since the assessment was conducted, there would be little value in duplicating efforts to perform the same assessment (and insufficient resources allocated to do so). Instead the objective of this task is to build on the data gathered previously and supplement it with additional data gathering in order to develop appropriate abatement cost data.

2.3 Stakeholder Consultation

Information from stakeholders was sought on:

- Market data for 1,4-DCB products (such as on quantities, prices, number of actors and trends);
- Market data for potential alternative products (again covering quantities, prices, number of actors and trends);
- Available alternatives and the costs of implementing those alternatives in practice.

A questionnaire to collect the required information was developed in collaboration with ECHA. This was used as a basis for collecting information via telephone and written consultation with:

- manufacturers and suppliers of 1,4-DCB-based toilet blocks and air fresheners; and
- professional downstream users (i.e. cleaning companies).

The questionnaire was also sent to key trade associations representing the professional downstream users in the cleaning industry. In total, 81 organisations were contacted (see Table 2.1). However, only 17 organisations provided information for the study and only three questionnaires were completed and returned. There are, therefore, some relatively large data gaps, which we have attempted to fill using other sources of information. A list of consultees is presented in Appendix A.

Table 2.1 Overview of Stakeholder consultation

Stakeholder categories	Geographic remit of consultees	Number contacted	Number of responses
Associations			
The A.I.S.E Air Fresheners Product Stewardship Programme	Europe	23	2
Allpura - Verband Schweizer Reinigungs-Unternehmen	Switzerland		
ANCST Legacoop	Italy		
Asociacion Profesional de Empresas de Limpieza – ASPEL	Spain		
Associação Portuguesa Facility Services- AFPS	Portugal		
Bundesinnung der Denkmal-, Fassaden- und Gebäudereiniger – BIG	Austria		
Bundesinnungsverband des Gebäudereiniger-Handwerks - BIV	Germany		
Česka Asociace Úklidu A Cištění – CAC	Czech Republic		
Cleaning and Support Services Association - CSSA	United Kingdom		
Danish Service Industries Federation - DI	Denmark		
Eurochlor	Europe		
European Federation of Cleaning Industries	Belgium		
Fédération des Entreprises de Propreté et services associés - FEP	France		
Fédération Luxembourgeoise des Entreprises de Nettoyage - FLEN	Luxembourg		
Federazione Imprese di Servizi - FISE - ANIP	Italy		
Finnish Property Maintenance Association	Finland		
MATISZ	Hungary		
NHO Service	Norway		
Obrtna Zbornica Slovenije	Slovenia		

Stakeholder categories	Geographic remit of consultees	Number contacted	Number of responses
Ondernemersorganisatie Schoonmaak en Bedrijfsdiensten - OSB	Netherlands		
Polish Cleaning Chamber of Commerce	Poland		
Serviceentreprenörerna - ALMEGA	Sweden		
Union Générale Belge du Nettoyage - UGBN/ABSU	Belgium		
Industry			
Current and previous manufacturers of 1,4-DCB	France, Germany, Poland	15	3
Current and previous manufacturers, suppliers and importers of 1,4-DCB-based toilet blocks and air fresheners	Finland, Poland, UK, US	12	4
Current and previous of 1,4-DCB-based toilet blocks and air fresheners	Austria, Belgium, Finland, France, Germany, Greece, Italy, the Netherlands, Romania, Switzerland, UK	31	8

Whilst the response rate from the data collection exercise has been low and the level of additional information obtained beyond that present in the RPA report is limited, this in itself provides an indication of the relatively low importance to many companies of the current market, with information from our consultation suggesting that many companies have moved away from use of the substance in recent years, following the classification of the substance as a carcinogen.

3. General Market Overview

3.1 Introduction

This chapter provides a brief overview of the current market for 1,4-DCB in air fresheners and toilet blocks by domestic and professional users in Europe based on stakeholder consultation and existing information.

3.2 Manufacture of 1,4-DCB

The RPA (2010) study indicated that there were two active European manufacturers of 1,4-DCB (located in Germany and Poland) with an overall production level of just above 30,000 tonnes/year. It is understood however that for both companies sales to European manufacturers of air fresheners and/or toilet blocks represents a very small part of their total sales, with the remainder being used for the production of polyphenylene sulphide (PPS) resin, as an intermediate in the production of other chemicals such as 1,2,4-trichlorobenzene and to a lesser extent as a fumigant for the control of moths and moulds. Consultation for the present study found that only one of these companies knowingly supplies 1,4-DCB for the manufacture of air fresheners and toilet block products.

3.3 Manufacture and Supply of 1,4-DCB-based Toilet Blocks and Air Fresheners

3.3.1 Overview

Information has been collected from available literature, online searches and direct stakeholder consultation. In particular, efforts were made to check whether companies identified as being suppliers of 1,4-DCB-based blocks and/or air fresheners in relevant Safety Data Sheets are still active in the market.

3.3.2 Toilet blocks

Stakeholder consultation suggests that there are a limited number of currently active EU-based manufacturers of toilet blocks using 1,4-DCB. No single company has been identified through consultation as still using 1,4-DCB in the manufacture of toilet blocks. One company that previously manufactured 1,4-DCB-based urinal blocks (for professional usage) indicated that, at the height of production (in the early to mid-2000s), there may have been between 15-20 active European manufacturers of 1,4-DCB-based toilet blocks. The company in question stopped producing 1,4-DCB-based blocks around five years ago and now imports them from China. It took the decision to not invest in the new machinery required to switch to producing alternatives. However, this company anticipates that it will stop importing 1,4-DCB-based urinal blocks altogether by 2013 (in order to avoid having to register the substance under REACH). This company indicated that their experience is common across Europe with many of former manufacturers now either importing 1,4-DCB-based blocks from outside Europe or only offering 1,4-DCB-free alternatives. Imports are understood to come largely from China. Consultation with industry further suggests that manufacture of such products also takes place in India and the USA.

For the purposes of this study it is assumed that there are currently a maximum of five active European manufacturers of 1,4-DCB-based toilet blocks[1]. As a practical working assumption, it is assumed that, of the remaining 15 companies that previously manufactured 1,4-DCB toilet blocks, half of them continue to supply 1,4-DCB products through imports, whilst the other half have replaced them entirely with alternative products. In practice, the numbers of companies may be lower (or higher) than the above estimates.

Consultation suggests that there are a significant number (100s) of European companies that distribute 1,4-DCB-based toilet blocks, but typically for professional use (in public toilets). This is based on our own review of suppliers' websites. The vast majority of these also offer a range of alternative products in their product catalogues. Several suppliers indicated during consultation that, following the change in classification of the substance (in 2004), they decided to stop selling 1,4-DCB-based products and switched to supplying alternatives. This corresponds with the findings of the RPA (2010) study.

3.3.3 Air fresheners

It is understood that there are two applications for 1,4 dichlorobenzene-based air fresheners:

- In a small size (typically 80g per unit) within a container to deodorise rooms for domestic or professional user; or
- In large sizes (up to 10kg per unit), called 'super blocks'[2] which may be used in industrial settings by professional users only.

The RPA (2010) report concluded that there may be around 10 companies in Europe producing 1,4-DCB air fresheners for use at home. Input was received from only one company currently producing 1,4-DCB-based air fresheners in Europe. This company is located in Poland and supplies both the domestic and professional markets with small units to deodorise rooms. A number of suppliers importing 'super blocks' from the USA have been identified, but no qualitative information was obtained. No other information was received from industry on the use of 1,4-DCB in air fresheners. Therefore, for the purposes of the current study it is assumed that there are a maximum of ten active European manufacturers of 1,4-DCB-based air fresheners[3].

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[1] It should be made clear that this is an assumption based on available information and that five companies have not been identified during the course of this study.

[2] For instance, EA Supplies (2012)

[3] It should be noted that this is an assumption based on the available information and that only one company has confirmed sales to consumers.

3.4 Current Best Estimates

3.4.1 Toilet blocks

RPA (2010) estimated that 980 tonnes of 1,4-DCB-based toilet blocks were consumed in the EU in 2009 (963t of urinal blocks for professional usage and 17t of toilet rim blocks for domestic usage). Consultation with suppliers of these products indicates that this may be an overestimate for the current market, although no improved quantitative information has been made available by the organisations consulted.

As described above, it is understood that following the change in the classification of the substance as a carcinogen (Category 2 according to Regulation 1272/2008) a number of previous manufacturers and suppliers stopped selling products based on 1,4-DCB and switched to 1,4-DCB-free alternatives. In the absence of more detailed information, it is assumed that of the estimated 20 European companies[4] previously producing toilet blocks using 1,4-DCB, in 2012:

- Up to five companies continue to use 1,4-DCB to produce toilet blocks in Europe;
- Half of the remainder (i.e. 7-8 companies) now import 1,4-DCB-based toilet blocks to the EU; and
- The remainder (i.e. 7-8 companies) produce 1,4-DCB-free blocks.

It should be noted that the above are only assumptions, based on the previous data and anecdotal suggestions from consultation during the present study. They should not be interpreted as being fully accurate as insufficient data were available to provide complete estimates.

Table 3.1 shows estimates (based on available information and consultation) of total volumes of 1,4-DCB-based toilet blocks manufactured in, imported to and exported from the EU. It is estimated that the overall market for 1,4-DCB-based toilet blocks has decreased by around 35% from that estimated in the RPA (2010) study to a size of around 600-650t per year.

Table 3.1 Summary of assumed manufactured, imported, exported and used quantities of toilet blocks, in 2011

	Quantity	Comment
Number of companies manufacturing 1,4-DCB toilet blocks in EU	Up to 5 companies	This is based on consultation with industry and trends noted in the RPA (2010) study, as noted in the main text. These companies have not been directly identified. Each company is assumed to produce approximately 50 t/yr of 1,4-DCB-based toilet blocks.
1,4-DCB blocks manufactured in the EU	250 t	It is assumed that each company produces approximately 50t of 1,4-DCB toilet blocks.

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[4] Based on consultation with industry and findings of the RPA (2010) study.

	Quantity	Comment
Imported to the EU	350-400 t	It is assumed that the change in behaviour was mirrored by a proportional change in the quantity replaced by imports e.g. 7-8 companies importing 50 t/yr of 1,4-DCB toilet blocks each. Imports known to come from China, India and the US.
Exported from the EU	N.A.	No specific information available.
Total consumption of 1,4-DCB toilet blocks in EU in 2011	600 - 650 t	98% (e.g. 588 - 637 t/yr) for professional use 2% (e.g. 12 - 13 t/yr) for domestic use

*

Note: Given the small number of information sources the tonnages in the table should be considered as indicative only.

3.4.2 Air fresheners

Very little information was from available on the consumption of 1,4-DCB-based air fresheners in the EU for either the domestic or professional market. The RPA (2010) study estimated that 83 tonnes of 1,4-DCB air fresheners were consumed in EU in 2009 for domestic usage. The study estimated that approximately 100 tonnes of 1,4-DCB air fresheners were consumed in EU in 2009 for professional usage. Information was only received from one company using 1,4-DCB in the production of air fresheners. As only very limited information was provided from industry during consultation on this usage, the RPA (2010) figures quoted have been used for the cost curve analysis.

Efforts were made to obtain information on volumes of imported air fresheners containing 1,4-DCB, in order to supplement the information obtained through consultation. However, Comext[5] only contains aggregated import and export data on “preparations for perfuming or deodorising rooms, including odoriferous preparations used during religious rites” and does not disaggregate by specific substance used. This does not provide sufficient disaggregation to obtain a reliable estimate of imports of 1,4-DCB-based products.

Table 3.2 shows estimates (based on available information and consultation) of total volumes of 1,4-DCB-based air fresheners manufactured in, imported to and exported from the EU.

Table 3.2 Summary of assumed manufactured, imported, exported and used quantities of air fresheners in 2011

	Quantity	Note
Number of companies manufacturing 1,4-DCB-based air fresheners in EU	Up to 10 companies	Whilst only one company was identified through consultation, it is assumed that there are more companies using 1,4-DCB in the production of air fresheners in Europe. 10 companies are considered to be the maximum number.

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[5] Comext is the Eurostat reference database for external trade. It contains both recent and historical data from the European Union Member States and a significant number of third countries covering the value of exports and imports of products.

	Quantity	Note
1,4-DCB air fresheners manufactured in the EU	No data	No specific information available.
Imported to the EU	No data	No specific information available.
Exported to the EU	No data	No specific information available.
Total consumption of 1,4-DCB air fresheners in EU	183 t	100t for professional use as per RPA (2010) 83t for domestic use as per RPA (2010)

Note: Given the small number of information sources the tonnages in the table should be considered as indicative only.

3.5 Market Statistics for the EU air fresheners and toilet block market

Information in the literature on these markets of these products is relatively limited.

Consultation with stakeholders suggests that, in 2010, the EU air fresheners market amounted to 85,000t[6]. The total sales value of the market was estimated to be around €1,965 million in 2009 (RPA, 2010). Table 3.2 indicated that the EU market for 1,4-DCB air fresheners is approximately 183t per year. The market value for these products is approximately €4 million per year. In market value terms, therefore, the 1,4-DCB air fresheners is estimated to represent less than 1% of the total EU air freshener market.

No source of information on the EU urinal blocks market was identified. According to (RPA, 2010) there are a total of 12.5million flushed urinals in Europe (excluding waterless urinals). This is considerably lower than the number estimated in a recent JRC (2011) study (42 million units, excluding waterless urinals). The latter figure is taken to be the more up-to-date. Table 3.1 indicated that the EU market for 1,4-DCB toilet blocks is 612.5t. This translates to a total number of approximately 440,000 urinals treated per year (see Table 4.1). This represents around 1% of the total number of flushed EU urinals.

According a recent JRC (2011) study, the total EU27 toilet stock is estimated at around 394 million units, including 277 million domestic toilets (70%) and 117 million non-domestic toilets (30%). The RPA (2010) study estimated total toilet rim blocks sales in the EU-27 in 2008 to have been in the region of €709 million (of which 70% were assumed to be for the domestic market e.g. €496 million). Table 3.1 indicated that the EU domestic market for 1,4-DCB toilet blocks is approximately 12.5t. The market value for these products is approximately €4.4 million per year. In these terms, the 1,4-DCB blocks therefore represent less than 1% of the total EU toilet rim block market.

This information is summarised in Table 3.3.

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[6] Personal communication with A.I.S.E, February 2012

Table 3.3 EU air fresheners and toilet block market

	Air fresheners		Toilet rim blocks		Urinal blocks	
	In the EU	**1,4-DCB-based air fresheners consumed**	**In the EU**	**1,4-DCB-based blocks consumed**	**In the EU**	**1,4-DCB-based blocks consumed**
Tonnage	85,000t	183t (100 for the professional market and 83t for the domestic market)	No data	12.5t	No data	612.5t [1]
Toilets/urinals treated per year	N.A.	N.A.	277 million domestic toilets in EU *	0.01 million domestic toilets (<1%)	42 million flushed urinals in EU *	0.44 million urinals (1%) [2]
Market value (€)	1,965 million	4.0 million (<1%)	496 million	4.4 million (<1%)	No data	0.2 million

Notes:

1) In 2010, RPA estimated that 1,4 dichlorobenzene-based urinal blocks accounted for 15-30% of the EU urinal block market.
2) In 2010, RPA estimated that 1,4 dichlorobenzene-based urinal blocks were used in ca. 6% (or ca. 710,000) of flushed urinals in the EU.

* Including urinals and toilet bowls that are not treated.

3.6 Alternatives

3.6.1 Toilet blocks

For both urinal and toilet rim blocks there appear to be a very large range of alternative products readily available on the market which come in a variety of forms (such as adhesive in-bowl discs, cistern blocks, in-bowl block and various others). RPA (2010) examined in detail the compositions of alternative toilet block products. These alternative products appear to be made up of a number of key components, including:

- Fragrances e.g. d-limonene, linalool;
- Surfactants e.g. peg hydrogenated castor oil;
- Preservatives e.g. benzyl salicylate;
- Dyes e.g. CI21095;
- Solvents e.g. ethanol;
- Thickeners e.g. xanthan gum; and
- Stabilisers e.g. propylene glycol.

These components are typically present in much lower concentrations in the final product than the typical concentration of 1,4-DCB (which is typically above 98%). The RPA (2010) study found that surfactant-based blocks such as sodium dodecylbenzenesulphonate (a type of linear alklybenzene sulphonate) are the main alternatives to 1,4 dichlorobenzene-based products. This has been corroborated through consultation with industry for the present study, with two companies suggesting that these would be the most likely substances to be used as alternatives. Other blocks may be based on enzymes (biological blocks) or specialised fragrances.

These alternative products have a higher water solubility and, therefore, it is thought that they have a shorter 'lifespan' than 1,4-DCB-based blocks in high traffic urinals (RPA, 2010). This was corroborated during stakeholder consultation by one company, although no quantitative information was provided. A separate company that previously sold 1,4-DCB-based blocks indicated that their alternative product has a comparable longevity.

The RPA (2010) study found that alternative products based on surfactants or enzymes may offer additional cleaning properties as well as a descaling action which are not offered by traditional 1,4-DCB-based products.

On the other hand, it is reported by industry that such alternative products are not as effective at masking malodours as 1,4-DCB which has a very strong deodorising effect. Consultation with suppliers of blocks suggests that additional cleaning may be required when using alternatives to address the underlying malodour problems (which is seemingly contradictory). A combination of alternatives could be used to provide the same function. Therefore, there may be a trade-off for end-users in replacing 1,4-DCB products with alternatives. However, no information has been received from end-users using alternative products and it is likely that the extent to which any additional cleaning is needed/undertaken will depend on individual circumstances and preferences. It has not been possible to build this complexity into the cost curve.

3.6.2 Air fresheners

There are a wide variety of alternative air-freshener products readily available on the market which also come in variety of different delivery formats, such as aerosols, gels, wick-in liquids and plug-in units. The latter two are thought only to be used by domestic users. No detailed information is available on the preference of these products by users.

3.7 Prices

Prices of 1,4-DCB-based products were analysed in detail in the RPA (2010) report. The prices of professional urinal blocks have been updated based on our own analysis (see Appendix B). Prices of the other products are based on the RPA (2010) study but have been updated for inflation to 2011 prices using the Harmonised Index of Consumer Prices (HICP).

Prices of alternative products were analysed in detail in the RPA (2010) report. It has been agreed that for the purposes of the abatement cost calculation, 100% uptake of the least-cost option should be assumed. In practice, it is likely that a range of different alternatives would be used, depending on users' preferences, with some being equal to or higher in price than products based on 1,4-DCB. Furthermore, consultation with industry indicated that some end-users do not actively select the products they use, instead they are guided by their suppliers' catalogues. This could be a reason for end-users not to opt for less expensive alternatives in 100% of cases.

The prices used for the remainder of the assessment are presented in Table 3.4.

Table 3.4 Prices of 1,4-DCB based products vs. available alternatives

	1,4-DCB-based product	Alternative products	Source
Professional urinal blocks	Average: €7.19/kg or €0.58 per 80g unit	Surfactant-based (av.): €0.40 per unit Enzyme-based (av.): €0.87 per unit* Fragrance-based (av.): €2.32 per unit*	See Appendix B.
Domestic toilet blocks	Average: €1.32 per 70g unit	Bottom-of-range cistern-block: €0.16 per unit Average of all alternatives: €0.98 per unit*	RPA (2010) prices adjusted for inflation to 2011 prices using HICP and VAT removed.
Professional air fresheners[1]	Average: €1.75 per 80g unit	Bottom-of-range aerosol spray: €0.28 per unit Average of all alternatives: €4.43 per unit*	RPA (2010) prices adjusted for inflation to 2011 prices using HICP and VAT removed.
Domestic air fresheners	Average: €2.10 per 80g unit	Bottom-of-range aerosol spray: €0.28 per unit Average of all alternatives: €4.43 per unit*	RPA (2010) prices adjusted for inflation to 2011 prices using HICP and VAT removed.

Notes:

1) No specific information was available on prices of professional air fresheners. Therefore prices of domestic products have been used for the remainder of the assessment.
2) * Not used for cost curve calculation – for context only.

3.8 Trends

Whilst limited information was received during consultation from industry, a few companies provided details of market trends in the use of 1,4-DCB in toilet blocks. Important points to note include:

- Several manufacturers of toilet blocks have indicated they no longer use 1,4-DCB following the change in the classification of the substance with Commission Directive 2004/73/EC of 29 April 2004. Whilst some of the companies now import 1,4-DCB blocks, others have switched entirely to alternative products.

- Several suppliers of toilet blocks have indicated that they no longer sell 1,4-DCB-based products following the change in the classification of the substance. Those suppliers that continue to stock

> 1,4-DCB-based blocks also sell a range of alternative products. There appear to be a great variety of alternative toilet rim and urinal block products (see Appendix B) readily available on the market.

These trends suggest a decline in the use of 1,4-DCB in toilet blocks. This supports the findings of the RPA (2010) study which observed similar trends. However, such projected trends may not necessarily lead to the elimination of the use of 1,4-DCB in these products, particularly with regards to urinal blocks. Consultation with industry suggests that strong odour masking properties and low water solubility means that these products remain popular, particularly for professional usage. One company indicated that no current alternative products offer such effective odour masking properties as 1,4-DCB urinal blocks.

Scarce information was available on the use of 1,4-DCB-based air fresheners in the EU. However, some general observations can be made. 1,4-DCB products are considered "old-fashioned" (RPA, 2010) and as there are a variety of suitable alternatives available (see Section 3.6.2.) it is likely that their consumption will decline.

4. Data Analysis

4.1 Uses and Releases

4.1.1 Overview

Much of the information presented in this section is based on the RPA (2010) study and is supplemented with the limited information received during stakeholder consultation for the current study.

4.1.2 Toilet blocks

As discussed in Section 3.4.1, it is estimated that in 2011 around 600 to 650 tonnes/year of 1,4-DCB was used in the production of toilet blocks, of which around 2% (12 - 13 t/yr) was for the domestic market and 98% (588 - 637 t/yr) was for the professional market, which is a reduction from the equivalent figure in 2009, when 980 tonnes/year were estimated to be used (a decrease of approximately 40%). This is thought to be in line with the declining trend in the use of this substance for this market as indicated in the RPA (2010) study and information received during consultation.

Consultation with industry has suggested that the majority of European companies that formerly used 1,4-DCB in the production of toilet blocks have now switched to alternatives or have replaced stocks with imports. There are, therefore, assumed to be a maximum of five companies manufacturing toilet blocks based on 1,4-DCB in the EU currently (based on consultation and trends noted in the RPA (2010) study). There were thought to be between 15-20 companies previously. No further information has been provided to allow for a more detailed assessment on the location or specific number of sites.

Table 4.1 provides a summary of the quantity of 1,4-DCB-based toilet block markets for both domestic and professional usage.

Table 4.1 Assumptions on number of toilet block products sold annually in the EU, 2011

Product	Quantity of 1,4-DCB used (t/yr)	Weight per unit (g)[1]	Number of units sold per year (approx.)	Number of toilet/urinals treated per year
Domestic – toilet rim blocks	12.5	70	180,000	12,000[2]
Professional – urinal blocks	612.5	80	7,650,000	440,000[3]

Note:

1) RPA (2010) study.
2) Based on an average lifetime of 25 days (RPA, 2010)
3) Based on an average lifetime of 21 days (RPA, 2010)

4.1.3 Air fresheners

As discussed in Section 3.4.1, it is estimated that in 2011 approximately 180 tonnes/year of 1,4-DCB was used in the production of air fresheners (around 80 t/yr for the domestic market and 100 t/yr for the professional market). These figures are based on the RPA (2010) study as very limited data was made available by industry stakeholders on current levels of usage.

There are assumed to be a maximum of ten companies manufacturing air fresheners based on 1,4-DCB in the EU currently (based on consultation and trends noted in the RPA (2010) study). No further information has been provided for the current study to allow for a more detailed assessment on the location or specific number of sites.

Table 4.2 provides a summary of the quantity of 1,4-DCB-based air fresheners markets for both domestic and professional usage.

Table 4.2 Assumptions on number of air freshener products sold annually in the EU,

Market	Size of market (t/yr)	Weight per unit (g)[1]	Number of units sold per year (approx)
Domestic	83	80	1,000,000
Professional	100	80	1,250,000

Note:

1) These figures are taken from the RPA (2010) study.

4.1.4 Environmental releases

Environmental releases of 1,4-DCB from use in toilet blocks and air fresheners are not considered here as emissions to the environment are not the main concern. Therefore, abatement cost curves have been developed in terms of cost per unit of use removed rather than per unit of environmental release abated.

4.2 Current and Planned Abatement Measures

Most of the existing legal requirements for 1,4-DCB are not targeted at air fresheners and toilet blocks specifically, as they relate to more general requirements. In particular, the change in classification of 1,4-DCB with Commission Directive 2004/73/EC of 29 April 2004 to a carcinogen has led to a significant decline in the use of this substance for the products of concern. This trend is further described in Section 3.8. Furthermore, it is of note that ECHA is preparing a restriction dossier for 1,4-DCB in air fresheners and toilet blocks on request from the European Commission. In addition, Sweden has a national restriction on chemical products containing 1,4-dichlorobenzene and intended to mask smells (from 1990).

4.3 Possible Future Abatement Measures

4.3.1 Information sources

Based on the information reviewed above, a number of potential further measures are possible. Measures identified in the literature include use of alternative products to replace the use of 1,4-DCB in toilet blocks and air fresheners.

4.3.2 Measures included in the cost curve

It has been agreed that for the purposes of the abatement cost curve development, the 1,4-DCB-based blocks would be substituted by the least-cost alternative.

The following measures have been included in the cost curve:

- Substitution of 1,4-DCB-based urinal blocks for professional usage with urinal blocks based on surfactants;
- Substitution of 1,4-DCB-based toilet rim blocks for domestic usage with bottom-of-range cistern blocks;
- Substitution of 1,4-DCB-based air fresheners for professional usage with bottom-of-range aerosol sprays;
- Substitution of 1,4-DCB-based air fresheners for domestic usage with bottom-of-range aerosol sprays.

In practice, it is likely that a range of different alternatives would be substitute some of the 1,4-DCB use, depending on users' preferences. For example, fragrance-based urinal blocks and enzyme-based urinal blocks are currently available on the market at prices of €2.32 and €0.87 per unit respectively. However, for simplicity, only the least-cost options are taken forward as part of the cost curve development. It should also be noted that demand may be affected by the significantly lower price of the alternatives. However, this is not considered quantitatively.

Appendix C of this report provides details on the data sources and assumptions used in assessing the potential for each of the measures to reduce use of the substance and the associated costs.

4.4 Cost Curve for 1,4-DCB in Toilet Blocks and Air Fresheners

Table 4.3 provides a summary of the key data on each of the measures for inclusion in the cost curve. Further details are included in the supplementary spreadsheet.

Figure 4.1 present the cost curve, showing marginal costs.

We have attempted to calculate all costs in line with ECHA Guidance on Compliance Costs[7]. However, it is important to note that the cost curves are based on the current prices of alternatives that are already on the market. In practice, one of two outcomes is possible for companies that currently manufacture 1,4-DCB-based products in the EU, either:

1. They will cease this area of their business. It is understood that it would not be possible to use some of the existing machinery for other applications than producing 1,4-dichlorobenzene products[8]. This equipment therefore has no alternative uses and its opportunity cost is effectively zero. Any residual value which the equipment had prior to the imposition of the restriction is therefore rent which is transferred from consumers to producers in the form of prices higher than opportunity cost. For the purposes of the current analysis, this rent has not been included in the costs curves. It is estimated that the residual value of this capital equipment could be perhaps €250-300,000 or around €60,000 as an annualised value[9].

2. Or, they would make necessary investments to supply alternatives. It is assumed that the costs of such investments are reflected in the current prices of the alternatives that are already on the market.

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[7] Addendum to the Guidance on Socio-economic Analysis – Restrictions: Calculation of compliance costs. Available at: http://echa.europa.eu/documents/10162/17087/appendix1-calculation__compliance_costs_case_restrictions_en.pdf.

[8] According to RPA (2010): "Cost of decommissioning existing machinery: the price for a press with a compacting tool is approximately between €80,000 and €250,000 per machine. However, given the long lifetime of these machines, those currently in use could well be old ones, worth only a fraction of their original price. It has been argued that, as these machines do not have a dual use (alternative formulations cannot be manufactured in these presses), the residual investment in these machines for use with 1,4-DCB would be lost."

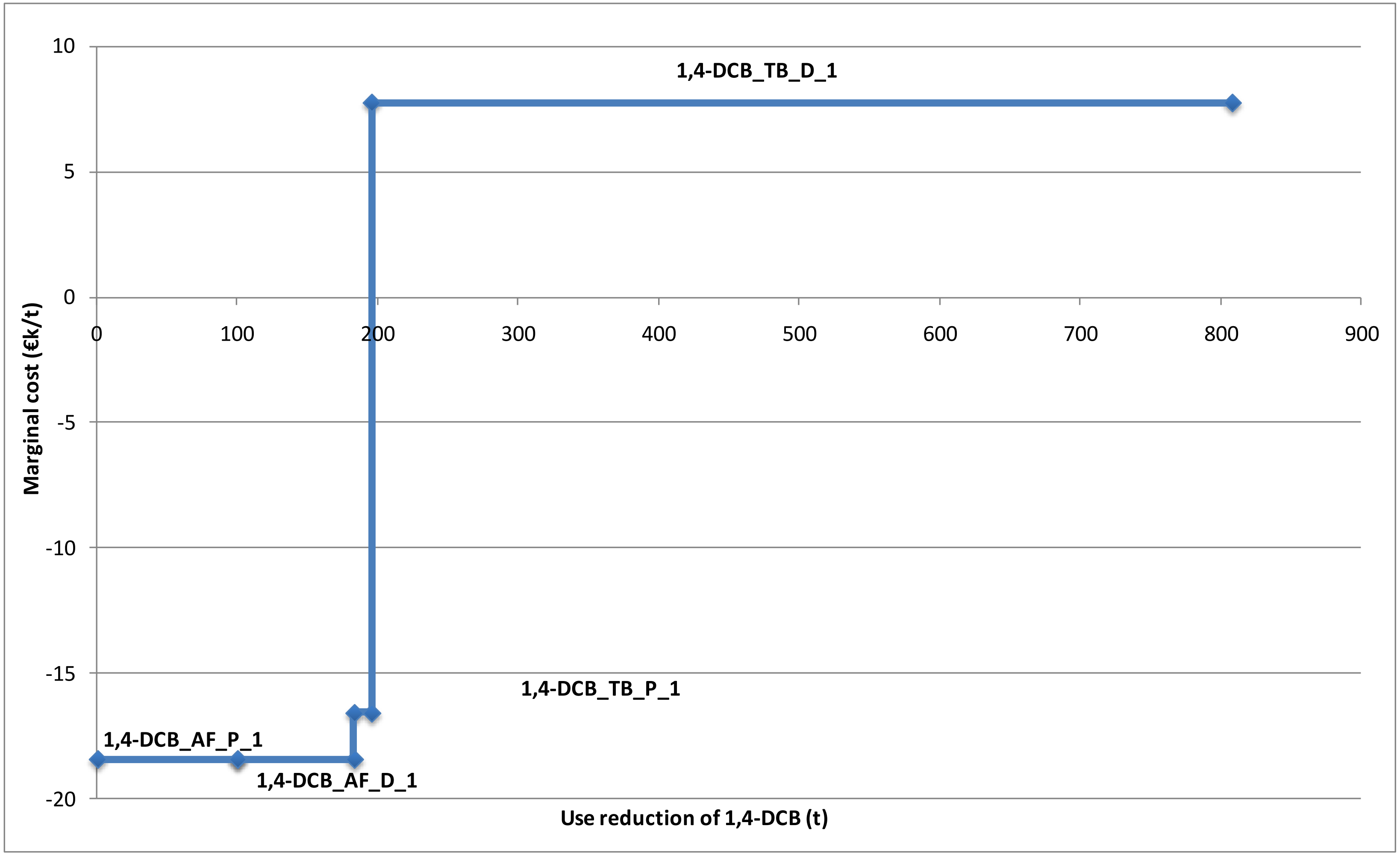
[9] Assuming a value when new of €80-250k per machine (taking the mid-point), and that the residual value is one third of the value when new, using a discount rate of 4% and an assumed average 5-year remaining useful lifetime. Based on one machine per company (five companies in total).

Table 4.3 Summary of cost curve data

Measure	Single-measure cost (€k)	Single-measure emission reduction (t)	Cost-effectiveness (€k/t)	Incremental cost (€k)	Incremental emission reduction (t)	Incremental cost-effectiveness (€k/t)	Notes
1,4-DCB_AF_P_1	-1841.8	100.0	-18.4	-1841.8	100.0	-18.4	100% uptake of least-cost option assumed.
1,4-DCB_AF_D_1	-1528.7	83.0	-18.4	-1528.7	83.0	-18.4	100% uptake of least-cost option assumed.
1,4-DCB_TB_D_1	-206.7	12.5	-16.5	-206.7	12.5	-16.5	100% uptake of least-cost option assumed.
1,4-DCB_TB_P_1	4746.9	612.5	7.8	4746.9	612.5	7.8	100% uptake of least-cost option assumed.

Notes: All data on costs and emission reductions are at an EU level.

Figure 4.1 **Marginal cost curve for 1,4-DCB in toilet blocks and air fresheners**



5. Conclusions

The cost curves developed illustrate the relative costs, in €/t of use avoided, of substituting 1,4-DCB in different applications. The cost curves are based on the current prices of alternatives that are already on the market. This assumes that the costs of any investments required by current producers of 1,4-DCB-based products are reflected in the current prices of the alternatives that are already on the market. The residual value of capital equipment has not been considered in the cost curves. It is important to note, therefore, that costs incurred by individual companies currently involved in the production and supply of 1,4-DCB-based products are not included in the cost curves. These costs could involve significant implications for the companies concerned (related to loss of market and associated employment) if they are not able to supply alternatives and/or loss of the residual value of their investment in capital equipment that they currently use to produce 1.4-DCB-based products.

Substitution of air fresheners for professional usage was found to be the most cost-effective measure (-€18.4k/t) in terms of use, whilst substitution of toilet blocks for professional use was found to be the least cost-effective measure (€7.8k/t).

However, these costs may underestimate the true costs of restrictions for two reasons. Firstly, the measures assume 100% uptake of the least-cost alternative. This is a simplistic analysis as, in practice, it is likely that a range of different alternatives would be used, depending on users' preferences. More sophisticated cost curves would require further data collection and analysis in order to understand the likely uptake of different alternatives.

Secondly, there is evidence that these alternatives are not as effective at masking malodours as 1,4-DCB which has a very strong deodorising effect. Additional cleaning or a combination of alternatives may be required to provide the same level of odour control as 1,4-DCB products. However, alternative products may offer additional cleaning properties, which are not offered by traditional 1,4-DCB-based products. However, no information has been received from end-users using alternative products and it is likely that the extent to which any additional cleaning is needed/undertaken will depend on individual circumstances and preferences. As a result, it has not been possible to quantify this possible impact in the cost curves [10].

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[10] If there were sound evidence that companies were having to spend additional time cleaning as a result of using alternative products this could be calculated as an ongoing cost and incorporated in the cost curve.

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Wray Bros (2012b) Product detail available online: http://www.wraybros.co.uk/shop/product/215-bc077/

Yate Supplies (2012) Product detail available online: http://www.yatesupplies.com/product.php?product=CBP&category=0004001300150001&utm_source=google&utm_medium=products

Appendix A
List of Organisations Consulted

The table below lists all the organisations that were contacted, indicating those that provided information that was used in this report, as well as those that provided a completed questionnaire.

Company	Information provided?
Aarti Industries / Alchemie Europe	
A.I.S.E	✓
Allegri Cleaning	
Allpura – Verband Schweizer Reinigungs-Unternehmen	
Amity International	
ANCST Legacoop	
Arkema	
Aronia N.V.	
Asociacion Profesional de Empresas de Limpieza – ASPEL	
Associação Portuguesa Facility Services- AFPS	
Biltrec SA	
Bogdol GMBH	
Bundesinnung der Denkmal-, Fassaden- und Gebäudereiniger – BIG	
Bundesinnungsverband des Gebäudereiniger-Handwerks – BIV	
Ceda Chemicals	
Česka Asociace Úklidu A Cištění – CAC	
Chevron Phillips	✓
CLANDREX SERVICES	✓
Cleaning and Support Services Association – CSSA	
Cleenol	✓
Danish Service Industries Federation – DI	
DOSIM SA	
Dr. Sasse Gebäudereinigung AG	
EA Supplies	
Ecological	
Eurochlor	✓
European Federation of Cleaning Industries	
Evans Vanodine	✓*

Company	Information provided?
FARE	
Fédération des Entreprises de Propreté et services associés – FEP	
Fédération Luxembourgeoise des Entreprises de Nettoyage – FLEN	
Federazione Imprese di Servizi – FISE – ANIP	
Finnish Property Maintenance Association	
Fresh Products	
Gebäudereinigung – Krankenhausservice Zehnacker GmbH	
GEPE-Gebäudereinigung PETERHOFF	
Global Group	
GRG – Grossberliner Reinigungs-Ges. Hans-Jochen Schwarz KG	
Halliburton	
HECTAS Gebäudedienste Stiftung & Co. KG	
Initial (part of Rentokil)	
ISS Mediclean Limited	
James Briggs UK	✓
Jeyes	✓
Kalvei	
Klüh Cleaning GmbH	
Lanxess	✓
lassila-tikanoja	✓*
LSR Associates Ltd	
LUXELACALIS	
Master Cleaning Services	✓
MATISZ	
Multiclean	✓
NHO Service	
Obrtna Zbornica Slovenije	
OCS Support Services Limited	
Ondernemersorganisatie Schoonmaak en Bedrijfsdiensten – OSB	
ORKA d.o.o.	
PCC Rokita	✓
Piepenbrock Unternehmensgruppe GmbH & Co. KG	
Plural Servicepool GmbH	
Polish Cleaning Chamber of Commerce	
Principle Cleaning Services Limited	

Company	Information provided?
Recochem	✓
Reiwag Facility Services GmbH	
Rtkpalvelu	
SCUOLA NAZIONALE SERVIZI	
Serviceentreprenörerna – ALMEGA	
Sky Chemicals	
SSS	
Staples Disposables Ltd UK	✓
Stormindustriediensten	✓
TAKATA-PETRI	
Tampen and Tampen	
Ticona GmbH	
Toray International Europe GmbH	
TOSOH EUROPE B.V.	
Trust Hygiene	✓
Union Générale Belge du Nettoyage – UGBN/ABSU	
WISAG GEBÄUDEREINIGUNG HOLDING GMBH & CO. KG	
Zakład Produkcyjny IRBIS Dulanowicz	✓*

Notes:

* Questionnaire completed and returned.

Appendix B
Price Analysis of Professional Toilet Blocks

The RPA (2010) report found that 1,4-DCB-based urinal blocks ranged in price from €6.25-13.62/kg (including VAT) while DCB-free blocks ranged from €8.75-35/kg. This information was updated with more up-to-date prices where possible (e.g. for Lemon Channel Blocks, Citrus Channel Cubes, Ribo Special and Dr. Becher Extra). The prices of the remaining two products, Fresh Urinal Para Block and 1,4 –DCB product A (RPA, 2010) were inflated to 2011 prices using the HICP. Additional information was collected on the current prices of other 1,4-DCB-based urinal blocks available on the market. In total, the analysis covered ten products (see Table C.1).

The updated information suggests that the price range is €4.29-10.97/kg excluding VAT (€5.15-13.16/kg including VAT) which correlates well with the RPA (2010) study. The average price of 1,4-DCB containing toilet blocks was €7.19 per kg or €0.58 per unit excluding VAT (€0.71 per unit in the RPA (2010) report).

Table C.1 Prices of selected 1,4-DCB-based urinal blocks

Product name	Price in € (incl. VAT)	Quantity	Price in € per kg (excl. VAT)[1]	Source
Conqueror	24.62	3kg	6.84	Consultation
1066 Conqueror channel/toilet blocks (Evans)	23.18	3kg	6.44	Total Cleaning Supplies
Citrus Channel Cubes	32.17	3 kg	8.94	MSC Industrial Supply Co. (2012a)
NILCO Citrus toilet blocks	20.76-27.42	3.25kg	5.32 – 7.03	MSC Industrial Supply Co. (2012b) Mammoth Cleaning Supplies (2012a)
Staples Disposables Channel & Urinal Blocks (Lemon Channel Cubes)	24.30	3kg	6.75	Mammoth Cleaning Supplies (2012b)
Lemon Channel Cubes	15.45	3kg	4.29	Beaucare Medical (2012)
Ribo Special		1kg	6.66	Doro Flotter Feger (2012)
Dr. Becher Extra		1kg	10.97	Hygi.de (2012)
1,4 dcb product A		1kg	5.47	RPA (2010)
Fresh Urinal Para Block		1kg	9.41	RPA (2010)
TOUGH Guy, Super Block, para	734.82	10kg	61.24	EA Supplies (2012)

Product name	Price in € (incl. VAT)	Quantity	Price in € per kg (excl. VAT)[1]	Source
Notes: 1) An average EU VAT rate of 20% has been assumed in order to calculate prices excluding VAT.				

The prices of alternatives, i.e. 1,4-DCB free toilet blocks were also analysed in detail (see Table C.2). This was to provide greater detail into the prices of specific alternative types e.g. surfactant-based vs. enzyme-based. In total, the analysis covered 19 products[11].

Table C.2 Prices of selected 1,4-DCB-free urinal blocks

Product name	Price in € (incl. VAT)	Quantity	Price in € per kg (excl. VAT)[1]	Price in € per unit (excl. VAT)	Source/ comments
LAS	26.11	3kg	7.25	0.36	Consultation. Surfactant based. 20 units
Urinal blocks (non Pdcb)	21.35-35.78	3kg	5.93 – 9.94		Formulated with active cleaning agents, highly perfumed Janitorial Supplies (2012) Gentworks (2012) Alliance UK (2012)
Bio-productions – biological toss blocks	17.18	1.1kg	13.02		Gentworks (2012) Surfactant and bacteria-based.
Maxima Urinal Freshener Blocks	15.68-23.61	3kg	4.36 – 6.56		Viking (2012) Yate Supplies (2012)
Arma Lemon Channel Cubes	17.17	3kg	4.77		Stephensons (2012)
Sachets Lemon Fresh Urinal Blocks	26.19	3kg	7.28		Tool Gurus (2012)
Urinal Channel Blocks (ENOV)	16.80-33.02	3kg	4.67 – 9.17		Janitorial Supplies (2012 Alliance UK (2012)
Jeyes Professional Sanilav Urinal Channel Blocks Ocean	17.11-22.68	3kg	4.75 – 6.30		Covert Cleaning (2012) Click Cleaning (2012) Nexus Cleaning Supplies (2012) Lasts up to 4 weeks. High perfume content

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[11] Prices are expressed in Euro per kg and where available in Euro per unit. The prices in Sterling were converted to Euro using the Bank of England exchange rate (13 February 2012). Prices that included VAT have been adjusted using a rate of 20% which is assumed to be roughly equal to an EU average.

Product name	Price in € (incl. VAT)	Quantity	Price in € per kg (excl. VAT)[1]	Price in € per unit (excl. VAT)	Source/ comments
Lemon Channel Blocks (GreYland)	18.21	3kg	5.06		Harrtops (2012)
Non Pdcb Urinal Blocks Jangro	23.63	3 kg	6.56		Wray Bros (2012a)
Jangro biological blocks	14.63	1.1 kg	11.09		Wray Bros (2012b)
dr becher Gruene (DE)	19.5	35 pieces		0.46	Surfactant based. Hygi.de (2012)
dr becher Standard (DE)	12.9	30 pieces		0.36	Surfactant based. Hygi.de (2012)
Biological product A (DK)		1kg	30.63		RPA (2010)
Biological product B (DK)		1kg	15.31		RPA (2010)
Biological product C (DK)		1kg	7.66		RPA (2010)
ribo bio (DE)	9.05	na	7.54		RPA (2010)
Fresh 40 (CZ)		0.75	11.25		RPA (2010)
Fresh Urinal Toss Block (CZ)	34.65	20 pieces		1.44	RPA (2010)
TOUGH Guy, Super Block, non para	563.64	7kg	67.10		EA Supplies (2012)
Notes: 1) An average EU VAT rate of 20% has been assumed in order to calculate prices excluding VAT.					

According to the RPA (2010) report 1,4-DCB-free urinal blocks ranged in price from €8.75-35/kg including VAT. The updated information suggests for the price range is €4.36-30.63 (€5.23-36.75/kg including VAT) which correlates well with the original research. It should be noted that the high end of the range is represented by a single estimate of €30.63 while the prices of the remaining 18 products are below €15.31. Excluding this estimate from the analysis, the price range is €4.36-15.31/kg. While a wide range of products appear to exist and there are 1,4-DCB-free urinal blocks products on the market that are of the same or even lower price than those containing 1,4-DCB, on average alternative products are more expensive. In particular, the average price of 1,4-DCB-free urinal blocks is €8.15 per kg or € 0.65 per unit (vs. €7.19 per kg or €0.58 per unit[12] for 1,4-DCB containing urinal blocks).

With regards to particular types of alternative products, the RPA (2010) study suggests that the relative costs of 1,4-DCB-free urinal blocks could range between -15% for surfactant-based blocks, up to +50% for enzyme-based blocks and up to +400% for specialised fragrance based blocks. Our own analysis of surfactant-based products, found the average price per unit to be € 0.40 (-30% relative cost to 1,4-DCB-based block). It was not possible, however, to derive average prices per unit for enzyme and fragrance-based blocks. Instead the RPA (2010) assumptions were used for further calculations.

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[12] Calculated assuming 80g per average toilet block as estimated in the RPA (2010) study.

Appendix C
Data for Incorporation into Cost Curves

Table D.1 Summary of measures for inclusion in cost curves

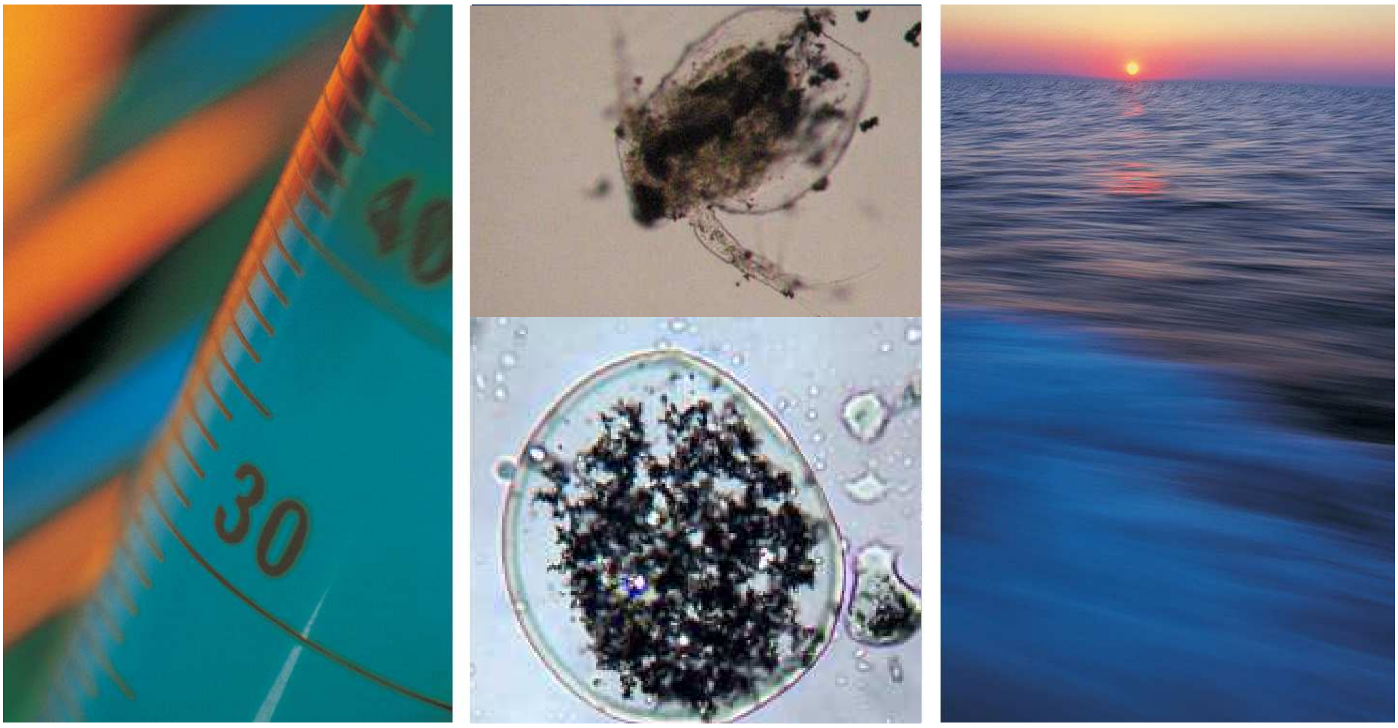
Measure	Details of key elements of measure
Replacement of 1,4-DCB in urinal blocks for professional use	*Annual ongoing costs:* Based on 100% uptake of least-cost option (e.g. surfactant-based blocks at €0.40 per unit). Substitution factor of 3 has been applied which is based on RPA (2010). Total annual ongoing costs = €4.75 million. *Emissions reduced:* 100% per cent substitution of 1,4-DCB urinal blocks for professional use assumed. Total emission reduction = 612.5t of 1,4-DCB. *Applicability of measure:* Assumed 100% of use is replaced with surfactant-based blocks as these are thought to be the least cost option. In practice this is unlikely to be the situation as a range of products are currently available on the market and information from consultation suggests that factors other than price will also affect choice of alternatives. It has not been feasible to assess this in the cost curves. *Specific measures taken forward:* 1,4-DCB_TB_P_1: Replacement of 1,4-DCB in urinal blocks with surfactant-based blocks (100% of use). *Main uncertainties and limitations:* • 100% uptake of least cost option is unlikely to occur in practice. • Substitution factor of alternatives is averaged across all products and based on RPA (2010) report does not reflect variation between individual products. Some alternatives are marketed as being of roughly equal longevity.
Replacement of 1,4-DCB in toilet rim blocks for domestic use	*Annual ongoing costs:* Based on 100% uptake of least-cost options (e.g. bottom-of-the range cistern blocks at €0.16 per unit). Substitution factor of 1 has been applied which is based on RPA (2010). Total annual ongoing costs = - €0.21 million. *Emissions reduced:* 100% per cent substitution of 1,4-DCB toilet rim blocks for domestic use assumed. Total emission reduction = 12.5t of 1,4-DCB. *Applicability of measure:* Assumed 100% of use is replaced with bottom-of-the range cistern blocks as these are thought to be the least cost option. In practice this is unlikely to be the situation as a range of products are currently available on the market and information from consultation suggests that factors other than price will also affect choice of alternatives. It has not been feasible to assess this in the cost curves.. *Specific measures taken forward:* 1,4-DCB_TB_D_1: Replacement of 1,4-DCB in toilet-rim blocks with cistern blocks. *Main uncertainties and limitations* • 100% uptake of least cost option is unlikely to occur in practice.
Replacement of 1,4-DCB in air fresheners for professional use	*Annual ongoing costs:* Based on 100% uptake of least-cost options (e.g. bottom-of-the range aerosol at €0.28 per unit). Longevity of alternative products has been assumed to be the same as 1,4-DCB products. Total annual ongoing costs = - €1.84 million. *Emissions reduced:* 100% per cent substitution of 1,4-DCB-based air fresheners for professional use assumed. Total emission reduction = 100t of 1,4-DCB.

Measure	Details of key elements of measure
	Applicability of measure: Assumed 100% of use is replaced with bottom-of-the range aerosols as these are thought to be the least cost option. In practice this is unlikely to be the situation as a range of products are currently available on the market and information from consultation suggests that factors other than price will also affect choice of alternatives. It has not been feasible to assess this in the cost curves. *Specific measures taken forward:* 1,4-DCB_AF_P_1: Replacement of 1,4-DCB in air fresheners with bottom-of-the-range aerosols. *Main uncertainties and limitations:* • No data on prices of professional air fresheners. Prices of products for domestic use are used based on RPA (2010) study. • 100% uptake of least cost option is unlikely to occur in practice.
Replacement of 1,4-DCB in air fresheners for domestic use	*Annual ongoing costs:* Based on 100% uptake of least-cost options (e.g. bottom-of-the range aerosol at €0.28 per unit). Longevity of alternative products has been assumed to be the same as 1,4-DCB products. Total annual ongoing costs = - €1.53 million. *Emissions reduced:* 100% per cent substitution of 1,4-DCB-based air fresheners for domestic use assumed. Total emission reduction = 83t of 1,4-DCB. *Applicability of measure:* Assumed 100% of use is replaced with bottom-of-the range aerosols as these are thought to be the least cost option. In practice this is unlikely to be the situation as a range of products are currently available on the market and information from consultation suggests that factors other than price will also affect choice of alternatives. It has not been feasible to assess this in the cost curves. *Specific measures taken forward:* 1,4-DCB_AF_P_1: Replacement of 1,4-DCB in air fresheners with bottom-of-the-range aerosols. *Main uncertainties and limitations* • 100% uptake of least cost option is unlikely to occur in practice.

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Lead in shot – Final Report



AMEC Environment & Infrastructure UK Limited

December 2012

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Document Revisions

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1	Draft final report and preliminary results	31 May 2012
2	Final report and results	6 July 2012
3	Amended final report taking into account ECHA comments	20 July 2012
4	Amended final report taking into account ECHA comments	31 October 2012
5	Amended final report taking into account ECHA comments	10 December 2012

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Lead in shot – Final Report

AMEC Environment & Infrastructure
UK Limited

December 2012

Report for

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Executive Summary

This report is one of a series providing the results of work on collection of data on abatement costs of reducing the use and/or environmental emissions of certain hazardous chemicals. The report concerns lead shot used in hunting. The work involved developing abatement cost curves based on information readily available in the literature in combination with consultation with industry and relevant associations. It has been undertaken on behalf of ECHA by AMEC.

The report includes best estimates of the current production volumes of lead shot in Europe. Stakeholder consultation identified seven companies producing lead shot in Europe with a combined production level of between 35,000 and 45,000 tonnes/year.

In total, it has been estimated that around 660 million lead shotgun cartridges are consumed by around 6 million hunters in Europe on an annual basis. This corresponds to an estimated 21,200t of lead emissions to the environment per year. Using recent waterfowl bag data, it is estimated that approximately 7% of hunting takes place on wetlands. This is estimated to result in approximately 360t of lead emissions per year, or around 1.7% of the total lead emissions. The latter figure is lower than the share of total hunting with shotguns that is estimated to take place on wetlands, due to the existence of several partial or total bans on hunting with lead on wetlands within various EU Member States.

The cost curves developed only include costs related to the use of alternative products to replace the use of lead shot for hunting, rather than measures to reduce use or emissions. There are a number of alternative products available on the market including steel shot, tungsten shot and bismuth/tin alloy shot. However these offer different ballistic properties to lead shot and are met with opposition in some quarters.

The cost curves developed illustrate the relative costs, in €/t of use/emission avoided, of substituting lead shot for hunting in wetlands and other areas. The measures assume 100% uptake of the least-cost alternative (i.e. steel shot). In practice, it is likely that a range of different alternatives would be used, depending on users’ preferences, although the majority of non-lead shot currently used is indeed steel.

Total annualised costs of reducing all uses of lead in shot for hunting are estimated at around €190 million per year (this includes both one-off and ongoing costs; it is important to note that the one-off costs are significant and would be incurred up-front following the implementation of any restriction). Total costs at net present value, over twenty five years, are estimated at €2.7 billion. In comparison, the total annualised costs of implementing a ban on the use of lead shot in wetlands are estimated at around €3.3 million per year with a net present value, over twenty five years, of at €45 million.

The cost effectiveness (in terms of cost per unit of lead shot avoided) of implementing a ban on wetlands and in remaining areas has been assumed to be the same for wetland and non-wetland areas (€9.1k/t). However, in reality, the costs of introducing a ban for hunting only on wetlands may be higher, because some of the hunters will only shoot on wetlands part of the time, but may still be required to test and change their guns (and shot) so that they are able to use non-lead shot when on wetlands. Insufficient quantitative data was available to account for this effect.

The measures in the cost curves only include the additional prices of alternative shot and the costs of testing and potentially replacing existing guns to use non-lead alternatives. They do not include other costs, such as investments in new equipment by shot and gun manufacturers (which are assumed to be already reflected in the prices of alternatives) or sunk costs related to the residual value of any capital equipment that is currently used to produce lead shot and which could no longer be usefully employed.

1. Introduction

1.1 The Project

The European Chemicals Agency (ECHA) has commissioned a project to provide information on "abatement costs for certain hazardous chemicals" (contract number ECHA/2011/140). The work is being undertaken by AMEC Environment & Infrastructure UK Limited ("AMEC").

The present report is intended to provide a summary of the data collected on abatement costs of reducing the use of lead in shot. It is one of a series of similar reports covering lead and other hazardous substances.

The data collected is intended to be used for:

- Supporting the Agency in assessing the most appropriate risk management options for the substances addressed;
- Furthering the understanding of the usefulness of data on use/emissions abatement costs in risk management decision-making; and
- Supporting the Agency in the preparation of restriction dossiers.

1.2 Project Context

This project follows on from a 2010 project on "*Abatement cost curves for substances of concern*" conducted by AMEC Environment & Infrastructure UK (previously Entec UK) for the Environment Agency, ECHA, the Health and Safety Executive (HSE) and RIVM. The main aim of that project was to develop a suitable method for estimating abatement costs to reduce emissions of chemicals and to apply and test the method with three selected case study substances. That study provided a first illustration of the benefits of being able to compare unit abatement costs amongst different substances and different uses.

The objective of this project is to assist ECHA in establishing capability to assess the abatement costs of reducing the use or emissions of hazardous substances. Under this lot ("Lot 4"), abatement potential and costs for lead have been assessed, specifically focusing on lead in shot.

Lead is not currently included in Annex XIV (List of substances subject to authorisation) and lead metal used in shot is not restricted under Annex XVII of the REACH Regulation (although some lead compounds are restricted for certain other uses).

The current study has concentrated on assessing abatement costs of direct alternatives (i.e. alternative substances) in order to provide up-to-date information on the costs of restricting the use of lead in shot for hunting purposes. The information might be used to support the possible evaluation of whether such a restriction is feasible or appropriate. However, it is not linked to any specific legislative initiative of ECHA.

The main outputs of the work, for this substance and for the other substances being assessed under different lots are expected to be as follows:

- Data on abatement costs of reducing the use or emissions of the chemical. Different applications of the same substance may/will introduce different abatement costs.

- An overview on the functioning of the markets for the substances in question, including information on prices, amounts of the substance on the markets (including import and export), the relative shares of the substance used for different applications, number of actors involved in the business, as well as possible trends in the relevant market.

1.3 Structure of this Report

This report is structured as follows:

- Section 2 of this report provides an overview of the process of data gathering for this study and presents an overview of the data received;

- Section 3 sets out an overview of the market for lead in shot;

- Section 4 presents the data analysis and resulting abatement cost curves;

- Section 5 presents conclusions.

The appendices to this report include various other background data.

2. Data Collection

2.1 Overview

The aim of the study was to gather abatement costs data for the use of lead in shot[1].

The toxicity impacts of lead on the environment and human health are well known. The environmental effects of lead from spent shotgun and rifle ammunition has been the subject of much research (e.g. Beintema, 2001). A single shotgun cartridge (12-bore calibre) may contain between 100 and 600 pellets. After exiting the barrel, the pellet mass spreads out so that, even if the target is hit, many shot are likely to miss and fall to earth. Spent lead pellets can be ingested by birds and other fauna. Predatory and scavenging birds may ingest shot or bullet fragments embedded in the tissues of prey wounded or killed by hunters. This can lead to acute or chronic lead poisoning. Furthermore, there are concerns that lead from spent ammunition may contaminate the aquatic and terrestrial systems. Lead is thought to fulfil the PBT (persistent, bioaccumulative, toxic) criteria under the REACH Regulation[2].

This chapter briefly summarises the process of data gathering for this study and presents an overview of the data received.

2.2 Review of Existing Data Sources

In 2004, COWI conducted a study on the potential impacts of restricting the marketing and use of lead in ammunition, fishing sinkers and candle wicks. This included an analysis of lead shot.

In the intervening period, a number of Member States have implemented a partial or total ban on lead in shots. A survey of relevant Member State legislation was undertaken in 2009. This has been updated to account for recent developments and is presented in Appendix A.

ECHA also provided AMEC with access to various other data on lead. These data sources have been supplemented with further relevant sources identified during the course of the study.

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[1] Cartridges containing lead can be divided into: lead shot; airgun pellets and bullets. Shot is used in particular for hunting and competition (sports shooting). The same applies to bullets which are also used in military applications. Pellets are used primarily for sports shooting and in a few instances for hunting (COWI, 2004). As agreed at the inception meeting with ECHA, this study is concerned with lead shot for hunting purposes. Shot refers to a spherical pellet of lead which is loaded into a shotgun cartridge. A single shotgun cartridge may contain between 100-600 single pellets. Lead shot used in sport shooting has been excluded from the abatement costs analysis.

[2] It appears, however, that whilst the B and T criteria do apply, the P criterion may not be applicable to lead (Entec, 2011).

2.3 Stakeholder Consultation

A questionnaire to collect the required information was developed in collaboration with ECHA. This was used as a basis for collecting information via telephone and written consultation with:

- Manufacturers and importers of lead shot and associated trade associations;
- Manufacturers and importers of lead-free shot and associated trade associations; and
- Hunting and sport shooting associations.

In total, 25 organisations provided information.

3. General Market Overview

3.1 Lead Shot

3.1.1 Production, Import and Export

Historical Information

As shown in the table below, the principal use of lead metal is in lead-acid batteries and in sheet form for the building industry. Another application of lead is the production of projectiles for firearms. Lead metal is formed into spherical lead shot to be used in both ammunition and steelmaking. Lead has also been used in the manufacture of bullets for many centuries. Lead shot is used in shotgun cartridges for both hunting and competition (sport shooting) purposes. According to Association of European Manufacturers of Sporting Ammunition (AFEMS), approximately half of all lead shot consumed in the EU is used for target shooting and the other half is used for hunting (ILA, 2008).

To manufacture lead shot pellets, lead ingots with a low melting point are melted first of all. The molten lead is then poured over sieves with pre-determined hole sizes. The lead drops from the sieve and forms into a ball as it drops from between 40 and 55 metres into 1 metre of water in a container. The container is emptied and the lead balls are run over glass to sort them into different size categories. This process makes most of the sizes needed for shot for both clay and game shooting. The larger sizes are pressed from lead wire. The shot pellets are subsequently loaded into a case (along with the propellant, wadding system and primer) to make the shotgun cartridge.

As indicated in Table 3.1, the International Lead and Zinc Study Group (ILZSG) (2002) estimated that the total consumption of lead in 2000 for ammunition in the EU was 56,600 t/year. According to COWI (2004), this consumption decreased to approximately 38,600 t/year of lead in 2003. Lead shot ammunition was estimated to account for about 34,600t of lead (with 17,920t for hunting and the remainder for sport shooting). Lead bullets and pellets accounted for the remaining 4,000 tonnes.

An update has been provided by ILZSG for this study (see Table 3.2). This table suggests a reduction in lead used for shot and ammunition purposes (a reduction of approximately 15% from 2007 to 2011). However, annual data show some fluctuations and not all of the data reported are consistent in terms of the specific end-uses and Member States covered.

Table 3.1 Estimated use of lead by sector in the EU (ILZSG, 2002)

End-use	2000 (tonnes)	%
Batteries	1,008,900	61
Rolled and extruded products	242,400	14
Pigments and other compounds	200,800	12
Shot/ammunition	**56,600**	**3**
Alloys	39,600	2
Cable sheathing	31,300	2
Gasoline additives	19,400	1
Miscellaneous	78,200	5

Table 3.2 EU consumption of lead for shot/ammunition (ILZSG, 2012)

Year	Tonnes/year
2007	57,876
2008	54,390
2009	45,018
2010	49,117
2011	48,937

Note: The figures above reflect the average end use of lead shot not only in ammunition but also for free-machining steels and special steels.

Current Best Estimates

Table 3.3 shows current best estimates (based on submitted registration dossiers and consultation with industry) of total volumes of lead shot manufactured in, imported to and exported from the EU per year. Based on the information available, production of lead shot in the EU was estimated to be estimated to be between 35,000 and 45,000t in 2011. Import to the EU was estimated to be 50t for 2011. Export to customers outside of the EU is thought to be approximately 5,000t/year.

Table 3.3 Summary of manufactured, imported, exported and used quantities

	Quantity	Notes
Lead shot manufactured in the EU, t/year	35,000 – 45,000	Consultation with industry
Number of EU lead shot producers	7	Seven companies have been identified during the course of this study through the association AFEMS and online searches. COWI (2004) previously identified eight companies. Consultation with AFEMS, suggests that three previous producers have since left the lead market and two new companies have entered.
Imports into the EU, t/year	50	Consultation with industry
Exports from the EU, t/year	5,000	Consultation with industry
Net EU supply, t/year	30,050 – 40,050	Consultation with industry

3.2 Availability of Substitutes for Lead Shot

3.2.1 Introduction

Alternatives to lead in shot include:

- Steel: Steel shot is the most widely available alternative to lead shot. However, the ballistic properties of lead and steel differ significantly (this is expanded upon in Table 3.4);

- Tungsten composites: Tungsten has been used as a lead-replacement constituent in ammunition. At 19.3 g/cm^3, it is denser than lead (11.5 g/cm^3); and

- Bismuth/tin alloy: Bismuth (Bi) is another alternative to lead in ammunition but requires the addition of a small amount of tin to reduce its brittleness (Fera, 2010). According to Fera (*ibid*), its performance is comparable to that of lead in shotgun ammunition.

Other metals and composites materials like zinc and molybdenum have also been developed but seem to not to have penetrated the market significantly[3]. According to AFEMS, zinc has poor ballistic properties owing to its weight (which is 40% lower than lead, meaning there is a high risk of wounding animals) and hardness (zinc is 70% harder than lead which leads to a high risk of ricochet). The price of zinc is between 5 and 8 times higher than the cost of lead shot ammunition. Furthermore, zinc has been found to be toxic to birds when ingested (e.g. Levengood et al., 1999) and has therefore been prohibited from use in hunting in some Member States (such as the Flemish region of Belgium) and actively discouraged in others[4]. Only two (Austria and Spain) out of the ten

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[3] Information was received from one company based in Germany that previously produced a zinc/tin alloy shot. However, production was stopped due to insufficient market demand.

[4] Consultation with FACE, July 2012.

Member States that responded during consultation for this study listed zinc as an available alternative. Furthermore, it is not approved as non-toxic in the USA and Canada. It is therefore not considered further in this study.

Therefore, steel, bismuth/tin alloy and tungsten have been assessed as potentially feasible alternatives in this study.

According to KEMI (2008), the available evidence indicates that bismuth and steel shots are less toxic to birds and mammals in comparison to lead shots. Tungsten shots have been found to be less toxic compared to lead shots for birds ingesting shots, although some studies indicate that tungsten shots may give rise to tumours in mammals.

3.2.2 Technical Issues

According to FACE[5], the quality (and cost) of non-lead alternatives has improved over the last 10-15 years, although lead is still considered superior by many hunters due to its ballistic qualities, usability in older shotguns and price of cartridges.

Table 3.4 summarises the key differences between lead, steel, bismuth and tungsten shot in terms of density, hardness, ballistic properties, gun suitability and comparative advantages and disadvantages. This information has been compiled from available literature and information provided during stakeholder consultation.

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[5] Consultation with FACE, May 2012.

Table 3.4 Alternatives to lead in shot, summary of key technical information

Material	Density (g/cm^3)	Hardness (HV)	Ballistic properties	Gun suitability	Advantages	Disadvantages
Lead	11.3	20	Given its high density, it delivers maximum striking energy with the least surface area and air resistance.	Lead has been the favoured choice of metal by hunters for many years and therefore guns have been designed to be compatible with lead.	Lead shot is preferred for its high density which means more energy for penetrating into the target which results in less wounds (hunting) and better breakage (clay target).	Concerns regarding the toxicity impacts of lead on environmental health. Any fauna that ingests spent lead may be at risk of lead poisoning. There are also concerns regarding the fate and behaviour of lead in the aquatic environment.
Steel	7.9	100	Steel shot is lighter and harder than lead shot and therefore ballistic properties are significantly different. It has less pellet deformation, denser patterning, shorter shot strings and lower retained energy at long ranges. However, using larger size steel shot and shortening shooting distance can overcome these issues. According to FACE UK "performance has evolved, along with acceptability".	Risk of damaging gun barrels particularly in older guns with thin-walled barrels. According to FACE, this is a serious concern in Eastern European Member States. This is particularly the case for Russian manufactured shotguns which have fixed chokes with full or ¾ choke in the barrels. For steel shot it is recommended not to use more than half choke. Most older shotguns are not proofed for higher pressures. Under CIP rules[1] "Standard" steel is suitable for use in most nitro-proved guns, except some e.g. Damascus barrelled. "High Performance" steel is only for use in steel-shot proved guns.	Available in wide range of loadings (21-63g). Comparable price to lead (see Table 3.6).	Hardness requires care in use and compatibility with guns so compliance with CIP guidance is necessary. May not be compatible with some old/Damascus-barrelled guns. Hardness requires containment in plastic wads which may limit usage of steel cartridges by locality (especially clay target shooting). Furthermore, due to the hardness of steel, the production process is more energy-demanding and expensive. Hardness also means increased chance of ricochet on hard surfaces; However, according to FACE UK concerns regarding ricochet are "exaggerated".
Bismuth/ tin	9.6-9.8	20	Generally good, provided shot size is increased to allow for lower density than lead.	Suitable in all appropriately-proved guns.	Can be used as though lead without concerns over compatibility with guns. Available in most gauges and a wide variety of loadings.	Major disadvantage is price (see Table 3.6).

Material	Density (g/cm^3)	Hardness (HV)	Ballistic properties	Gun suitability	Advantages	Disadvantages
Tungsten	10-2.5	~8	Density of tungsten makes for good ballistics/performance, so percentage of tungsten in shot material is important.	Suitable in appropriately-proved guns but earlier hard types need to be treated as steel under CIP.	Good ballistics/performance, depending on density, from most current types, with no impacts on gun except for particularly hard earlier types. Density factors can allow smaller-than-usual shot size thereby increasing shot number and pattern density.	Earlier or some imported hard types must be treated as steel under CIP and care taken in use. Major disadvantage is price (see Table 3.6). Spain have noted in their response that tungsten alloys are not available domestically as feasible alternatives currently – no further information was available on this.

This information is largely drawn from responses from consultation through FACE and their members.

1) The Permanent International Commission for Firearms Testing (CIP) is an international organisation whose members are 14 governments, mainly European. The CIP safeguards that every civil firearm and all ammunition sold in CIP member states are safe for the users. Furthermore, the CIP also enforces the approval of all ammunition a manufacturer or importer intends to sell in any of the CIP member states

3.2.3 Quantities

According to available literature and consultation, there is only one European company currently producing steel shot, based in Germany. This is in line with the COWI (2004) report which identified one steel shot manufacturer. No specific information was available on quantities produced. According to consultation for the current study, the vast majority of steel shot is currently imported from China. However, AFEMS estimates that, of the 720 million shotgun cartridges consumed per year in Europe by hunters, only 8% (60 million) are lead-free alternatives (this is expanded upon in greater detail in section 4.1.2). According to the association, these cartridges are assumed to all be steel-shot cartridges, as the other alternative materials are practically non-existent on the market.

One company based in Germany was previously a producer of zinc shot. There were three products available: Grillo-Zinc-Shot (ZnSn2), Grillo-Shot (SnZn50) and Grillo-Woodshot (SnZn30)[6]. However, these were all withdrawn from the market, reportedly following poor demand. No other companies have been identified as currently producing zinc pellets for cartridges.

Over the course of the study, one European company was identified as producing bismuth pellets for use in ammunition. No information was available on quantities produced.

One European company has been identified as producing tungsten pellets as an alternative to lead pellets. This company states that this is an “insignificant” volume (no further detail was provided). No other companies have been identified as currently producing tungsten shot.

Overall, this seems to corroborate the view that the main alternative currently available and in use is steel shot.

3.3 Prices

Based on consultation with industry, prices vary enormously from country to country and over time as metal prices fluctuate on the financial markets. It is therefore difficult to assign average prices. According to one shotshell[7] producer, as a general rule, in game hunting, both lead and steel are the same price, and the other alternatives can be 6 to 20 times more expensive; whilst in sports shooting, lead cartridges can be more expensive than steel, but much less expensive than bismuth. Prices vary depending on gauge[8], load and shot size.

The UK hunting associations (through FACE) have provided an in-depth analysis on current prices of cartridges (both lead and non-lead alternatives) based on the four main cartridge suppliers in the UK[9]. The price information

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[6] http://www.grillo.de/index.php?id=64&L=2

[7] A shotgun shell (shotshell) is a self-contained cartridge loaded with shot or shotgun slug designed to be fired from a shotgun.

[8] Shotgun shells are generally measured by “gauge”. However in the UK and some other locations the term “bore” is used. The gauge is the weight, in fractions of a pound, of a pure lead round ball that is the same diameter as the internal diameter of the barrel. A shotgun is called 12-gauge because a lead sphere that just fits the inside diameter of the barrel weighs 1/12 of a pound.

[9] Data were taken from www.justcartridges.com for the four main cartridge companies in the UK (Eley, Hull, Gamebore and Lyalvale Express).

on lead cartridges is summarised in the table below. Other information provided during consultation on prices is provided in Appendix B, covering data from Austria, Belgium (Walloon Region), Greece, Italy, Malta, Slovenia, Spain and Sweden.

Table 3.5 Prices of 12-gauge lead shotgun cartridges (FACE UK, 2012)

Load, grams	Price range, €/cartridge (VAT excluded)	Average price, €/cartridge (VAT excluded)
21	0.19 - 0.21	0.20
24	0.21 - 0.34	0.28
26	0.30 - 0.34	0.32
28	0.21 - 0.37	0.29
30	0.26 - 0.42	0.34
32	0.27 - 0.41	0.34

Note: According to COWI (2004), whilst lead content can range from 20-50+ grammes, 32 grammes is assumed to be the average load for all hunting shotgun cartridges. Therefore €0.34 is used as the average price of a lead shogun cartridge. [Prices were originally provided by FACE including VAT; prices have been reduced so as to exclude VAT, assuming a VAT rate of 20%].

Table 3.6 presents the relative price of non-lead alternatives compared to lead. Please note that the second column in the table presents historic information from COWI's 2004 study. The 2004 data are not used in the subsequent calculations and are only included to show historical changes in the absolute and relative price of lead shot and of the alternatives for comparison with 2012 data.

Table 3.6 Prices of substitutes relative to lead

Material	Price of shot relative to lead shot (COWI, 2004)	Price of shot relative to lead shot, based on consultation (2012)
Lead	100%[1]	100%
Iron/steel	120%	100 - 120% [2]
Bismuth	300-500%	400 - 500%
Tungsten	300-1000%	400 - 700%

1) The price of 1 lead shot cartridge was assumed to be in the range of €0.13 - 0.27.

2) When equal loads are compared, the steel cartridge prices match those of lead. However, as noted in Table 3.4, because of the difference in ballistic properties it is necessary to use a larger size steel shot.

3.4 Trends

Consultation with industry suggests that production of lead shot has decreased in recent years and that this trend is expected to continue in future years. This is thought to be attributable to the various restrictions introduced on the use of lead shot for hunting in many Member States (see Appendix A).

Hunter numbers over the past decade appear to have remained relatively stable. The COWI (2004) study reported a total of 6.2m hunters in the EU-15. The available data collected during the course of this study estimates a total of 6.1m hunters in the EU-27. According to FACE, over a longer time period there have been declines, however over the past five it appears to have stabilised with some increases in countries such as Germany and Poland. There appear to be regional variations, for instance Mondain-Moval et al. (2012) note that in the Camargue region of France hunter numbers (particularly on public land) have decreased significantly.

In terms of alternatives, the use of steel shot has increased over the last decade, now accounting for approximately 8% of all shotgun cartridges used in Europe. The other alternatives, however, have had very limited market penetration. Full or partial bans on the use of lead shots on wetlands that are operational in 16 Member States represent a significant driver for substituting lead shots previously used in hunting on wetlands with alternatives. As these restrictions are further enforced and continue to come into effect, an increase in the penetration of non-lead shot may be expected.

4. Data Analysis

4.1 Uses and Releases

According to AECAC[10] there are more than 12 million lead shot users in Europe including hunters and sport shooters.

4.1.1 Number of Hunters

An earlier estimate of the number of hunters made by FACE was 7 million hunters in the EU (COWI, 2004). FACE requested an update from their associated national bodies for the current study. The information collected is summarised in Table 4.1.

Table 4.1 Number of hunters in the EU

Country	Number of hunters	Number of hunters using shotguns	Average number of shotguns per hunter
Austria	120,305	90,000	2
Belgium (Flanders)	16,000	16,000	1
Belgium (Region Wallonne)	18,000	18,000	2
Bulgaria	No data received		
Czech Republic	No data received		
Cyprus	No data received		
Denmark	165,000	165,000*	
Estonia	15,000	15,000	
Finland	310,000	300,000	2
France	1,230,000	1,168,500	1.4
Germany	241,098	241,000	
Greece	200,000	200,000	1.8-2.5
Hungary	58,000	54,966	1.44
Ireland	300,000	88,000	2
Italy	750,000	750,000	
Latvia	25,000	25,000	
Lithuania	No data received		

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[10] The European Association of the Civil Commerce of Weapons.

Country	Number of hunters	Number of hunters using shotguns	Average number of shotguns per hunter
Luxembourg	2,000	2000	
Malta	11,500	11,500	2
The Netherlands	27,500	27,500	
Poland	110,000	110,000*	
Portugal	241,560	210,000	1.65
Romania	60,000	60,000	
Slovakia	58,000	58,000*	1.4
Slovenia	21,040	21,000*	
Spain	850,000	850,000	3
Sweden	300,000	250,000	1.2-1.5
UK	480,000	480,000	2.4
Total	**5,610,003**	**5,211,466**	
EU Average	243,913	226,585	**2.02**
Adjusted to EU-27	**6,585,656**	**6,117,808**	

* This is assumed in the absence of further data

The number of hunters is estimated at 5.6 million hunters across 23 Member States. France, Italy, Spain and the UK are the four countries with the highest numbers of hunters.

Information is missing for four Member States including Bulgaria, Cyprus, Czech Republic and Lithuania. The EU-27 total has been calculated based on the average hunter numbers in 23 Member States. This results in an estimate of 6.6 million hunters which is broadly in line with the previous estimate provided by FACE. Of these, 6.1 million are assumed to be using shotguns. This estimate has been used to derive the abatement cost curves.

4.1.2 Consumption of Shot

According to AFEMS, the trend of lead shot usage over the last 10 years has been near to stable, although there have been fluctuations as highlighted earlier in the report, and other data suggest a moderate decline in use. Steel shot usage has had a slowly positive trend, arriving near to 8% of the total consumption. The other materials which constitute possible alternatives to lead shot have not had any significant influence on consumption volumes.

Data received during the consultation is summarised in the table below.

Table 4.2 Quantity of lead-based and lead-free shots used by hunters (AFEMS, 2010)

Column Heading	Quantity of lead based shots used per year, tonnage or units or %		Quantity of lead-free alternative shots (e.g. steel) used per year, tonnage or units or %		Total
	Cartridges	**Tonnage**	**Cartridges**	**Tonnage**	**Cartridges**
Hunters	663 million (91.7%)	21,223t	60.3 million (8.3%)	1,830t	720 million

4.1.3 Bag Data

Total bag

The total number of wildfowl bagged in Europe is estimated at approximately 103 million birds per year (see Table 4.7). However, lead shot is also used to kill small mammals such as foxes and rabbits. Only partial information has been received from Member States on small mammals shot. Therefore it has not been possible to estimate the total numbers of small mammals bagged in the EU.

Table 4.3 Annual shooting bag figures by country (2005)

Member State	Number of birds shot annually
Austria	284,904
Belgium	1,175,326
Bulgaria	No data
Cyprus	669,250
Czech Republic	988,361
Denmark	2,150,265
Estonia	21,804
Finland	1,173,000
France	25,676,403
Germany	2,299,984
Greece	10,025,871
Hungary	688,910
Ireland	3,058,046
Italy	17,054,468

Member State	**Number of birds shot annually**
Latvia	44,261
Lithuania	14,765
Luxembourg	2,903
Malta	397,690
Netherlands	1,022,300
Poland	284,490
Portugal	No data
Slovakia	171,198
Slovenia	50,834
Spain	11,147,285
Sweden	553,734
Switzerland	38,285
United Kingdom	22,149,024
Total	**101,143,361**
Adjusted to EU-27	**102,915,283**

Source: Hirschfeld & Heyd (2005) (from CABS website)

Waterfowl

A previous estimate of the number of waterfowl killed is 1.5 million across the EU (Matteo, n.d). However, the Committee Against Bird Slaughter (CABS) estimates the total number of waterfowl bagged in Europe as around 6.9 million birds per year (see Table 4.4).

Table 4.4 Official numbers of waterfowl (*Anseriformes*) shot annually in Europe

Species	**Quantity bagged per year**
Mallard	4,524,449
Common Teal	960,027
Wigeon	849,839
Goldeneye	152,618
Merganser	4,769
Tufted Duck	No data
Common Eider	104,495

Species	Quantity bagged per year
Long-Tailed Duck	27,953
Common Scoter	23,737
Greylag Goose	107,813
Canada Goose	64,297
Pink-footed Goose	20,208
Taiga Bean Goose	35,914
Total	**6,876,119**

Source: CABS website: http://www.komitee.de/en/projects/hunting-bags

Further data were collated by FACE on the number of waterfowl killed. This is summarised in the table below. The CABS data is taken as being the most complete dataset and is used in the remainder of the analysis.

Table 4.5 Data provided by Member States during consultation

Member State	Number of waterfowl shot	Notes
Austria	80,000	
Belgium (Region Wallonne)		No data: 5-15% of total hunting
Belgium (Flanders)	23,291	2006/2007
Denmark	914,600	1999/2000
Estonia	22,606	
Finland	533,000	
France		No data: 10.5% of total hunting
Germany	65,617	
Greece		No data: 2.2% of total hunting
Hungary	55,026	
The Netherlands	251,372	2006/07
Poland	116,248	
Slovakia		No data: 14% of all feathered game
Slovenia	1,493	
Sweden	196,000	
UK	1,017,000	

Member State	Number of waterfowl shot	Notes
Total	**3,024,881**	
EU-27 (adjusted)	**5,796,637**	This has been adjusted based on available hunter numbers in each Member State

4.1.4 Environmental Releases

Releases of lead from hunting can be estimated in one of two ways:

1. Using annual bag data and average number of cartridges fired per animal; or
2. Using total number of cartridges consumed per year and average lead content.

In the absence of data on total numbers of mammals shot annually in Europe, total emissions of lead from hunting in the EU have been calculated using the second approach outlined above. In total, approximately 21,000 t/year of lead are estimated to be released from hunting in the EU (see Table 4.6).

Based on available waterfowl bag data, it is estimated that currently around 6.7% of hunting takes place in wetland areas in the EU. Emissions of lead from hunting in wetlands has been calculated using information available on the number of hunters per Member State and number of Member States that have introduced legislation prohibiting the use of lead shot in wetlands (16 countries in total – see Section 4.2).

No data are available on the extent to which lead is used on wetland and non-wetland areas, or on how this varies amongst Member States and across the EU. However, it is assumed that the existing restrictions (which often prohibit use of lead shot on wetlands) mean that more non-lead shot will be used on wetlands and less on non-wetland areas as a proportion of the total (i.e. lead shot use on wetlands will be less as a percentage of total lead use than the assumed 6.7% of hunting that takes place on wetlands). In the absence of better data, to estimate the amount of lead that is used on wetlands, the following approach was taken:

- For Member States with a full ban on wetlands, it was assumed that none of the hunters shoot with lead on wetlands[11].
- For Member States with a partial ban, it was assumed that 50% of shooting on wetlands uses lead.
- For Member States with no ban, it was assumed that lead is used at the same level as the average EU proportion of shooting that takes place on wetlands (6.7%) and that all hunters can use lead.

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[11] There is evidence to suggest that compliance with existing restrictions in some Member States is (possibly substantially) less than 100% and this is explored later as a sensitivity.

In total, approximately 360 t/year of lead are estimated to be released from hunting in wetland areas in the EU (see Table 4.6). .

Table 4.6 Total lead emissions from hunting

	Quantity	Notes / source
Number of lead cartridges consumed by EU hunters, cartridges/year	663,000,000	AFEMS (2010)
Lead content per cartridge, kg/cartridge	0.032	COWI (2004)
Annual shooting bag figures, birds/year	102,915,283	Hirschfield and Heyd (2005)
Number of waterfowl bagged, birds/year	6,876,119	CABS website. [It is assumed that all waterfowl are shot in wetland areas.]
Proportion of hunting that takes place on wetlands	6.7%	Based on available bag data. COWI (2004) estimated this figure at 20%.
Total EU emissions of lead from hunting, t/year	**21,216**	COWI (2004) estimated the total amount of lead released from hunting between 17,100-18,000t in the EU-15. The estimate in this study is for the whole EU-27 region.
Emissions of lead from hunting in wetlands, t/year	**357**	Based on the number of hunters per Member State, data on full and partial bans on the use of lead shot on wetlands (use of steel is assumed). For the remaining Member States, it is assumed that the proportion of lead used on wetlands is the same as the EU average for hunting on wetlands (6.7%). Note: COWI (2004) estimated the release of lead to the environment from hunting on wetlands between 3,400-3,600t in the EU-15. The approach adopted in this study assumes a much lower percentage (6.7%) of hunting taking place on wetland areas.
Emissions of lead from hunting in non-wetland areas, t/year	**20,859**	

4.2 Current and Planned Abatement Measures

4.2.1 Existing Measures

Measures for the control of lead emissions are extensive. These measures include:

- Regulation (EC) No 1907/2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH).
- Directive 2008/50/EC on ambient air quality and cleaner air for Europe sets an EU-wide minimal air quality standard for lead of 0.5µg/m^3 measured as an annual average.

- Directive 80/68/EEC on the protection of groundwater against pollution caused by certain dangerous substances, to be repealed in 2014 by Directive 2006/118/EC on the protection of groundwater against pollution and deterioration.
- Directive 86/278/EEC on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture contains a limit value for lead in sludge for use on land.
- IPPC Directive 2008/1/EC concerning integrated pollution prevention and control.
- Regulation (EC) No 1223/2009 on cosmetic products.
- Directive 2006/66/EC sets collection and recycling targets for batteries and accumulators including the prohibition of final disposal of automotive and industrial batteries into landfill and incineration and a target recycling efficiency of at least 65% by average weight of lead-acid batteries and accumulators, including recycling of the lead, to be met by 2011.
- Directive 98/83/EC (drinking water) on the quality of water intended for human consumption contains quality standards for lead. EU-wide guideline for Pb in drinking water is 10 µg Pb/L.
- Directive 2006/11/EC on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community.
- Directive 2000/53/EC on end-of life vehicles.
- Decision 2000/479/EC (EPER) Lead is part of EPER (European Pollutant Emission Register).
- Water Framework Directive 2000/60/EC and EQS Directive 2008/105/EC: EU EQS for lead in surface waters of 7.2 µg Pb/l (annual average). Lead is a Priority Substance, requiring progressive reduction in emissions, discharges and loses to surface water.
- Directive 75/442/EEC on waste, Directive 91/689/EEC on hazardous waste and the related Decisions 2000/532 and 2001/118.
- Directive 1999/31/EC on the landfill of waste with general provisions with regard to the control of discharges.
- Directive 2000/76/EC on waste incineration contains specific provisions with regard to lead emissions. Directive 2000/76/EC applies to virtually all waste incineration and co-incineration plants and sets (amongst other things) stringent limits on emissions of certain pollutants. For lead and its compounds, the emission limit values are:
  - A limit value for emissions to air of 0.5 mg/m3 (1 mg/m^3 until the end of 2007 for hazardous waste installations permitted before the end of 1996); and,
  - A limit value for discharges of wastewater from the cleaning of exhaust gases of 0.2 mg/l.

- Directive 2002/95/EC on the restriction of the use of certain hazardous substances in electrical and electronic equipment (RoHS) and Directive 2002/96/EC on waste electrical and electronic equipment (WEEE).

- Directive 98/70/EC relating to the quality of petrol and diesel fuels, sets environmental fuel quality specifications to be applied (with effect from 1 January 2000 and 1 January 2005) to fuels for vehicles equipped with positive ignition engines (petrol) and with compression ignition engines (diesel). The Directive led to the ban of sales of leaded petrol in the EU, although tetra ethyl lead additive is still produced within the EU for sale outside of the European market. This has, in turn, led to a dramatic decline in ambient lead levels in the EU environment.

- Directive 2001/80/EC on large combustion plants (LCPs). The Directive is intended to reduce emissions of acidifying pollutants, particles, and ozone precursors. It requires significant reductions in emissions from existing plants (those licensed before 1 July 1987) by 1 January 2008; either through compliance with emission limit values (ELV) for NO_x, SO_2 and dust or through a national emission reduction plan that achieves an equivalent overall reduction. Whilst this does not set emission limit values for lead, the abatement techniques expected to be applied for reducing dust emissions are expected to have a significant impact on reducing emissions of heavy metals, including lead.

More specifically with regards to the use of lead in ordnance, the African-Eurasian Migratory Waterbird Agreement (AEWA) calls for the phase out of lead shot for hunting in wetlands. Details may be found in Appendix A, but an overview from the 27 EU Member States is summarised below:

- 1 Member State, Denmark, has a ban on the use and trade of lead shot (Ordnance 1998:944).

- 2 Member States, the Netherlands, plus the Flemish region of Belgium have banned all use of lead shot for hunting and sport shooting. [Note: the Walloon Region of Belgium has a ban on the use of lead shot in wetlands.]

- 14 Member States (BG, CY, CZ, HU, IT, ES, FI, FR, LV, PT, SE, UK) have banned lead shot either on wetlands or for waterfowl hunting on the whole of their territory or on part of their territory (DE, BE).

- 5 Member States (AT, EE, GR, LU, SK) are expected to implement a ban on the use of lead shot which will enter into force in the near future.

- In 3 Member States (IE, LT, SI) discussions are underway to take legal measures to ban the use of lead shot for hunting in wetlands.

- 3 Member States (MT, PL, RO) have no ban on lead shot. Malta has no wetlands where hunting is permitted. FACE is engaging with their Member organisations in Poland and Romania to initiate progress towards phasing out the use of lead shot in wetlands.

4.3 Possible Future Abatement Measures

4.3.1 Range of Possible Measures Available

KemI (2008) considers a number of economic policy instruments to encourage a shift away from lead-shot ammunition in Sweden. The report considers the introduction of a tax, or premium, on the use of lead ammunition; tradable quotas for lead; a deposit-refund system; subsidies for alternative materials; subsidies for bullet/shot retrieval arrangements and subsidies for cleaning up contaminated sites.

It was concluded by KemI that the first four instruments aim at a significant but not total shift to alternatives. The quota system was judged to be the most difficult to implement due to administrative reasons. The deposit refund system was considered difficult to implement as the different metals will be mixed up during retrieval. Subsidies for alternative materials would have a similar affect to that of a tax, but as a single policy instrument would not be in accordance with the polluter pays principle. The study concludes that a tax on lead in ammunition would be the most effective to stimulate increased use of alternative ammunition.

At an EU-wide level, two possible risk management options (RMOs) have been considered within the scope of this study:

- General restriction on the use of lead shot for hunting;
- Restriction on the use of lead shot for hunting in wetland areas.

It is beyond the scope of this study to compare the merits of the economic instruments suggested by Kemi (2008) with a potential EU-wide restriction.

Appendix C of this report provides details on the data sources and assumptions used in assessing the potential for each of the measures to reduce use of the substance and the associated costs.

4.3.2 Measures Included in the Cost Curve

For the purposes of the abatement cost curve development, it has been assumed that lead shot would be substituted by the least cost-alternative. In practice, it is likely that a range of different alternatives would be replace some of the lead shot use, depending on users' preferences. COWI (2004) assumed that lead shot would be substituted with a mix of alternatives – 50% steel, 20% tungsten alloy, 20% bismuth and 10% tin. However, for simplicity, the least-cost option (steel shot) has been taken forward as part of the cost curve development. This is borne out by the market data for alternatives in Section 3.2 which indicates that steel shot seems to be the only alternative adopted in significant quantities.

4.4 Cost Curves for Lead Shot used for Hunting Purposes

Table 4.3 provides a summary of the key data on each of the measures assessed. Further details are included in the supplementary spreadsheet.

Figure 4.1 presents the cost curve, showing cumulative costs compared to emissions/use abated. Figure 4.2 presents marginal costs.

As shown in the graphs, the cost effectiveness of implementing a ban on wetlands and in remaining areas has been assumed to be the same (€9.1k/t) for the two options. However, implementing a ban on wetlands only reduces emissions of lead to the environment by approximately 0.4kt (of an estimated total of 21kt). This is because of the significant restrictions already in place on the use of lead on wetlands.

We have attempted to calculate all costs in line with ECHA Guidance on Compliance Costs[12]. However, it is important to note that the cost curves are based on the current prices of alternatives that are already on the market. In practice, one of two outcomes is possible for companies that currently manufacture lead shot in the EU, either:

1. They will cease this area of their business. It is understood that the existing technology used for lead shot manufacturing cannot be adapted to alternative metals (COWI, 2004)[13]. Furthermore, having regard to the insignificant role of exports, this equipment is therefore assumed to have no alternative uses and its opportunity cost is effectively zero. Any residual value which the equipment would have prior to the imposition of a possible restriction has not been included in the costs curves. This residual value, however, is assumed to be small given that the lead shot manufacturing process (described in Section 3) is fairly simplistic and does not involve complex machinery. Whilst no specific data have been obtained for the current study, the residual value of this equipment is likely to be low, particularly when compared to the value of the lead shot market: with the current annual EU consumption of around 660 million cartridges at a price of €0.34 per cartridge, the total sales value per year is €225 million.

2. Or, they would make necessary investments to supply alternatives. It is assumed that the costs of such investments are reflected in the current prices of the alternatives that are already on the market so these investment costs are not included in the abatement cost data.

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[12] Addendum to the Guidance on Socio-economic Analysis – Restrictions: Calculation of compliance costs. Available at: http://echa.europa.eu/documents/10162/17087/appendix1-calculation__compliance_costs_case_restrictions_en.pdf.

[13] According to COWI (2004): "The manufacturing of steel shot is distinctly different from lead shot. The machinery needed is generally more complex and is entirely different from lead manufacturing. In general neither machinery nor know how of lead manufacturing processes can be transferred to lead manufacturing of substitute metals." Furthermore, "the companies in question are typically specialised in lead processing and therefore would have no basis for entering new markets (e.g. production of steel shot)".

Table 4.7 Summary of cost curve data for possible ban on lead in shots for hunting on wetlands and for remaining areas

Measure	Overall total equivalent annual cost, k€	Overall total emission reduction, t Pb	Single measure cost-effectiveness, k€/t	Incremental cost, k€	Incremental emission reduction, t Pb
Ban on lead in shots (hunting/wetlands)	3,260	357	9.1	3,260	357
Ban on lead in shots (hunting/remaining)	190,274	20,859	9.1	190,274	20,859

Notes:

1. k€ is used to represent 1000€
2. All data on costs and emission reductions are at an EU level. All costs have been calculated in line with ECHA Guidance on Compliance Costs[14]. Incremental costs and emission reductions are the same as the overall values because the measures are considered separately. The total potential emission reduction is 21,216t and the total cost €193,534k.

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[14] Addendum to the Guidance on Socio-economic Analysis – Restrictions: Calculation of compliance costs. Available at http://echa.europa.eu/documents/10162/17087/appendix1-calculation__compliance_costs_case_restrictions_en.pdf.

Figure 4.1 Cumulative cost curve for lead shot for hunting

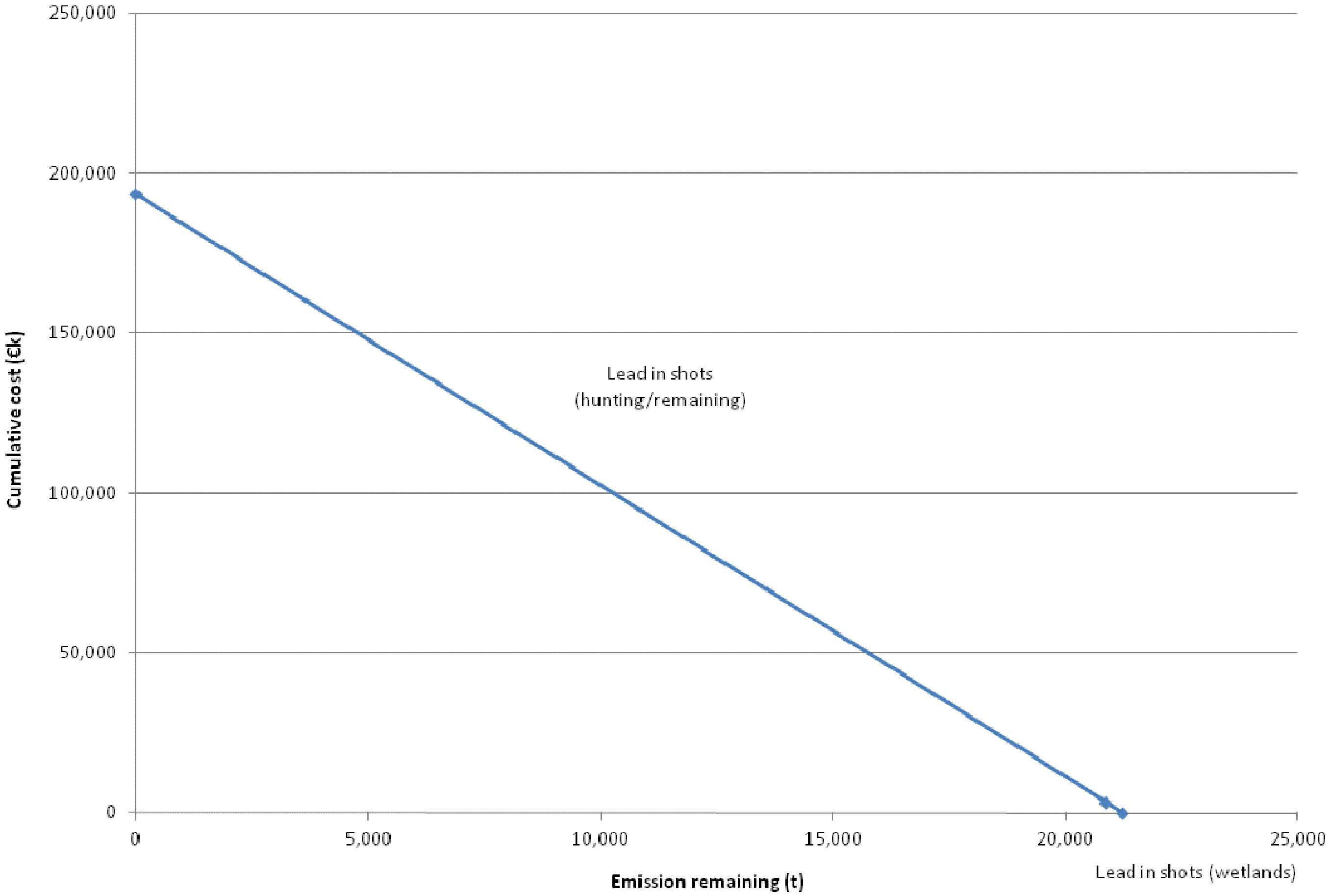
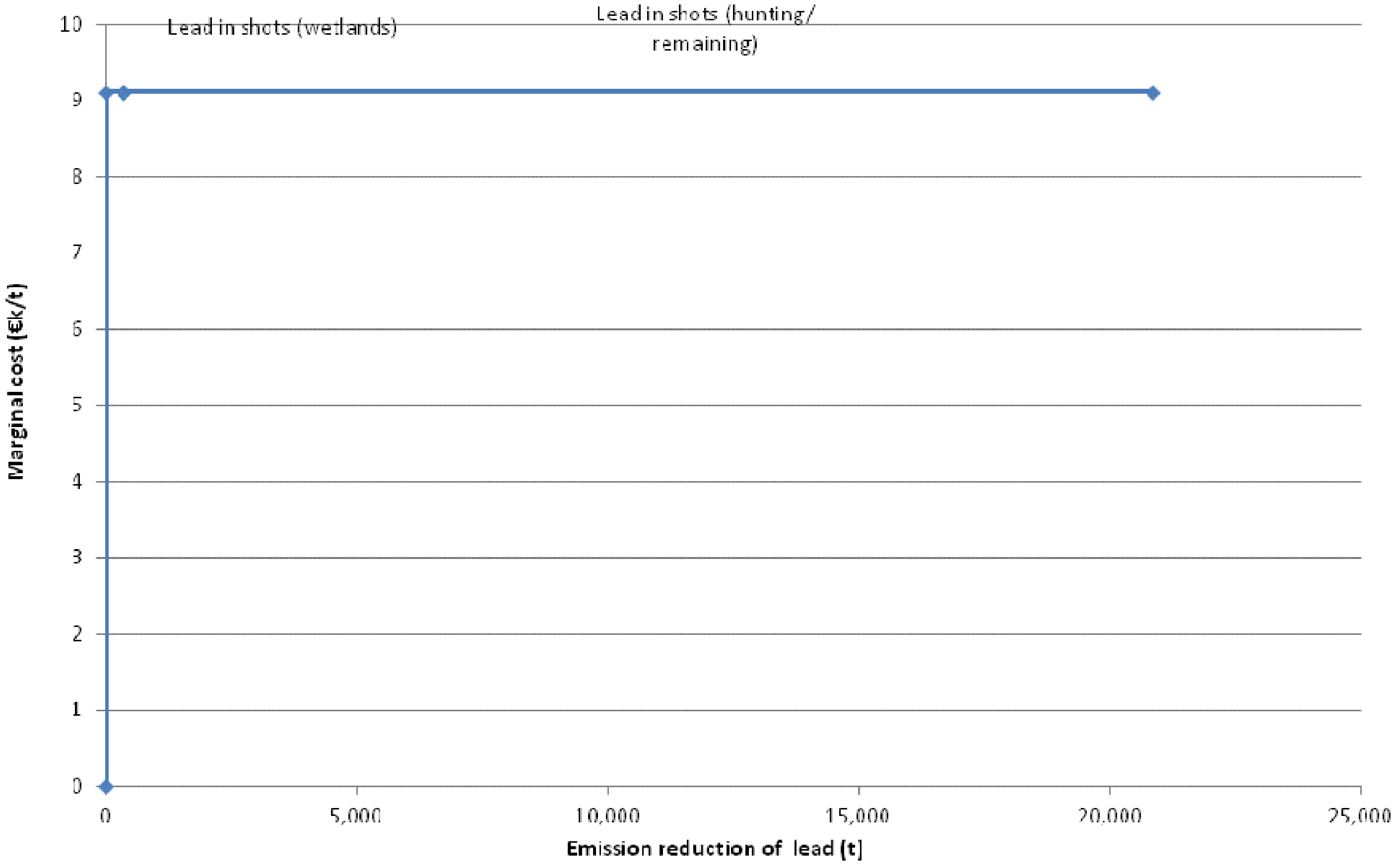


Figure 4.2 Marginal cost curve for lead shot for hunting



5. Key Assumptions and Uncertainties

The detailed assumptions used to derive the cost curves are provided in Appendix C. In this chapter, we briefly outline the key assumptions used and the main uncertainties of the approach adopted and the implications for the results.

In developing the curves we have assumed a number of one-off costs to the hunter. These include testing of shotguns to ensure suitability for use with steel shot. It has been assumed that, for the assumed 92% of hunters that use lead shotgun cartridges, 95% of shotguns would require testing. Furthermore, it is assumed that 15% of older unsuitable shotguns would then need to be replaced with new guns following such testing in the event of a restriction. It is assumed that these one-off costs would be incurred up-front following the implementation of any restriction.

Annual ongoing costs are based on 100% uptake of the least-cost option – steel shot. Based on a price analysis, steel shot is taken to be, on average, 110% of the lead shot price per cartridge (i.e. 10% higher price).

There are a number of uncertainties and limitations with the approach used. Firstly, data on one-off costs are based on limited quantitative information and are subject to uncertainty.

Secondly, it is simplistic to assume that all lead shot would be replaced by steel shot and it is likely that there would be at least some use of the other possible alternatives (such as steel, bismuth and tungsten), depending on users' preferences. This has the potential implication of underestimating the abatement costs as the other alternatives have significantly higher prices.

Thirdly, compliance with a possible restriction has been assumed to be 100%. According to FACE, recent studies and indications from the national hunting associations indicate that provision of information about non-lead shot and availability of alternatives is key to ensuring compliance. However, experience to date in certain Member States indicates that there are issues with non-compliance (e.g. Cromie et al., 2010) and these may potentially be significant. This has not been included in the cost curves but a set of sensitivity scenarios were developed, assuming 75%, 50% and 25% non-compliance in the Member States that currently have a total or partial ban on the use of lead shots on wetlands. The results of this sensitivity analysis are shown in the table below.

Table 5.1 Results of the sensitivity assessment (non-compliance)

	Base case	**Sensitivity scenario: 25% non-compliance**	**Sensitivity scenario: 50% non-compliance**	**Sensitivity scenario: 75% non-compliance**
Emissions of lead: total, tonnes	21,216	21,216	21,216	21,216
Emissions of lead from hunting on wetlands, tonnes	357	622	887	1,152
Emissions of lead from hunting on non-wetland areas, tonnes	20,859	20,594	20,329	20,064

	Base case	Sensitivity scenario: 25% non-compliance	Sensitivity scenario: 50% non-compliance	Sensitivity scenario: 75% non-compliance
Total annualised abatement costs (non-wetland areas), million Euro	190.3	187.9	185.4	183.0
Total annualised abatement costs (wetlands)	3.26	5.7	8.1	10.5

This shows that the potential future reduction in emissions associated with a restriction on wetlands could be greater if the potential existing levels of non-compliance are reduced. This would also entail additional costs , although these costs are essentially attributable to the existing (national) restrictions, rather than any wider restriction that might in the future be considered.

The assessment of abatement cost does not consider the impact of multiple gun ownership as a response to introduction of a ban. According to the information available the average number of guns per hunter in the Member States where total or partial bans are in place at present is almost exactly the same as the number of guns per hunter in Member States with no such bans. It is, therefore, considered that introduction of a wider restriction on use of lead shots in hunting on wetlands is unlikely to result in a relatively higher number of guns being required by hunters, although clearly there could be some need for additional replacement of existing guns.

6. Conclusions

The cost curves developed illustrate the relative costs, in €/t of use and emissions avoided, of substituting lead shot for hunting purposes. The cost curves are based on the current prices of alternatives that are already on the market. This assumes that the costs of any investments required to increase production of alternatives are reflected in the current prices of the alternatives that are already on the market. The residual value of capital equipment has not been considered in the cost curves. It is important to note, therefore, that costs incurred by individual companies currently involved in the production and supply of lead shot are not included in the cost curves. These costs could involve significant implications for the companies concerned (related to loss of market and associated employment if they are not able to supply alternatives and/or loss of the residual value of their investment in capital equipment that they currently use to produce lead shot.

Total annualised costs of reducing all uses of lead in shot for hunting are estimated at around €190 million per year, including both one-off and ongoing costs. Total costs at net present value, over twenty five years, are estimated at €2.7 billion. In comparison, the total annualised costs of implementing a ban on the use of lead shot in wetlands are estimated at around €3 million. Total costs at net present value, over twenty five years, are estimated at €45m. The cost effectiveness of implementing a ban on wetlands and in remaining areas has been assumed to be the same (€9.1k/t). However, in reality the costs of introducing a ban for hunting on wetlands may be higher, because some of the hunters will only shoot on wetlands part of the time, but may still be required to test and change their guns (and shot) so that they are able to use non-lead shot when on wetlands. We do not have sufficient quantitative data to account for this effect.

Furthermore, there are two key limitations with the analysis. Firstly, the measures assume 100% uptake of the least-cost alternative (steel). This is a simplistic analysis as, in practice, it is possible that a range of different alternatives would be used, depending on users' preferences. (However, it should be noted that steel seems to be the most widely adopted alternative so far.) This has the potential effect of underestimating the abatement costs.

Secondly, compliance with a restriction has been assumed to be 100%. Experience in the UK however indicates that there are issues with non-compliance (e.g. Cromie et al., 2010) and the same is thought to be true in other Member States with existing restrictions. According to FACE, recent studies and indications from the national hunting associations that provision of information about non-lead shot and availability of alternatives is key to ensuring compliance. This has not been included in the cost curves, however three sensitivity runs assuming existing levels of 25%, 50% and 75% non-compliance were developed and are discussed in the preceding section.

In the longer term, for instance if a restriction were to come into force in several years, the initial costs would be lower, because more older guns would have been replaced with guns that are also suitable for use with alternatives. It is possible that the higher costs of alternative shot would come down, as the market share of these alternatives increases. Furthermore, more Member States would be likely to have already implemented their own restrictions (i.e. under the baseline), hence potentially reducing the cost (and effectiveness) of any EU restriction.

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Appendix A
Member State legislation

Figure B.1 **Summary of EU-27 MS legislation relating to lead used in ordnance (based on 2009 survey with updates from consultation with FACE and member associations)**

Key	**Description**
🟩	Member State-wide ban including ban on commercial sales
🟩	Regional ban
🟩	Regional ban including ban on buffer zones
🟨	Regional ban but no ban on buffer zones
🟧	No ban currently in place but new legislation is anticipated
🟧	No ban currently in place and none anticipated

Member State	Ban either in certain areas (wetlands) or on certain species or on certain species in certain areas (state of play in November 2010) (various definitions of wetlands)	Ban on buffer zones	Total ban	Ban on commercial sale	Observations	DATE Legislation came into effect	New legislation adopted or under preparation - Date of implementation
DK	YES	YES	YES	YES		1993 (all hunting&trade 1996)	NO
NL	YES	YES	YES	NO		1995	NO
BE	Flanders - YES	Flanders - YES	Flanders - YES	NO		1998 (all hunting 2008)	NO
	Wallonia - YES	Wallonia - YES	Wallonia - NO	NO	Wallonia: "Nickel lead" is allowed	2006	yes (nickel-coated lead no longer legal)
	Brussels - YES	Brussels - YES	Brussels - YES	NO	Hunting prohibited everywhere	na	NO
	Federal - YES	Federal - YES	Federal - YES	NO			NO
BG	YES	YES	NO	NO	Buffer zone = 200 m around the wetland	2009	NO
FR	YES	YES	NO	NO	Buffer zone = 30 m around the wetland	2006	NO
HU	YES	YES	NO	NO	Possible buffer zone = 100 m around the wetland	15/08/2005	NO
IT	YES total ban in Natura 2000 wetlands; in several Regions total ban in every wetlands, also outside the Natura 2000 sites	YES	NO	NO	Buffer zone = 100 m around the wetland	2009	NO
CY	YES	YES	NO	NO	Buffer zone = 300 m around the wetland	2003	NO
AT	YES	?	NO	NO	legislation published for waterbird hunting	01/07/2012	
CZ	YES	?	NO	NO	Applies on waterfowl.	01/01/2011	NO
DE	YES in 11 länder	NO	NO	NO	Lead shot banned in wetlands in 11/16 länder Total ban in land-owned forests in Brandenburg (suspended for bullets due to safety concerns - ricochets)	progressive process since 1993, actual dates not known Note: Recommendation in 1993 co-signed by the Federal Minister for Agriculture and the President of DJV, for the use of non-lead shot for hunting of wildfowl in wetlands. Currently lead shot is banned in 11 of 16 Länder.	YES in 2 Länder
ES	YES	NO	NO	NO		2001 (extended 2007)	
FI	YES	NO	NO	NO	Applies on waterfowl - No ban in Åland	1996	NO
LV	YES	NO	NO	NO	Applies on waterfowl.	2000 (wetland SPA's)	NO
PT	YES	?	NO	NO	Applies to waterfowl on certain wetlands.	2010	NO
SE	YES	NO	NO	NO	Derogations possible for trap, double trap or skeet (sport shooting)	2002	NO
UK	England -YES	England -NO	England - NO	NO	All birds below high water mark in certain designated wetland SSSIs and all ducks, geese, coot and moorhen everywhere	1999	NO
	Wales - YES	Wales - NO	Wales - NO		All birds below high water mark in certain designated wetland SSSIs and all ducks, geese, coot and moorhen everywhere	2002	NO
	Scotland - YES	Scotland - NO	Scotland - NO		Any species over areas of wetland as defined by RAMSAR	2005	NO
	Northern Ireland - YES	Northern Ireland - NO	Northern Ireland - NO		Any species over areas of wetland as defined by RAMSAR	2009	NO
IE	NO	NO	NO	NO	On-going public consultation. An outline agreement has been reached with the main shooters representative body NARGC		?
EE	NO	NO	NO	NO	Certain species.		YES - 01/01/2013
LU	NO	NO	NO	NO	New regulation not adopted yet. Certain species.		YES - 2012
LT	NO	NO	NO	NO	On-going discussions with hunters and the ammunition sector. Hunting forbidden on the most important wetlands.		NO
MT	NO	NO	NO	-	No wetlands in Malta where hunting is permitted		?
SI	NO	NO	NO	-			Analysis under way
SK	NO	NO	NO	NO	Certain species in certain areas		YES - 01/01/2015
GR	NO	NO	NO	NO			Yes - 06/09/2013 to come into force
PL	NO	NO	NO	NO			NO
RO	NO	NO	NO	NO	progress likely in near future		NO

Appendix B
Prices of Shotgun Cartridges

Information received during consultation on the prices of shotgun cartridges is presented in the table below.

Table D.1 Prices shotgun cartridges, € cartridges

Country	Lead	Steel	Bismuth	Tungsten	Zinc	Notes[1]
Austria	No data	No data	€2.00	No data	Equal to lead	Information provided by the Zentralstelle Österreichischer Landesjagdverbände.
Belgium (Wallone Region)	No data	12 gauge normal pressure: €0.27 20 gauge normal pressure: €0.30 12 gauge high pressure: €0.48 20 gauge high pressure: €0.55	No data	No data	No data	Information provided by the ASBL Wallonne du Royal St. Hubert Club de Belgique.
Greece	€0.20 – €0.32	No data	No data	No data	No data	Information provided by the Hellenic Hunters Confederation
Italy	No data	Increase of cost about + 10%, + 15% in comparison to the respective products charged with lead shots	Increase of cost of about 20 times greater than products charged with lead shots	Increase of cost of about 20 times greater than products charged with lead shots	No data	Information provided by CNCN and FACE Italy.
Malta	€0.38	€0.38	No data	No data	No data	Information provided by FKNK.
Slovenia	Gauge 20/70 and 76, load 24-32 grams: €0.48 – €1.00 Gauge 16/70, load 20-30 grams: €0.52 – €1.00 Gauge 12/70 and 76, load 24-38 grams: €0.56 – €1.00	No data	No data	No data	No data	Information provided by Slovenian Hunters Association.
Spain	€0.36	€0.50	€2.00	No data	€1.20	Information provided by RFEC
Sweden	Gauge 12, 20: €0.28	Gauge 12, 20: €0.32				Information provided by the Swedish Association for Hunting and Wildlife Management.

1) Notes: No further information provided on source of price information.

Appendix C
Data for Cost Curves for lead in shot

Data for incorporation into cost curves

The table below outlines the assumptions and data used in developing specific measures for the cost curve for lead in shot.

Table E.1 Summary of measures for inclusion in cost curves

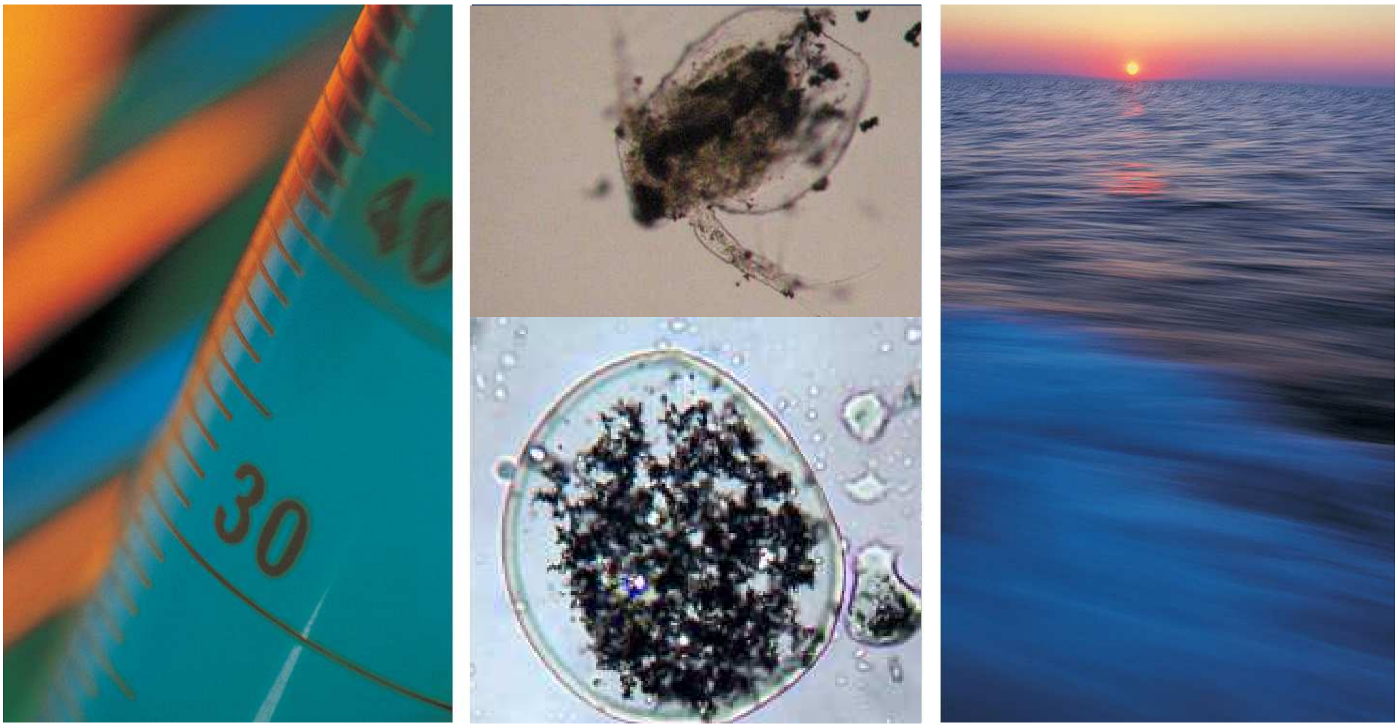
Measure	Details of key elements of measure
Total ban on lead in shot for hunting - use of alternatives	*One-off costs:* Includes: Of the 92% of hunters that use lead-shot cartridges, it is assumed that 95% of shotguns will need to be tested to ensure that they are suitable for use with steel shot. The price of testing is taken as €59/test in 2011 prices (based on COWI, 2004). Furthermore, it is assumed that of the 92% of hunters that use lead-shot cartridges, 15% of shotguns will need to be replaced as they are unsuitable for use with steel shot (based on COWI, 2004). The average cost of a new gun is estimated at €1,180 in 2011 prices (based on COWI, 2004). An amortisation period of 25 years has been used (as agreed with ECHA). This is based on the estimated turnover rate of shotgun ownership. Equivalent annual one off-costs have been estimated at €171m. These will be incurred up-front following the implementation of any restriction. *Annual ongoing costs:* Based on 100% uptake of least-cost option. An analysis of price information obtained during consultation with EU hunting associations found that steel shot is the least cost option (see Section 3.3) at 110% of lead shot price (average price of a lead shogun cartridge is estimated as €0.34). Annual ongoing costs have been estimated at €22.5 million. *Emissions reduced:* 100 per cent substitution of lead shot assumed. Total emission reduction = 21kt. *Main uncertainties and limitations* • Data on one-off costs are based on limited information and are subject to uncertainty. • Assumed that all lead shot will be replaced by steel – this is an over-simplification of what may happen in practise. *Specific measure taken forward:* Replacement of lead shot for hunting with steel shot. Main uncertainties and limitations: • 100% uptake of least cost option may not occur in practice; instead is likely that a range of different alternatives would be used, depending on users' preferences. COWI (2004) estimate that a likely mix would be: 50% steel, 20% tungsten alloy, 20% bismuth and 10% tin. • Non-compliance has not been accounted for. It was only considered in a separate sensitivity runs for the assumed 25%, 50% and 75% non-compliance with partial or full bans

Measure	Details of key elements of measure
Total ban on lead in shot in wetland areas for hunting - use of alternatives	*One-off costs:* Includes: Using available European bag data, it is estimated that approximately 6.7% of hunting takes place on wetlands. To estimate lead emissions from the use of lead shots on wetlands the following assumptions were made: none of the hunters shoot with lead on wetlands in the MS with a full ban on wetlands; 50% of shooting on wetlands uses lead in the MS with a partial ban and the standard EU proportion for shooting on wetlands is assumed in the remaining MS (6.7%). This resulted in an estimate of 1.7% of lead emissions originating on wetlands. Testing of 95% of shotguns to ensure they are suitable for use with steel shot. The price of testing is taken as €59/test in 2011 prices (based on COWI, 2004). Replacement of 15% of shotguns with new guns that are unsuitable for use with steel shot (based on COWI, 2004). The cost of a new gun is estimated at €1,180 in 2011 prices based on COWI, 2004). An amortisation period of 25 years has been used (as agreed with). This is based on the estimated turnover rate of shotgun ownership. Equivalent annual one off-costs have been estimated at €2.9m. These will be incurred up-front following the implementation of any restriction. *Annual ongoing costs:* Based on 100% uptake of least-cost option. Price analysis (see Section 3.3) found that steel shot is the least cost option at 110% of lead shot price (average price of a lead shogun cartridge is estimated as €0.34). Total annual ongoing costs have been estimated at €0.4 million. *Emissions reduced:* 100 per cent substitution of lead shot assumed. Total emission reduction = 0.4kt. *Specific measure taken forward:* Replacement of lead shot for hunting with steel shot. Main uncertainties and limitations: • 100% uptake of least cost option may not occur in practice; instead is likely that a range of different alternatives would be used, depending on users' preferences. COWI (2004) estimate that a likely mix would be: 50% steel, 20% tungsten alloy, 20% bismuth and 10% tin. • The cost effectiveness of implementing a ban on wetlands and in remaining areas has been assumed to be the same (€9.1k/t). However, in reality the costs of introducing a ban for hunting only on wetlands may be higher, because some of the hunters will only shoot on wetlands part of the time, but may still be required to test and change their guns (and shot) so that they are able to use non-lead shot when on wetlands. • Non-compliance has not been accounted for. It was only considered in a separate sensitivity runs for the assumed 25%, 50% and 75% non-compliance with partial or full bans

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Hexabromocyclododecane (HBCDD) – Final Report



AMEC Environment & Infrastructure UK Limited

December 2012

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Document Revisions

No.	Details	Date
1	Draft final report and preliminary results	06 December 2012
2	Final report taking into account ECHA comments	20 December 2012

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

Lot 5 (Hexabromocyclododecane) – Final Report

AMEC Environment & Infrastructure
UK Limited

December 2012

Report for

European Chemicals Agency (ECHA)
Contract No: ECHA 2011/140
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Contents

Executive Summary

This report provides the results of work on collection of data on abatement costs of reducing the use of hexabromocyclododecane (HBCDD). The work involved developing cost curves mainly based on information readily available in the literature. It has been undertaken on behalf of ECHA by AMEC and is part of a wider project examining the abatement costs for various chemicals.

HBCDD was identified as a Substance of Very High Concern (SVHC) pursuant to Article 57(a) as it is classified as a PBT substance. It was therefore included in the candidate list for authorisation following ECHA's decision ED/67/2008 on 28 October 2008. HBCDD is now included on Annex XIV of the REACH Regulation with a sunset date of July 2015 and a latest application date of February 2014. Therefore the scope of the current study covers uses of HBCDD that are not intermediate uses and that are hence potentially subject to a requirement for authorisation.

HBCDD is used as a flame retardant in four principal product types: Expanded Polystyrene (EPS), Extruded Polystyrene (XPS), High Impact Polystyrene (HIPS) and Polymer dispersion for textiles. The consumption of HBCDD in each of these applications in 2006 was estimated to be 5,300 t in EPS, 5,900 t in XPS and 200t in both textiles and HIPS. It is clear that HBCDD is not widely used in HIPS and textiles across the EU and use in these applications is understood to have declined still further since 2006. Therefore, the study focuses on its use as flame retardant in polystyrene foam types EPS and XPS. These are mainly used in the construction sector as insulation materials for buildings in order to comply with fire safety regulations.

In total, it is estimated that around 0.7 million tonnes of EPS-containing HBCDD was consumed in the EU in the construction sector, of which around 44% was for floor insulation, 41% for external wall insulation and 15% for roof insulation. In terms of XPS, best estimates are that around 0.5 million tonnes of XPS-containing HBCDD are consumed in the EU each year, of which around 40% was for floor insulation, 40% for external wall insulation and 20% for roof insulation.

Based on the information reviewed, a number of potential abatement measures to address the use of HBCDD in XPS or EPS are possible. These could include flame retardant chemical substitution, product redesign or material substitution. Whilst the first measure (chemical substation in EPS/XPS) is probably the most likely in the medium-to-long term, it is understood to not be feasible in the shorter-term, as new flame retardant technologies are being developed. The second option (product redesign to avoid flame retardant use) is not an option in certain Member States with specific fire safety requirements. The third measure regarding the use of alternative insulation materials to replace the use of HBCDD in EPS/XPS foams is the only measure considered in the cost curve data within this report, because information on prices are available. However, in practice, it is likely that a significant proportion of HBCDD will be replaced with chemical alternatives, once these are available, but no data on prices are currently available.

The cost curves developed illustrate the relative costs, in €/t of use avoided, of removing HBCDD use in EPS/XPS. The measures assume 100% uptake of the least-cost alternative for which cost information is available in each of the applications considered (floor, external wall and roofing insulation). In practice, it is likely that a range of different alternatives would be used, depending on users' preferences.

The costs per tonne of HBCDD removed are all based on the assumption that the alternative adopted is that which has the least cost to achieve an equivalent level of insulation as EPS/XPS containing HBCDD. In the case of EPS, the alternatives generally have higher cost to achieve the same level of thermal insulation (ranges from 25 to 175 €k/t of HBCDD removed), whereas for XPS, the alternatives appear to have lower cost (giving a negative value for the cost per tonne of HBCDD use avoided (ranges from -67 to -187 €k/t of HBCDD removed). It is important to note, however, that the costs only include the difference in prices between HBCDD-based products and the alternatives. They do not include the (potentially significant) cost implications that could occur as a result of the lost market, and lost residual value of capital equipment, for the current suppliers of HBCDD-based EPS and XPS to the insulation market. If such costs are included, it is therefore possible that there could be significantly higher costs for substitution of EPS and positive rather than negative costs for substitution of XPS. It has not been possible to quantify these effects here. Furthermore, other uncertainties include price variations in EPS/XPS and the alternatives (e.g. related to raw materials, energy consumption or individual product quality) which will affect relative prices of materials, on a temporal and geographic basis.

1. Introduction

1.1 The Project

The European Chemicals Agency (ECHA) has commissioned a project to provide information on “abatement costs for certain hazardous chemicals” (contract number ECHA/2011/140). The work is being undertaken by AMEC Environment & Infrastructure UK Limited (“AMEC”).

The present report concerns the substance hexabromocyclododecane (HBCD or HBCDD) (CAS numbers 25637-99-4 or 3194-55-6, EC numbers 247-148-4 or 221-695-9). In the study HBCDD is used as a generic term for the substance.

The data collected is intended to be used for:

- supporting the Agency in assessing the most appropriate risk management options for the substances addressed; and
- furthering the understanding of the usefulness of data on use/emissions abatement costs in risk management decision-making.

1.2 Project Context

This project follows on from a 2010 project on “*Abatement cost curves for substances of concern*” conducted by AMEC Environment & Infrastructure UK (previously Entec UK) for the Environment Agency, ECHA, the Health and Safety Executive (HSE) and RIVM. The main aim of that project was to develop a suitable method for estimating abatement costs to reduce emissions of chemicals and to apply and test the method with three selected case study substances. That study provided a first illustration of the benefits of being able to compare unit abatement costs amongst different substances and different uses.

The objective of this project is to assist ECHA in establishing capability to assess the abatement costs of reducing the use or emissions of hazardous substances. Under this lot (“Lot 5”), abatement potential and costs for HBCDD have been assessed.

HBCDD was identified as a Substance of Very High Concern (SVHC) pursuant to Article 57(a) as it is classified as a PBT substance and was therefore included in the candidate list for authorisation following ECHA’s decision ED/67/2008 on 28 October 2008. HBCDD is now included on Annex XIV of the REACH Regulation with a sunset date of July 2015 and a latest application date of February 2014. Therefore the current study has concentrated on assessing abatement costs of direct alternatives (i.e. alternative substances or techniques) in all applications, excluding the use as an intermediate.

Additionally, HBCDD was nominated for inclusion in the Stockholm Convention on Persistent Organic Pollutants by Norway in 2008. The Persistent Organic Pollutants Review Committee, at its eighth meeting in October 2012, decided to recommend to the sixth Conference of the Parties (to be held from 28 April to 10 May 2013), for its consideration, the listing of HBCDD in Annex A to the Convention "with specific exemptions for production and use in expanded polystyrene and extruded polystyrene in buildings". Details of the possible specific exemptions are not provided but it is stated that the exemption could be given together with a description of the conditions for production and for these uses (Decision POPRC-8/3)[1]. It is stated that this would effectively end the use of HBCDD in textile and HIPS applications.

The main outputs of the work, for this substance and for the other substances being assessed under different lots are expected to be as follows:

- Data on abatement costs of reducing the use or emissions of the chemical. Different applications of the same substance may/will incur different abatement costs.
- An overview on the functioning of the markets for the substances in question, including information on prices, amounts of the substance on the markets (including import and export), the relative shares of the substance used for different applications, number of actors involved in the business, as well as possible trends in the relevant market.

1.3 Structure of this Report

This report is structured as follows:

- Section 2 of this report provides an overview of the process of data gathering for this study and presents an overview of the data identified;
- Section 3 sets out an overview of the market for HBCDD;
- Section 4 presents the data analysis and resulting abatement cost curves; and
- Section 5 presents conclusions.

The appendices to this report include various other background data.

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[1] Decision POPRC-8/3 is included in the document containing all information on HBCDD in preparation for COP6, available: http://chm.pops.int/Convention/POPsReviewCommittee/LatestMeeting/POPRC8/POPRC8Followup/HBCDDRecommendation/tabid/2912/D efault.aspx

2. Data Collection

2.1 Overview

The aim of the study was to gather abatement costs data on the use of alternative substances or techniques to HBCDD in all applications (with the exception of use as an intermediate). This chapter briefly summarises the process of data gathering for this study and presents an overview of the data received.

2.2 Review of Existing Data Sources

The starting point for this study are an Entec (2011) report for DG Environment which assessed the potential costs and benefits associated with the introduction of HBCDD as a Priority Substance under the Water Framework Directive (Directive 2000/60/EC) and an IOM (2008) report (supported by Entec) for ECHA, which sought to provide information (including data on uses, releases and potential alternatives) on substances of very high concern including HBCDD.

Knowledge for this study also relies on the European risk assessment report (RAR) for the substance (EC, 2008) and on the risk management evaluation for HBCDD (RME)[2] adopted by the Persistent Organic Pollutants Review Committee under the Stockholm Convention, at its seventh meeting in October 2011. An addendum[3] to the RME for HBCDD was adopted by the POPs Review Committee at its eighth meeting in October 2012 to include supplementary information on alternatives to HBCDD and its use in expanded polystyrene and extruded polystyrene.

In order to inform the evaluations prepared by the POPs Review Committee, the Norwegian Climate and Pollution Agency (Klif) engaged COWI, in 2011, to undertake a study on the alternatives to the use of flame-retarded EPS in buildings. That study focused on alternative insulation materials to flame retarded polystyrene.

This information presented here has been collected by using the sources mentioned above, supplemented with further relevant sources identified during the course of the study.

2.3 Stakeholder Consultation

Since extensive information on HBCDD and their uses is available in publicly accessible sources, it was agreed with ECHA to limit the scope of the stakeholder consultation to the Industry Working Group for HBCDD, a sector group of the European Chemical Industry Council (Cefic), which is the link to likely authorisation applicants under the REACH regulation. Cefic has been approached through e-mails and phone calls although limited information has been received at the time of writing this report.

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[2] UNEP, (2011) Decision POPRC-7/1, adopting a risk management evaluation on HBCDD (UNEP/POPS/POPRC.7/19/Add.1)

[3] The additional information has been adopted as an addendum to the RME and is available here: http://chm.pops.int/Convention/POPsReviewCommittee/LatestMeeting/POPRC8/MeetingDocuments/tabid/2801/Default.aspx

Additionally, a recent consultation undertaken under the Stockholm Convention has provided valuable input to this study. The POPs Review Committee at its seventh meeting in October 2011 invited the intersessional working group on HBCDD[4] to gather further information on:

a) Chemical alternatives to HBCDD, especially in expanded polystyrene or extruded polystyrene foam applications, in terms of their availability, cost, efficacy, efficiency and health and environmental impact, especially with regard to their persistent organic pollutant properties;

b) Production and use of HBCDD,

Twenty-six Parties and country Observers Responses submitted information in response to the POPs consultation (Argentina, Azerbaijan, Brazil, Bulgaria, Cambodia, Cameroon, Canada, China, Germany, Guatemala, Indonesia, Ireland, Israel, Italy, Kiribati, Latvia, Mali, Mexico, Monaco, Myanmar, the Netherlands, Norway, Poland, Romania, Thailand and United States of America). In addition, six non-governmental Observers submitted information (Great Lakes Solutions, Green Chemicals Srl, International POPs Elimination Network IPEN, PS Foam Industry, Extruded Polystyrene Foam Association, and jointly the industry associations EXIBA (a Cefic sector group) and EPS (PlasticsEurope), as well as former POPRC member Ian Rae. The information received was processed and included in an addendum to the risk management evaluation on HBCDD[5].

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[4] Decision POPRC-7/1 on hexabromocyclododecane (UNEP, 2011)

[5] The additional information has been adopted as an addendum to the RME and is available here: http://chm.pops.int/Convention/POPsReviewCommittee/LatestMeeting/POPRC8/MeetingDocuments/tabid/2801/Default.aspx efault.aspx

3. General Market Overview

3.1 Introduction

HBCDD is used as a flame retardant additive in a variety of industrial applications and end products with the purpose of delaying ignition and slowing down the spread of subsequent fire.

The RME (2011) identifies this substance under two different names: hexabromocyclododecane (CAS number 25637-99-4) and 1, 2, 5, 6, 9, 10-hexabromocyclododecane (CAS number 3194-55-6). HBCDD was nominated for inclusion under the POPs Convention under both names and there are no known differences in molecular structure or properties for these two CAS numbers (EC, 2008). The chemical substance has a cyclic ring structure with Br-atoms attached. The molecular formula of the compound is $C_{12}H_{18}Br_6$ and its molecular weight is 641.7 g/mol. HBCDD exists in three isomers (alpha, beta and gamma) with equal composition but slightly different structure, depending on where the bromine atoms are bonded on the molecule's main plane (DEPA, 2010). The CAS and EC numbers for HBCDD and its isomers are shown below in Table 3.1.

Table 3.1 CAS/EC Numbers for HBCDD

Substance	CAS Number	EC Number
Hexabromocyclododecane	25637-99-4	247-148-4
1,2,5,6,9,10-hexabromocyclododecane	3194-55-6	221-695-9
alpha-HBCDD	134237-50-6	
beta-HBCDD	134237-51-7	
gamma-HBCDD	134237-52-8	

The production process of HBCDD is described in the RAR (EC, 2008) as a batch-process, where elementary bromine is added to cyclododecatriene (CDT) in the presence of a solvent. The process temperature is 20 to 70°C, and the reaction takes place in closed systems. These materials are mixed to form a slurry that must be extensively washed in order to generate high purity HBCDD. The slurry is centrifuged, the liquids removed for reprocessing and the solid faction is dried, stored in a silo and packed. According to one manufacturer (reported in the RAR), production and transportation of the material to silo and then packaging are done in a closed system. Commercial HBCDD is a white odourless solid substance and the product is delivered as powder or pellets.

Some quantities of HBCDD are micronised in a grinding process to smaller particles to be used in some applications (e.g. for use in the textile industry).

3.2 Manufacture, import and export

3.2.1 Historical information

HBCDD has been commercialised since the late 1960s. Production has been reported in China, Europe, Japan, and the USA (RME, 2011). In Europe, HBCDD is only produced at present at one site in the Netherlands. The manufacturing volume of HBCDD in this site was around 6,000 t per year in 2005 (data in IOM, 2008).

Based on the data presented by IOM (2008), the total consumption of HBCDD in the EU was estimated to be about 11,580 tonnes in 2006. European demand increased between 2003 and 2007 by approximately 18%. The report indicated that it was unclear whether this trend would be likely to continue given the growing concern about the negative effects of HBCDD in the environment and the moves in Nordic countries to phase out its use. As the demand of HBCDD within the EU was greater than the production (6,000 t), net imports to the EU were estimated at around 6,000 tonnes in 2006. No reliable data on export of HBCDD as a simple substance or in mixtures out of the EU were made available.

3.2.2 Current best estimates

According to the RME (UNEP, 2011), the estimated global annual production of HBCDD is approximately 31,000 tonnes. This is the result of adding the production of the member companies of the Bromine Science and Environment Forum (BSEF) in Europe and the United States in 2009 (13,426 tonnes) and the reported Chinese production in 2011 (18,000 tonnes). In particular, it has been reported that Chinese production in 2011 increased by 20% compared to 2010, and exports out of China were around 5,500-6,000 tonnes of its annual production in 2011. Poland has indicated that 500 tonnes of HBCDD are imported from China annually.

The BSEF production in Europe has been reported together with that of the United States and totalled 13,426 tonnes in 2009. In the absence of more recent data, it is assumed that the annual production of HBCDD in the Netherlands site has remained stable at 6,000 tonnes, which would mean that Europe's production represented 45% of the reported consumption by BSEF member companies for Europe and the United States

Regarding market demand, the consumption and use of HBCDD mainly takes place in Europe and China. Available information suggests that use of HBCDD may be rising. In Europe, the total sales volume of HBCDD has increased among members of the European Flame Retardants Association (EFRA) according to a progress report (2011) of the Voluntary Emissions Control Action Programme (VECAP), described later in section 4. Sales data were collected through a survey carried out in 2011, which covered 97% of the volumes sold in the EU in 2011 by EFRA member companies[6]. The results of that survey are expressed in tonnage bands and presented in Table 3.2.

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[6] VECAP progress report (2011). Available at http://www.vecap.info/flipbook/vecap-sustainable 2/HTML/files/assets/downloads/publication.pdf

Table 3.2 Total Volume Sold of HBCDD in the EU by EFRA Member Companies (VECAP, 2011)

Survey year	**2008**	**2009**	**2010**	**2011**
Total Volume Sold of HBCDD (metric tonnes/year)	10,000-12,500	7,500-10,000	7,500-10,000	10,000-12,500

The data presented in the table reveal that the sales of HBCDD recovered in 2011 to similar volumes as those reported in 2008. As indicated previously, based on the production data reported in the RME (UNEP, 2011) it has been assumed that the average production of HBCDD in Europe was about 6,000 tonnes in 2011. Therefore imports could be around 6,000 tonnes per year. No information on exports has been identified.

It is relevant to note that, in 2012, the market for HBCDD has been restricted due to problems in production of its precursor cyclododecatriene (CDT). In this sense, in April 2012, there was a declaration of force majeure on FR 1206 HBCDD supplies by ICL Industrial Products, which owns the only European manufacturing site[7]. The company said it was likely the HBCDD plant at Terneuzen, in the Netherlands, which had been running on existing CDT inventories, would be shut that week as stocks of this substance run out. The reason for this was that an accident occurred at the CDT unit of the main European supplier of CDT, a key component in the manufacture of HBCDD.

3.3 Current uses

3.3.1 Overview

HBCDD is used as a flame retardant additive in a variety of industrial applications and end products with the purpose of delaying ignition and slowing down the spread of subsequent fire. In all products HBCDD is uniformly incorporated as an integral encapsulated component within the polymer matrix; however it is not bound to the matrix or transformed (IOM, 2008).

HBCDD is used in four principal product types:

Expanded Polystyrene (EPS)

Extruded Polystyrene (XPS)

High Impact Polystyrene (HIPS)

Polymer dispersion for textiles

The RME (UNEP, 2011) indicates that HBCDD is primarily used in the polystyrene foam types EPS and XPS for insulation and construction. The use in high impact polystyrene (HIPS) electric and electronic appliances and in

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[7] http://www.platts.com/RSSFeedDetailedNews/RSSFeed/Petrochemicals/8232581

flame retarding back coatings for certain textiles (including in automotive applications) are of a much smaller scale. In this sense, according to the U.S.EPA Chemical Data Reporting (CDR) database of 2006, less than 1% of the total commercial and consumer use of HBCDD was used for fabrics, textiles and apparel. Additionally China reports in the RME that, of the 12,000 tonnes of HBCDD used in China, 9000 tonnes is used for EPS and 3000 tonnes for XPS. The volumes of HBCDD flame retarded articles imported and exported globally are generally unknown, although polystyrene foam materials are usually tailor-made for the local construction market due to transport and cost considerations. Therefore it is likely that the main share of the production is locally consumed, and not exported.

At an EU level, there is no current information on the proportions of HBCDD used for different purposes. The industry association Cefic has indicated for the present study that HBCDD appears to be no longer used in HIPS and textiles (or may be only used in insignificant quantities) as technically suitable alternative substances and materials are available and globally are already used extensively.

Accordingly, it seems that, in Europe, HBCDD remain widely used in EPS and XPS foams. In 2006, there was a slightly greater proportion used in XPS (5,900 tonnes) than in EPS (5,300 tonnes) (ratio about 52:48). Recent data have been provided by Poland, indicating the use of 364 tonnes of HBCDD in EPS and 90 tonnes in XPS in 2011, although this is clearly not necessarily representative of the overall EU use.

3.3.2 Expanded Polystyrene (EPS)

Expanded Polystyrene (EPS) is a rigid cellular plastic used in a wide range of applications in Europe. EPS is primarily used as an insulation material for walls, roofs and floors in all kinds of buildings. Additionally, it is also used in other construction or civil engineering applications (e.g. frost insulation in road and railway construction). Outside the construction sector, EPS is also used in the packaging industry and in a variety of other professional and consumer uses, but on a smaller scale. Recent market data (BASF Plasticportal, 2011) indicate that, in Europe, 80% of EPS consumption goes into the construction sector, while the remaining 20% is used for packaging purposes.

In its pure forms, EPS is easily flammable or ignitable. Therefore, many European countries require EPS foams to meet regulatory fire safety levels. The EPS Industry[8] has indicated that approximately 80% of all EPS boards are processed to meet these national fire safety regulations, with HBCDD being the flame retardant of choice in these materials. The industry indicates that HBCDD is the only effective and durable flame retardant enabling EPS insulation foams to meet all existing national fire safety requirements. While in the construction sector nearly all EPS is flame retarded, only smaller quantities of EPS containing HBCDD are used in the packaging industry. EPS used in packaging does generally not contain any flame retardant additive (EC, 2008).

Flame retarded-EPS can be produced in a variety of densities providing a wide range of properties, shapes and sizes for specific applications (Klif 2011). The manufacturing process is described in IOM (2008). EPS containing HBCDD is manufactured in a batch process by suspension polymerisation of styrene in water (a one-step process). HBCDD powder is suspended at low temperatures in styrene prior to the addition of the water phase. HBCDD is

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[8] http://www.bsef.com/uploads/MediaRoom/documents/eps_xps_factsheet_november_final.pdf

incorporated as an integral and encapsulated component within the polymer matrix with uniform concentration throughout the bead. After complete conversion of the styrene monomer to EPS-beads, the reactor is cooled down and the beads are separated from the water by centrifugation.

The EPS beads are dried and classified into various size fractions and surface coated. These different grades are packed in bins, bags, or transported in bulk trucks to the EPS-converters. EPS foam is produced from EPS beads through pre-expansion of the beads with dry saturated steam, drying with warm air and shaping in shape moulds or in a continuous moulding machine. The foam can then be further formed by cutting, sawing or other machine operations (IOM, 2008).

The concentration of HBCDD in EPS boards is typically around 0.5% HBCDD by weight in the final product, while in EPS beads it is assumed to be at a maximum of 0.7 % (Klif, 2011). In Europe, the annual use of HBCDD in EPS increased from 3,500 tonnes in 2002 to 5,300 tonnes in 2006 (IOM, 2008). Taking into account these figures (0.5% HBCDD content and 5,300 tonnes consumption in 2006); estimated EPS consumption in 2006 is estimated to be about 1.1 million tonnes.

Recent information available in BASF Plasticsportal (2011) reveals that the EPS market in Europe (including Russia) is estimated to be 1.80 million tonnes with construction applications including building insulation accounting for 80% of demand. The remaining portion is mainly used in the packaging industry (20%). European consumption has increased by nearly 60% if compared to the levels of 2001 and in 2011 represented nearly 31% of the global EPS demand in 2011, which was 5.83 million tonnes.

EPS market data have also been provided by PlasticsEurope in 2010. The industry association indicates that, in Western Europe, demand for EPS was approximately 0.84 million tonnes in 2001, representing a value of approximately 3 billion Euro. The average annual growth was expected to be 2.5% per annum up to 2010. The outcome of applying this increase factor to the consumption reported for 2001 results in a consumption of 1.1 million tonnes of EPS in 2012. These data seem reasonable if compared with those provided by BASF, as it would mean that around 700 kt are consumed in Russia (39% of the total).

The EPS production sites are mainly located in Germany (27% of sites), Netherlands (13%) and France (12%). In general it is a fragmented sector with many plants (formulators and industrial users) geographically spread out across the EU (Entec, 2011). This data are presented in Table 3.3

Table 3.3 Distribution of production of EPS containing HBCDD in the EU (Entec, 2011)

Use	Source	No. of users	Geographical distribution (% of sites)
EPS formulators (provide raw material; EPS beads)	HBCDD industry group, 2011	22	Main MS: Germany (27%), Netherlands (13%) and France (12%)
EPS industrial users (manufacture this raw material, e.g. into EPS insulation foam boards)		Around 600	Across Europe

3.3.3 Extruded Polystyrene (XPS)

Extruded polystyrene (XPS) is a plastic foam based on polystyrene that is formed by adding gas during extrusion. XPS is mainly used in the construction sector for many of the same applications as EPS such as insulation of roofs, external walls, cavities and floors. XPS is a strong and highly moisture-resistant material. Therefore it is particularly suitable for applications that require high mechanical and water resistance, such as insulation of basement walls and foundations (Klif, 2011). The distribution of XPS insulation boards by application in buildings for 2006 shows that 38% was used for external walls; 38% percent was consumed for floor construction; 18% and 1% for flat roofs and pitched roofs respectively; and the remaining 5% in other applications (Klif, 2011, based on data provided for the German market). A smaller proportion of XPS is also used outside the building sector in civil engineering applications, cold stores and vehicles (ECHA, 2009).

As with EPS, fire safety regulations in many European countries require XPS insulation materials to be flame retarded. The XPS Industry[9] has indicated that practically all XPS boards are processed to meet national fire safety regulations, with HBCDD being the flame retardant of choice in these materials.

The manufacturing process of XPS containing HBCDD is described in IOM (2008) and Klif (2011). XPS foam begins with solid polystyrene crystals, are fed continuously to an extruder, along with special additives and a blowing agent. Within the extruder the mixture is combined and melted under controlled conditions of high temperature and pressure into a viscous plastic fluid or gel. The hot, thick liquid is then forced in a continuous process through an orifice called a die. As it emerges from the die, the blowing agent volatilises and it expands to foam, is shaped, cooled, and trimmed to dimension. XPS is extruded into smaller blocks, which may require gluing multiple billets together to achieve the necessary size.

The HBCDD is supplied either in powder or in low-dust granulated form in either 25 kg bags or in 1 tonne supersacks or “big bags”. The supersacks are emptied into hoppers designed to minimise dust emissions. The HBCDD is then carried to the point of mixing with screw or air driven metering equipment. The compounded polystyrene is extruded and cut into granules, and packaged. The extrudate is either air-cooled or cooled by running in a water bath.

As cited in Klif (2011), the concentration of HBCDD in XPS foams produced in Europe is up to 3% loading to meet technical and flammability foam requirements. In Canada, HBCDD levels in XPS produced are typically from 0.5 to 1%. In Europe, the annual use of HBCDD in XPS increased from 4,000 tonnes in 2002 to 5,900 tonnes in 2006 (IOM 2008). Taking into account these figures (1.5% HBCDD content and 5,900 tonnes consumption in 2006), estimated XPS consumption in 2006 was about 0.4 million tonnes.

XPS has a lower market volume than EPS. Its global consumption in 2011 was 30 million m^3 (or 1.05 million tonnes if using an average density of 35 kg/m^3). European demand, including Russia and Turkey, was estimated to be 50% of the global total, implying 0.53 million tonnes of XPS consumed. These data seem reasonable if compared with the above estimate of 0.4 million tonnes XPS consumption in 2006 (based on HBCDD content).

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[9] http://www.bsef.com/uploads/MediaRoom/documents/eps_xps_factsheet_november_final.pdf

The XPS production sites are mainly located in Germany (21% of sites), Italy (16%) and Spain (11%). In general it is a fragmented sector with many plants (formulators and industrial users) geographically spread out across the EU. Data provided in 2011 indicate that there are around 56 XPS production facilities. This data are presented in Table 3.4.

Table 3.4 Distribution of production of XPS containing HBCDD in the EU (Entec, 2011)

Use	Source	No. Users	Geographical distribution (% of sites)
XPS formulators (provide raw material; PS compound)	HBCDD industry group, 2011	56 (XPS uses around 90% flame retarded material)	Main MS: Germany (21%), Italy (16%) and Spain (11%)
XPS industrial users (manufacture this raw material into XPS insulation foam boards)	EC,2008	35	Across Europe

3.3.4 High Impact Polystyrene (HIPS)

The use of HBCDD in High Impact Polystyrene (HIPS) products is mainly for electrical and electronic appliances such as video and stereo equipment, distribution boxes for electrical lines in the construction sector and refrigerator lining (IOM, 2008).

Different sources estimate the HBCDD content of flame-retarded HIPS between 1-7% (w/w) and the EU Risk Assessment Report (cited in IOM, 2008) assumed as a realistic worst case, that HIPS contains 7% HBCDD. The annual use volume in Europe was estimated to be about 200 tonnes in 2006 (IOM, 2008), representing only 2% of the total consumption of HBCDD. As stated above, the volume may currently be lower than this due to the availability of technically suitable alternatives.

The addendum to the RME (UNEP, 2011), indicates that halogenated flame retardants, such as decabromodiphenylethane, and other alternatives appear to have largely replaced the use of HBCDD in HIPS, due to higher efficacy and equal prices to HBCDD. Relevant information on available alternatives is summarised in Table 3.5:

Table 3.5 Alternatives to HBCDD in HIPS (UNEP, 2012)

Chemical	Availability	Costs	Efficacy
Ethylenebis (tetrabromophthalimide) (EBTPI) CAS No: 32588-76-4	Commercially available and used extensively It is mostly used in HIPS, polyethylene, polypropylene, thermoplastic polyesters, polyamide, EPDM, rubbers, polycarbonate, ethylene copolymers, ionomer resins, and textiles.	Not stated	Technically feasible and used extensively
Decabromodiphenylethane (DBDPE) CAS No: 84852-53-9	Commercially available and used extensively. DBDPE became commercially important as an alternative to DecaBDE formulations. Europe does not produce DBDPE, but imports in 2001 were estimated to be between 1000 and 5000 tons, primarily to Germany. DBDPE is the second highest current use additive BFR in China with production increasing at 80% per year	According to one Party, DBDPE is commonly used in HIPS and textiles, with better effect than HBCD and approximately equal price as HBCD. Basically replaced HBCDD in 2011 in this application in China.	Technically feasible and used extensively
Decabromodiphenyl ether (DecaBDE) CAS No: 1163-19-5	Commercially available and used extensively. Many manufacturers have phased use out since the early 2000's. In Japan, there has been a clear shift in consumption away from DecaBDE to DBDPE.	Not stated	Technically feasible and used extensively
Triphenyl phosphate CAS No: 115-86-6	Commercially available and used extensively.	Not stated	Technically feasible and used extensively
Bisphenol A bis (biphenyl phosphate) (BDP) CAS No: 5945-33-5	Commercially available and used extensively.	Not stated	Technically feasible and used extensively
Diphenyl cresyl phosphate CAS No: 26444-49-5	Commercially available and used extensively.	Not stated	Technically feasible and used extensively

3.3.5 Textile coating

According to IOM (2008), micronised particles of HBCDD are used in textile applications to comply with flame retardant standards, mainly for upholstered furniture and seating in transportation, draperies, bed mattress ticking, interior and automobile textiles. The polymer industry formulates HBCDD to polymer-based dispersions (e.g. acrylic or latex) of variable viscosity, which are then processed in the textile finishing industry.

The typical concentration of HBCDD in the final layer of end products can be up to 25%, or alternatively 6 to 15% combined with about 4 to 10% of antimony trioxide that has a synergistic flame-retardant effect. The annual use volume in Europe was estimated to be about 200 tonnes in 2006, representing only 2% of the total consumption of HBCDD. As for HIPS, the volume may currently be lower than this due to the availability of technically suitable alternatives.

The addendum to the RME (UNEP, 2011), indicates that halogenated flame retardants, such as decabromodiphenylethane, and other alternatives appear to have largely replaced the use of HBCDD in textiles. Relevant information on available alternatives is summarised in Table 3.6:

Table 3.6 Alternatives to HBCDD in Textiles (UNEP, 2012)

Chemical	Availability	Costs	Efficacy
DecabromodiphenylethaneCAS No: 84852-53-9	Commercially available and used extensively. The substance became commercially important as an alternative to DecaBDE formulations. Europe does not produce DBDPE, but imports in 2001 were estimated to be between 1000 and 5000 tonnes, primarily to Germany. Decabromodiphenylethane is the second highest current use additive BFR in China with production increasing at 80% per year	According to one Party, DBDPE is commonly used in HIPS and textiles, with better effect than HBCDD and approximately equal price as HBCDD. Basically replaced HBCDD in 2011 in this application in China.	Technically feasible and used extensively
Decabromodiphenyl ether (DecaBDE) CAS No: 1163-19-5	Commercially available and used extensively. Many manufacturers have phased use out since the early 2000s. In Japan, there has been a clear shift in consumption away from DecaBDE to Decabromodiphenylethane.	Not stated	Technically feasible and used extensively
Chlorinated paraffins (C10-13) –CAS No: 85535-84-8	Available and used extensively.	Used extensively	Technically feasible and used extensively
Ammonium polyphosphate – CAS RN 68333-79-9	Available and used extensively.	Used extensively	Technically feasible and used extensively

3.4 Scope of uses covered in the current study

The scope of the current study covers uses of HBCDD that are not intermediate uses and that are hence potentially subject to a requirement for authorisation. Given that HBCDD is not widely used in HIPS and textiles across the EU, the remainder of the study focuses on its use as flame retardant in polystyrene foam types EPS and XPS. As discussed previously, EPS/XPS foams containing HBCDD are mainly used in the construction sector as insulation materials for buildings in order to comply with fire safety regulations. Therefore the analysis of alternatives will be dedicated to these uses.

Table 3.7 provides an overview of the different uses of HBCDD, based on the data provided by IOM (2008) for 2006. The total volume of HBCDD used in Europe was estimated at approximately 11,600 tonnes per year and XPS and EPS represented around 96% of the total use.

Table 3.7 Summary table of the different uses of HBCDD in 2006 (IOM, 2008)

Use	Tonnes HBCDD per year
Expanded Polystyrene	5,300
Extruded Polystyrene	5,900
High Impact Polystyrene	200
Textile coating	200
TOTAL	**11,600**

The low use of HBCDD in textiles and HIPS applications can be explained by the fact that there are technically suitable and commercially available alternatives, as indicated in the RME (UNEP, 2011). The inclusion of HBCDD in Annex IV to the REACH Regulation is likely to have driven the uptake of these alternatives, phasing out HBCDD in HIPS and textiles across Europe. Therefore it is likely that the volume currently used is lower than this.

3.5 Alternatives to HBCDD use in EPS/XPS

3.5.1 Overview

There are a number of alternatives that could be used to replace the use of HBCDD in XPS or EPS foams. These include flame retardant substitution (chemical), material substitution and product redesign.

The alternative substances and techniques described below have largely been identified from the Addendum to the POPs RME (2012) and from the report prepared by COWI for the Norwegian Climate and Pollution Agency (2011), referred to previously in this report.

3.5.2 Chemical substitutes to HBCDD

As indicated in previous sections, according to industry, HBCDD has proven to be the only flame retardant enabling EPS/XPS insulation foams to meet all existing national fire safety requirements in Europe. Whereas other brominated flame retardants are to some extent used in North America (Klif, 2011), currently only HBCDD can be used in the manufacturing one-step process production process applied in Europe, where all additives, including HBCDD are mixed in the styrene solution prior to polymerisation. In the ‘two-step’ process, the flame retardant is added into the ready-made bead, but HBCDD penetrates the beads poorly after polymerization. Therefore, European manufactures use the one-step process in order to reach the desired fire safety standards (UNEP, 2011).

Over the past decade, significant efforts have been made to identify and characterise alternative flame retardants to HBCDD. According to the Bromine Science and Environment Forum (BSEF, 2012) potential alternatives to HBCDD in EPS and XPS are at variously advanced development stages, although it will take several years before technically and commercially feasible alternatives covering the needs of the market can be phased in by the polystyrene foam industry.

In March 2011, Dow Global Technologies LLC (DGTL), a subsidiary of The Dow Chemical Company, announced the development of a new brominated polymeric flame retardant (Polymeric FR) that could replace HBCDD. The substance is claimed to be stable, high molecular weight, non PBT and suitable for processing in EPS and XPS foam building insulation products (suitable for the one-step process). In addition, other chemicals are under development in several regions by collaborative efforts between different stakeholders.

For its 8th meeting, the POP Review Committee under the Stockholm Convention gathered information on potential chemical alternatives to HBCDD, focusing on the "polymeric FR" drop-in alternative for EPS and XPS production. Relevant information on this substance is summarised in Table 3.8:

Table 3.8 Characterisation of chemical alternative to HBCDD in EPS/XPS (UNEP, 2012)

Characterisation	**Polymeric FR**
Identification	Benzene, ethenyl-, polymer with1,3-butadiene,brominated (CAS No:1195978-93-8)
Availability	The alternative will become commercially available gradually starting in 2012. Anticipated to be sufficient capacity to replace HBCDD within 3-5 years. Trade names are Emerald 3000 and FR122P.
Transition time	After any alternative becomes available in commercial quantities, it will take some time for the industry to seek qualification and re-certification of polystyrene bead and foam products for fire-rating. According to industry information from Canada, a period of at least 5 years is needed to fully convert to an alternative. Downstream users have already been testing this alternative and the results reported have been positive.
Costs	Precise cost estimates will not be available until the Polymeric FR is fully commercialized. Some sources indicated higher costs of the Polymeric FR compared to HBCDD. However, no financial values were included to support this. According to one producer of the Polymeric FR, manufacturing flame retarded products with the alternative to HBCDD is not anticipated to have any significant impact on the cost competitiveness of EPS or XPS. It remains unclear whether the flame retardant represents a significant factor in the price of the final product (EPS/XPS insulation). There will be additional one-off costs to the industry from e.g. plant pilot trials and product qualification. However, these costs are irrespective of the alternative and have been considered in Canada to be in the low millions of Canadian dollars.
Efficacy	Polymeric FR is reported to have essentially equivalent flame retardant efficiency to HBCDD when used at equivalent bromine content. The required load is comparable to that of HBCDD (0.5-2.5% HBCDD w/w) in PS foams. However, XPS producers report efficacy is 83% of HBCDD.
Health and environmental impact:	According to the MSDS information and the industry hazard assessment, Polymeric FR is potentially persistent, but not bioaccumulative or toxic. However, there no independent reviews on its properties have been identified.

Additionally the Addendum to the RME (UNEP, 2012) has identified that two other brominated flame retardants (Benzene, 1,1'-(1-methylethylidene)bis[3,5-dibromo-4-(2,3-dibromo-2-methylpropoxy)] CAS No: 97416-84-7 and Tetrabromobisphenol A bis(2,3-dibromopropyl ether) (TBBPA-DBPE), CAS No: 21850-44-2 with dicumene for XPS and dicumylperoxide for EPS as usual synergists i.e. Starflame/GC SAM) appear to be suitable for replacing HBCDD in EPS processes. However, there is no information whether the first chemical is available and the latter has not yet been commercialised.

3.5.2.1 Product redesign

This option considers the possibility of using non-flame retarded EPS/XPS as insulation material, ensuring that fire safety is not compromised. In some applications, flame retarded polystyrene foams may be replaced by grades without flame-retardants, if thermal barriers and fire-resistive construction principles are used.

In some countries, such as Germany, building regulation stipulates the use of flame-retarded grades for all applications, whereas in others (Scandinavia), the performance of the entire building element is tested. In the latter countries, EPS without flame-retardants are used for e.g. floor and flat roof insulation using constructions where the insulation material is covered with non-combustible materials with high thermal heat capacity, e.g. concrete (Klif, 2011).

Although using non-flame retarded EPS will have a similar price to that containing HBCDD, the costs of introducing thermal barriers and other construction principles have to be considered. However, the fact that this solution is widely used in some countries (Sweden and Norway) indicates that the costs for some applications would not be higher than the cost of changing to alternative flame retarded materials (UNEP, 2011).

Fire-safety requirements, however, vary in the EU and these alternatives are not currently feasible for most countries, such as Germany, because of current national technical standards and building codes. In these cases there is a need to ensure a continued supply of flame-retardant insulation.

3.5.3 Alternative insulation materials

There are a number of alternative forms of insulation that can be used in place of XPS or EPS. Mineral wool and polyurethane/polyisocyanurate (PUR or PIR) foams, along with EPS and XPS, are all widely used in the European construction industry, along with a number of other materials. Table 3.9 presents an overview of the insulation market:

Table 3.9 Insulation market in Europe (Shell Chemicals Magazine, 2012)[10]

Material	EPS	XPS	PUR	Mineral wool	Others
Share	35%	4%	8%	50%	3%

Mineral wools (glass and stone) represent 50% of the insulation market while plastic foams (EPS, XPS and PUR) take up about 47%, with EPS being the material more widely used. The choice of insulation material depends on a number of different factors, including building design, requirements for energy efficiency and cost implications.

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[10] http://s08.static-shell.com/content/dam/shell/static/chemicals/downloads/aboutshell/magazine-spring-2012buildingbenefitseps.pdf

Klif (2011) has evaluated alternatives to the use of EPS containing HBCDD flame retardant as insulation material in buildings. Although the study focuses on EPS, it notes that most of the alternatives analysed are also considered relevant alternatives to XPS for the main applications.

Technical feasibility

The Klif (2011) study has identified alternative insulation materials for the most common applications that use flame retarded EPS and XPS: external façade insulation, flat roof insulation, floor insulation and sandwich panels. Table 3.10 gives an overview of the different applications of the EPS/XPS types, as cited in Klif (2011):

Table 3.10 Distribution of different applications of EPS/XPS in buildings in Germany (Klif, 2011)

Use	EPS	XPS
External wall insulation	39%	38%
Floor construction	41%	38%
Flat roof	13%	18%
Pitched roof	1%	1%
Others	6%	5%

The table below provides a comparison of the technical suitability of a selection of key insulation materials, based on Klif (2011). These materials are marketed for the main applications of EPS and share several of its main advantages.

Table 3.11 Key Properties for Various Insulation Materials

Technical Solution	Density kg/m^3	Compressive Strength	Water Resistance (+/-)	Form [Note 1]	Key Applications [Note 2]
EPS sheets	15-35	Medium	-/+	S, L	W, FR, S
Stone wool	24-105	Low-medium	+	M, L	W, FR, S
Glass wool	16-24	Low-medium	+	M, R, L	W, FR, S
PUR/PIR [Note 3]	30-40	Medium	+	S	W, FR, S

Notes:
1: Slabs/boards, Mats/batts, Rolls, Loose fill
2: Exterior Wall / Flat Roofs / Floors / Sandwich elements
3: Rigid polyurethane (PUR) and polyisocyanurate (PIR)

It is clear that different insulation materials can be used interchangeably in various applications. EPS foams are mainly used in slab/board and loose fill forms, which are also used in the other alternatives marketed for the same applications. The key characteristic that makes EPS foams attractive for a number of applications is its low weight, which may facilitate logistics in the building process (e.g. transport costs). This is combined with good compression strength and a relatively low effect of moisture on insulation value. Industry consultation suggests that key application areas where the use of EPS is regarded as particularly attractive (and has a significant market share) are external thermal insulation, floor thermal insulation and terrace roofs. Flame retarded XPS is particularly suitable for applications where high mechanical and water resistance is required. All materials considered are expected to remain in service for the lifetime of the building (unless removed during major reconstruction/renovation works).

Mineral wools (glass and stone) and PUR/PIR foams are as effective as EPS/XPS in many applications and share some of the key properties regarding moisture resistance and low density (excluding stone wool). In addition, PUR/PIR foams have lower thermal conductivity than EPS, XPS or mineral wool, which enables a smaller insulation thickness to be used to achieve the same thermal insulation efficiency.

Klif (2011) indicates that replacing flame retarded EPS boards with the above alternatives will not compromise fire safety. These materials typically have better fire performance and in general are able to meet the same fire requirements, or higher, as the flame retarded EPS. Furthermore, unlike EPS which contains HBCDD, none of the substances contained in these materials have been demonstrated to be POPs or PBTs and none have been classified as CMR substances.

Therefore, it is considered that it would be technically feasible to replace EPS/XPS foam insulation with alternatives. However, this would probably necessitate changes to the design of certain new buildings, or reduced energy efficiency of buildings in cases where thicker insulation materials (which would be required when using alternatives such as mineral wool, but not polyurethane) could not be accommodated into existing buildings.

Price analysis

According to the Klif (2011) study, the prices of the EPS boards range from 11.6 EUR/m^2 to 17.9 EUR/m^2 at 100 mm thickness, although these differ much depending on the thermal conductivity and the compressive strength of the boards.

Based on price data of the German insulation market for 2006, Klif (2011) estimates that the price of the least expensive alternatives to flame retarded EPS ranges from more or less the same price as to approximately 30% more. More expensive alternatives have been marketed, but these would probably not be the first choice substitutes for general application.

The table below provides a summary of the prices of a number of potential alternatives to EPS/XPS insulation materials by application, quoted in terms of ability to achieve the same level of thermal resistance.

Table 3.12 Examples of Prices for Selected Products (Klif, 2011)

Material	Price for 100mm(€/m^2)	Price for Functional Unit (€/m^2) [Note 1]
Flat roof insulation		
EPS	€13-18	€13-18
XPS	€23-27	€22-24
Mineral wool	€22-40	€23-48
PUR/PIR	€24	€16
Floor insulation		
EPS	€13-18	€13-18
XPS	€20	€20
PUR/PIR	€23-25	€17-18
External wall insulation		
EPS	€15	€15
Mineral wool	€16-20	€16-21

Notes:
1: Functional unit: insulation needed for thermal resistance of 2.857 m2·K/W corresponding to 10 cm insulation at a thermal conductivity 0.035 W/(m*K).
2: It is important to note that prices for the insulation materials vary significantly depending on application (roof, wall and roof insulation) and quality of individual products will vary amongst applications and brands. Parameters such as the thermal conductivity or compressive strength of the materials required for each use have a great impact on the price. For example, the price of an EPS board increases by about 8% going from a board with a thermal conductivity of 0.040 W/(m·K) to a board of a thermal conductivity of 0.035 W/(m·K).
Source: Klif (2011)

As can be seen from the above, EPS has lower purchase costs than XPS and the alternatives analysed, for an equivalent volume. However, whilst PUR/PIR foams are generally more expensive than EPS when considered in terms of the same thickness of material, the prices are comparable to or lower than EPS/XPS when considered in terms of the amount required to achieve comparable thermal efficiency. On the contrary mineral wools, especially stone wool, have lower insulation efficiency per unit thickness and therefore are typically more expensive than EPS when compared in terms of achieving equivalent thermal insulation.

Additionally it is important to consider that lifecycle energy consumption of the materials differs. Although EPS has a lower price than most alternatives for similar thermal performance, it can have higher embodied energy in manufacture and disposal (as set out in Klif, 2011). PUR/PIR foams, for example, have lower life-cycle energy costs to produce each functional unit and disposal per functional unit compared to EPS. Stone wool, by contrast, has higher energy consumption throughout manufacture and disposal than both EPS and PUR/PIR. The costs associated with these differences have not been quantified here, not have associated changes in e.g. air pollutant emissions

3.6 Trends

HBCDD has been identified as a Substance of Very High Concern (SVHC) and is now on the list of substances subject to authorisation in Annex XIV of the REACH Regulation. It is clear that the inclusion of the substance on the Candidate List and then the authorisation list have been a driver for companies to move away from use of the substance. As a result, consultation for the current study has identified that the use of HBCDD in textiles and HIPS has been reduced significantly.

In addition, HBCDD is currently being reviewed under two parallel regulatory processes at UN level: the UNECE Convention on Long-range Transboundary Air Pollution (LRTAP) and the UNEP Stockholm Convention on Persistent Organic Pollutants (POPs). The decision on a potential listing of HBCDD is expected at the earliest by the end of 2012 at UNECE level and mid-2013 at UNEP level. In particular, the Persistent Organic Pollutants Review Committee, at its eighth meeting in October 2012, recommended listing HBCDD with time-limited exemptions for production and use in polystyrene foam in buildings.

The recommendation[11] of the POP RC to the Parties to the Convention states: “*Decides, in accordance with paragraph 9 of Article 8 of the Convention, to recommend to the Conference of the Parties that it consider listing hexabromocyclododecane in Annex A to the Convention with specific exemptions for production and use in expanded polystyrene and extruded polystyrene in buildings*”;

It was the UNEP’s conclusion that to enable a smooth transition in the substitution of HBCDD in expanded and extruded polystyrene (EPS/XPS), time limit exemptions could be given together with a description of the conditions for production and for uses. If the recommendation is adopted, it will allow sufficient time to phase in chemical drop-in alternatives to HBCDD and will stimulate manufacturers to switch to other flame retardants in the following years. In addition, it will effectively end the use of HBCDD in other applications where alternatives are technically and commercially available (HIPS and textiles).

As indicated previously, the POP RC decision was taken based on the Risk Management Evaluation (RME) adopted in 2011 at its 7th meeting and on additional information regarding production, use and alternatives to HBCDD gathered by the committee. The RME also lists other existing risk management measures applicable at a European level:

- HBCDD is included as part of the brominated flame retardants group in the List of Substances for Priority Action of The Convention for the Protection of the Marine Environment of the North-East Atlantic (the OSPAR Convention). Also the Helsinki Commission (HELCOM) has included HBCDD in the list of priority hazardous substances.

- The EU’s Directive on Waste Electrical and Electronic Equipment (WEEE Directive 2012/19/EU) requires the removal of plastics containing brominated flame retardants and of printed circuit boards

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[11] Decision POPRC-8/3 is included in the document containing all information on HBCDD in preparation for COP6, available here:http://chm.pops.int/Convention/POPsReviewCommittee/LatestMeeting/POPRC8/POPRC8Followup/HBCDRecommendation/tabid/2912/Default.aspx

from electrical and electronic equipment prior to recovery and recycling. (This is likely to be an additional driver for the move away from use of the substance in HIPS.)

- Regulation (EU) No 305/2011 of the European Parliament and of the Council of 9 March 2011 laying down harmonised conditions for the marketing of construction products and repealing Council Directive 89/106/EEC. The Regulation's objective is to ensure reliable information on construction products in relation to their performances. HBCD appears on the indicative list of regulated dangerous substances possibly associated with construction products under the CPD[12].
- A proposal for a national ban of HBCDD is currently under consideration by the Norwegian Ministry of the Environment (EBFRIP, 2010)[13].

Despite these management measures, recent industry data suggests that production of HBCDD has increased in recent years and that this trend could continue in future years. This is thought to be attributable to the combination of existing fire requirements on the use of insulation materials in many Member States and to the increased demand for energy and thermal efficiency in the construction sector, which has been largely driven by the EU Directive on energy performance in buildings (2010/31/EU).

It is important to note that the use of HBCDD is dependent on national fire safety requirements, which vary from country to country within the EU. In countries with very stringent fire safety regulations the use of HBCDD is expected to be higher than in other countries like Norway or Sweden with more flexible regulations for the fulfilment of the fire requirements needed.

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[12] Indicative list of regulated dangerous substances possibly associated with construction products under the CPD (DS 041/051 rev.12, 9 March 2012)

[13] http://www.ebfrip.org/uploads/Press/documents/EBFRIP%20Position%20on%20Norwegian%20action%20plan%20on%20BFrs_20100115.pdf

4. Data Analysis

4.1 Uses and Releases

4.1.1 Overview

Much of the information presented in this section is based on market data available in publicly available sources, mainly BASF Plasticportal (2011) and PlasticEurope, (2010). Data on emissions is mainly based on the IOM (2008) study.

4.1.2 Market overview

As discussed previously, the polystyrene foam industry is a highly decentralised and fragmented industry, with many small and medium-sized enterprises (SMEs) involved in the production and conversion of EPS and XPS boards. The industry[14] has indicated that this sector provides direct and indirect employment to some 140,000 people EU-wide. Moreover, it is estimated that the total value chain using insulation foams generates employment for 430,000 people across the EU.

Table 4.1 shows estimates (based on available information and consultation) of total volumes of HBCDD-based EPS/XPS consumed in the EU. As indicated previously it has been assumed that HBCDD is used in 80% of EPS foams and 100% of XPS foams. The overall market for HBCDD-based EPS is based on the data published by PlasticsEurope. In the case of XPS it is important to note that the consumption includes Russia and Turkey, but no reliable alternative figures appear to be available.

Table 4.1 Summary of assumed quantities of EPS/XPS containing HBCDD consumed in the EU in 2011

Use	Quantity (mt/year)	Comment
EPS-containing HBCDD	0.70	Assumes 80% of total production into construction and 80% of foams being flame retarded with HBCDD. (European consumption 1.1 million tonnes) [Note 1]. According to Klif (2011), EPS consumption share by application in construction industry is 39% for external walls, 41% for floor construction, 13% and 1% for flat roof and pitched roof respectively and the remaining 6% for other applications. The 6% use in other applications has been prorated among the main applications.
Roofing insulation (15%)	0.11	
External wall insulation (41%)	0.29	
Floor insulation (44%)	0.31	
XPS-containing HBCDD	0.53	Assumes 100% of total production into construction and 100% of foams being flame retarded with HBCDD (European consumption 0.53 million tonnes) [Note 2]. According to Klif (2011), XPS consumption share by application in construction industry is 38% for external walls, 38% for floor construction, 18% and 1% for flat roof and pitched roof respectively and the remaining 5% for other applications. The 5% use in other applications has been prorated among the main applications. The cheapest alternative has been considered for each of these applications.
Roofing insulation (20%)	0.11	
External wall insulation (40%)	0.21	
Floor insulation (40%)	0.21	

Notes:
1: The figure of 0.70 million tonnes from above is broadly comparable with estimated EPS consumption data for 2006 (1.06 million tonnes), derived from a consumption of 5,300 tonnes of HBCDD in EPS and a 0.5% HBCDD content in EPS boards (w/w) (see section 3.3.2)
2: The figure of 0.54 million tonnes from above is broadly comparable with estimated XPS consumption data for 2006 (0.39 million tonnes), derived from a consumption of 5,900 tonnes of HBCDD in EPS and a 1.5% HBCDD content in XPS boards(w/w) (see section 3.3.3)

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[14] http://www.bsef.com/uploads/MediaRoom/documents/eps_xps_factsheet_november_final.pdf

4.1.3 Environmental releases

HBCDD may be released into the environment during production and manufacturing, processing, transportation, handling or storage, and from the use or ultimate disposal of this substance or products containing this substance (RME, 2011). As HBCDD is used as an additive it is not bound chemically to the polymer material, having the potential to migrate or evaporate out of the polymer. Releases can be from point source discharges or diffuse releases from use of finished products.

Annual HBCDD emissions into air, surface water and wastewater in Europe were estimated at 649 kg, 924 kg and 1,553 kg, respectively, resulting in a total yearly release of approximately 3 tonnes of HBCDDD (IOM (2008), based on consumption figures for 2006 supplied by the HBCDD Industry Users Group). It has been indicated that the environmental releases of HBCDD occur during the whole cycle of products containing HBCDD, although those generated from production and manufacture are estimated to be relatively small. In particular, the RME (UNEP, 2011) has indicated that the estimated total releases of HBCDD from manufacture and use of EPS/XPS insulation boards (95% of consumption of HBCDD) and manufacture and use of textiles (2% consumption) were of the same magnitude. Significant emissions into water generally result from its use in textile applications, whereas air emissions are produced mainly by its use in EPS and XPS insulation boards.

Textiles have not been covered in any detail in the current assessment as it is understood that this use will no longer be supported in the EU. Indeed due to reductions in use (based on consultation with industry) it can be assumed that releases from textiles will now be lower than quoted in 2008.

Minor polystyrene particles (dust) containing HBCDD can be released during the production, installation and use of EPS or XPS insulation and ultimately during the refurbishment or demolition of buildings containing these products. Table 4.2 shows the total estimated emissions of HBCDDD from EPS/XPS uses in Europe (IOM, 2008). Releases from waste disposal are not included as these have been difficult to estimate because of the long lifetime of polystyrene foams once installed in buildings (potentially up to 50 years).

Table 4.2 Environmental releases from XPS/EPS containing HBCDD in the EU (IOM, 2008 and ECHA, 2009)

Use	Air (kg/year)	Wastewater(kg/year)	Surface water(kg/year)
Use of HBCDD in EPS formulation	30	75	330
Use of HBCDD in XPS formulation	14	84	10
Industrial use of EPS compound in the manufacture of flame retarded EPS	159	128	31
Industrial use of XPS	146	63	16
Installation of professional insulation boards (EPS and XPS)	236	0	236
Building insulation (EPS and XPS) during service life	70	0	0

The release of HBCDD from XPS/EPS products during service life is small. However important uncertainties remain regarding emissions during the consumer use of products and from waste disposal. It is expected that emissions may grow in the future, as increasing numbers of buildings containing HBCDD-treated EPS become subject to refurbishment or demolition (IOM, 2008).

As indicated previously, emissions from textiles have been excluded from the scope of analysis. These are considered to be currently low following a substantial reduction of this use in Europe during the last few years. It is of note that HBCDD releases from textile products during service life were smaller than that quoted for EPS/XPS (i.e. 26 kg/year compared to 70 kg/year) based on 2006 data (ECHA, 2009) even before the more recent reduction in use due to replacement of HBCDD in textiles.

Note that, for the purpose of this study, abatement cost curves have been developed in terms of cost per unit of use removed rather than per unit of environmental release abated.

4.2 Current and Planned Abatement Measures

Relevant mandatory measures that are already in place are discussed in Section 3.6.

In addition, European HBCDD and polystyrene manufacturers have initiated several voluntary initiatives, which are targeted at eliminating emissions from first line users of HBCDD. Key emission reduction programmes are listed below (BSEF, 2012):

- Self-Enforced Control of Use to Reduce Emissions (SECURE) is addressed to downstream users in the EPS and XPS sector. The members of the associations PlasticsEurope and EXIBA (European Extruded Polystyrene Insulation Board Association) that committed to SECURE represent 95% of the total HBCDD consumption of these associations (BSEF, 2012).

- Voluntary Emissions Control Action Programme (VECAP) is addressed to producers and downstream users. The programme involves the possibility of a certification procedure based on ISO 9001 and 14001 principles. Under the VECAP framework, the only site that produces HBCDD in Europe uses state of the art technology and has developed control methods for air, water and solid waste emissions. The plant has been certified under VECAP since 2009 (BSEF, 2012). More specifically, the 2011 VECAP report demonstrates a reduction of 11% in the potential emissions of HBCDD to the environment compared to 2010, while at the same time the total sales volume covered by the programme increased.

As a result of the implementation of best practices through these programmes, VECAP and SECURE, potential emissions of HBCDD to the environment have been reduced by 80% since 2008. Within the context of these programmes, a “Code of Good Practice” was developed to support users in their effort to reduce emissions, including advice on the best ways to store, handle and use products and waste. Industry’s aim is to ensure that all European users of HBCDD are covered by VECAP or SECURE (BSEF, 2012).

A key uncertainty at this stage is the extent to which the manufacturers and users of HBCDD will apply for (and be granted) authorisation under REACH. Depending on these issues, there could potentially be significant changes in the manufacture, use and releases of HBCDD in the near future.

4.3 Possible Future Abatement Measures

4.3.1 Information sources

Based on the information reviewed above, a number of potential further measures are possible. These could include flame retardant chemical substitution, product redesign and material substitution. The first measure will not be feasible in a period of time of about 3-5 years and the second one is not an option in certain Member States with strong fire safety requirements. The third measure regarding the use of alternative insulation materials to replace the use of HBCDD in EPS/XPS foams is the only measure considered in the cost curve data, because information on prices are available. However, in practice, it is likely that a significant proportion of HBCDD will be replaced with chemical alternatives, including the one under development (see section 3.5.2), but no data on prices are currently available.

4.3.2 Measures included in the cost curve

For the purposes of the abatement cost curve development, it has been assumed that HBCDD would be substituted by the least cost-alternative (for which cost information is available) in each of the applications considered (floor, external wall and roofing insulation). In practice, it is likely that a range of different alternatives would substitute some of the EPS/XPS use, depending on users' preferences, but insufficient information is available on the proportions in which they might be used.

The following measures have been taken forward as part of the cost curve development[15]:

- Replacement of HBCDD in EPS used for external wall insulation with mineral wool (100% of use);
- Replacement of HBCDD in EPS used for floor insulation with PUR/PIR (100% of use);
- Replacement of HBCDD in EPS used for roof insulation with PUR/PIR (100% of use);
- Replacement of HBCDD in XPS used for floor insulation with PUR/PIR (100% of use);
- Replacement of HBCDD in XPS used for external wall insulation with mineral wool (100% of use);
- Replacement of HBCDD in XPS used for roof insulation with PUR/PIR (100% of use).

4.4 Cost Curve

Table 4.3 provides a summary of the key data on each of the measures for inclusion in the cost curve. Further details are included in the supplementary spreadsheet and a description of the calculations and assumptions is provided in the Appendix to this report.

Figure 4.1 presents the marginal cost curve.

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[15] Substitution with either PUR/PIR or mineral wool has been assumed in all cases, with the least cost alternative being assumed to be used. In some cases, cost information was not available for some of the available alternatives.

We have attempted to calculate all costs in line with ECHA Guidance on Compliance Costs[16]. However, it is important to note that the cost curves are based on the current prices of alternatives that are already on the market. In practice, one of two outcomes is possible for companies that currently manufacture EPS/XPS in the EU, either:

1. They will cease this area of their business. It is understood that the existing technology used for EPS/XPS manufacturing cannot be adapted to alternative insulation materials. This equipment therefore has no alternative uses and its opportunity cost is effectively zero. Any residual value which the equipment had prior to the imposition of any restriction on use is would therefore be lost. For the purposes of the current analysis, this rent has not been included in the cost curves. However, this could be significant.

2. Or, they would make necessary investments to supply alternatives. It is assumed that the costs of such investments are reflected in the current prices of the alternatives that are already on the market.

The costs per tonne of HBCDD removed are all based on the assumption that the alternative adopted is that which has the least cost to achieve an equivalent level of insulation as EPS/XPS containing HBCDD. In the case of EPS, the alternatives generally have higher cost to achieve the same level of insulation, whereas for XPS, the alternatives have lower cost (giving a negative value for the cost per tonne of HBCDD use avoided). It is important to note, however, that the costs only include the difference in prices between HBCDD-based products and the alternatives. They do not include the (potentially significant) cost implications that could occur as a result of the lost market, and lost residual value of capital equipment, for the current suppliers of HBCDD-based EPS and XPS to the insulation market. If such costs are included, it is therefore possible that there could be significantly higher costs for substitution of EPS and positive rather than negative costs for substitution of XPS. It has not been possible to quantify these effects here. Furthermore, other uncertainties include price variations in EPS/XPS and the alternatives (e.g. related to raw materials, energy consumption or individual product quality) which will affect relative prices of materials, on a temporal and geographic basis.

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[16] Addendum to the Guidance on Socio-economic Analysis – Restrictions: Calculation of compliance costs. Available at http://echa.europa.eu/documents/10162/17087/appendix1-calculation__compliance_costs_case_restrictions_en.pdf.

Table 4.3 Summary of cost curve data

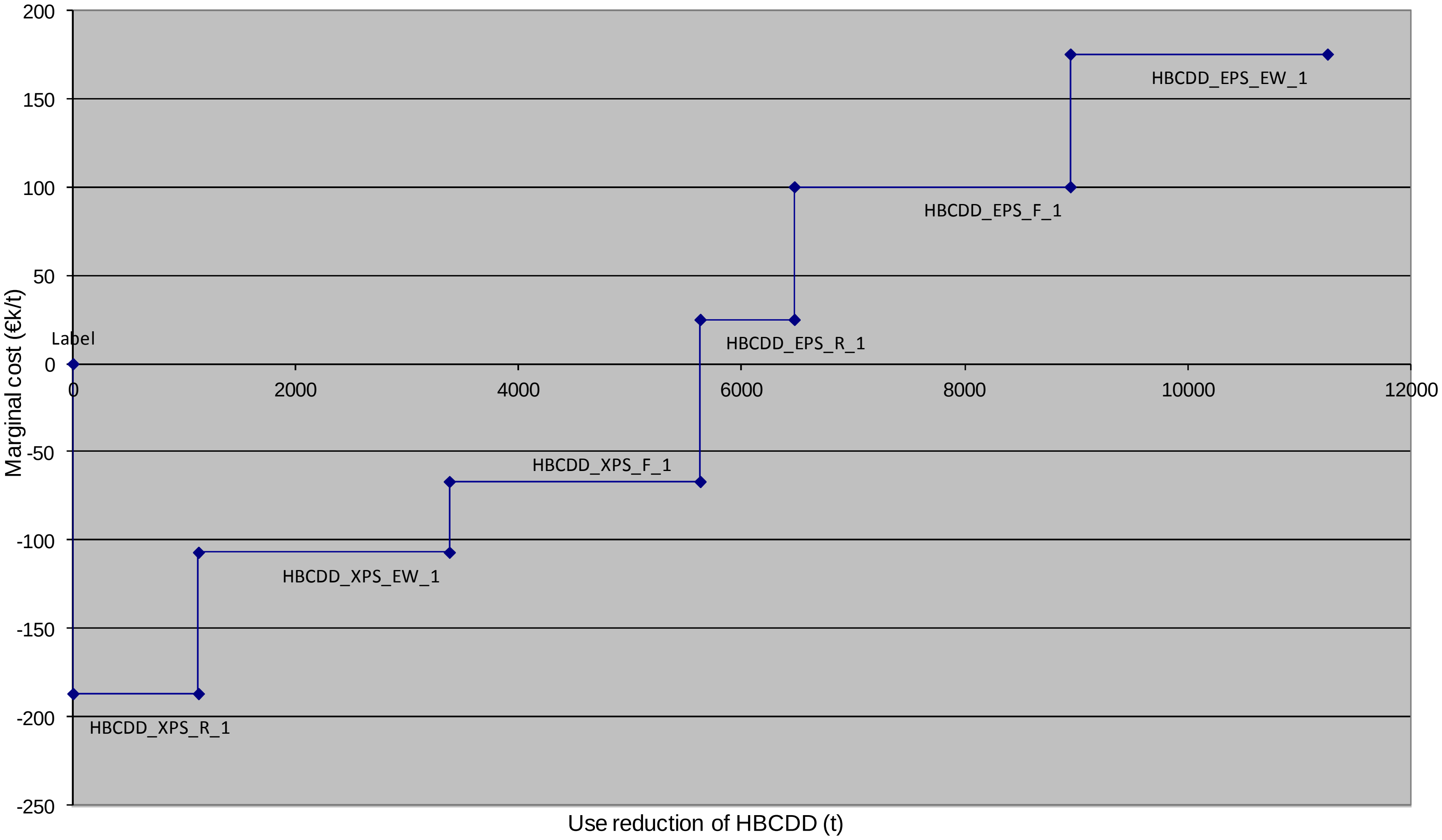
Measure	Incremental cost (€k)	Incremental emission reduction (t)	Incremental cost-effectiveness (€k/t)	Notes
HBCDD_XPS_R_1	-210,000	1,125	-187	Replacement of HBCD in XPS used for roof insulation with PUR/PIR
HBCDD_XPS_EW_1	-240,000	2,250	-107	Replacement of HBCD in XPS used for external wall insulation with mineral wool
HBCDD_XPS_F_1	-150,000	2,250	-67	Replacement of HBCD in XPS used for floor insulation with PUR/PIR
HBCDD_EPS_R_1	21,120	844	25	Replacement of HBCD in EPS used for roof insulation with PUR/PIR
HBCDD_EPS_F_1	247,808	2,475	100	Replacement of HBCD in EPS used for floor insulation with PUR/PIR
HBCDD_EPS_EW_1	404,096	2,306	175	Replacement of HBCD in EPS used for external wall insulation with mineral wool

Notes:

1) All data on costs and emission reductions are at an EU level.

2) See Appendix A for details on how these figures have been calculated

Figure 4.1 Marginal cost curve for HBCDD in EPS and XPS foams for insulation in buildings



5. Conclusions

The cost curves developed illustrate the relative costs, in €/t of use avoided, of substituting HBCDD in XPS/EPS with alternative insulation materials in each of the main applications where EPS/XPS has a significant share (floor, external wall and roofing insulation). However, as noted above, an alternative chemical to HBCDD is expected to be available in the near future. It is likely that alternative chemicals would be preferable to the use of alternative insulation materials (and more widely adopted), given that the likely implications for changes in plastics manufacture/processing would be much less significant. However, no data on prices are currently available and therefore chemical substitutes could not be considered in the cost curves.

Replacement of HBCDD in XPS used for roof insulation was found to be the most cost-effective measure (-€187k/t) in terms of use, whilst replacement of HBCD in EPS used for external wall insulation was found to be the least cost-effective measure (€175k/t). In general terms, alternatives have higher cost to achieve the same level of insulation of an EPS board, whereas for XPS, the alternatives have lower cost (negative values).

However, these costs may underestimate the true costs for two reasons. Firstly, the measures assume 100% uptake of the least-cost alternative (for which cost information is available). This is a simplistic analysis as, in practice; it is likely that a range of different alternatives would be used, depending on users' preferences. More sophisticated cost curves would require further data collection and analysis in order to understand the likely uptake of different alternatives. Furthermore, product prices vary significantly according to product type, geographical location, etc. and there are thus uncertainties associated with the figures used in the cost curves, which may make all of the figures higher or lower.

Secondly, the cost curves are based on the current prices of alternatives that are already on the market. This assumes that the costs of any investments required are reflected in the current prices of the alternatives that are already on the market. In the event of replacement of EPS/XPS with alternatives, there could be a loss in the residual value of capital equipment (for the EPS/XPS manufacturers and formulators) which has not been considered in the cost curves. It is important to note, therefore, that costs incurred by individual companies currently involved in the production and supply of HBCDD-based products are not included in the cost curves. These costs could involve significant implications for the companies concerned (related to loss of market and associated employment) if they are not able to supply alternatives and/or loss of the residual value of their investment in capital equipment that they currently use to produce HBCDD-based products.

Furthermore, there are a number of wider implications associated with the different insulation products which have not been taken into account in the cost curves, such as differences in energy use during aspects of the life-cycle other than in-service use (disposal and manufacture).

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Appendix A
Data for Incorporation into Cost Curves

Table A.1 Summary of measures for inclusion in cost curves

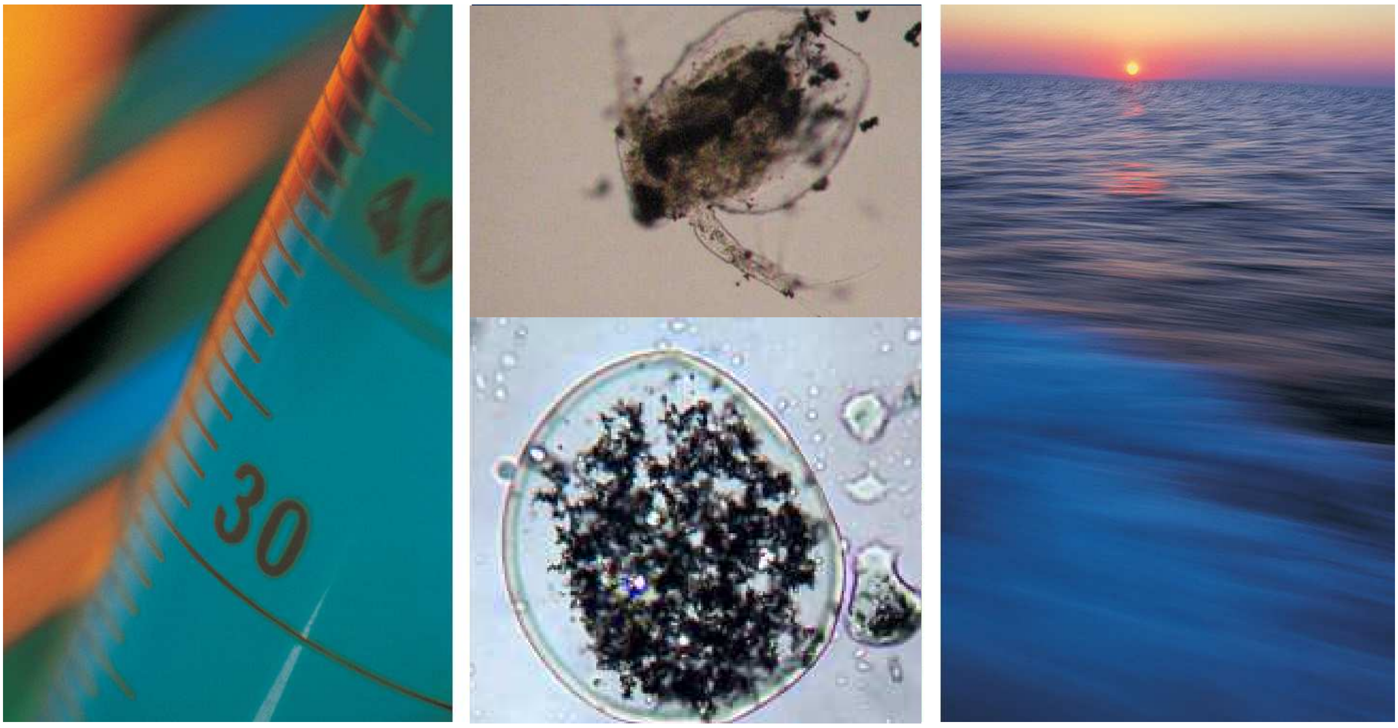
Measure	Details of key elements of measure
Replacement of HBCDD in EPS foams	*One-off costs:* Sufficient information on total R&D costs for replacement of HBCDD based EPS is not available. However, switching to other construction alternatives will imply plant pilot trials, process start up, and product qualification. Also there will be costs associated with rebranding of new products, as well as loss of the residual value of capital equipment of EPS manufacturers. *Recurring annual costs:* Operational costs have been calculated based on the price of alternatives in the applications considered (external wall, roofing and floor insulations). These have been extracted from the report prepared by COWI for the Norwegian Climate and Pollution Agency (Klif) in 2011. Prices correspond to 2011/10 and have not been adjusted for inflation. Also, the report does not indicate whether their price estimates include VAT – but we have assumed that they have not. Prices have been normalised at a functional unit that achieves the same level of thermal resistance. The functional unit corresponds to the insulation needed for a thermal resistance of 2.857 m2·K/W corresponding to 10 cm insulation at a thermal conductivity 0.035 W/(m*K) like a typical EPs board. The cost of raw materials; costs of other inputs such as energy and water and costs of worker health and safety have not been considered in the calculations. According to Klif (2011), EPS consumption share by application in construction industry is 39% for external walls, 41% for floor construction, 13% and 1% for flat roof and pitched roof respectively and the remaining 6% for other applications. The 6% use in other applications has been prorated among the main applications. The least expensive alternative has been considered for each of these applications. *Emissions reduced:* 100% per cent substitution of EPS containing HBCDD is assumed. Total emission reduction = 5.6 t of HBCDD. It has been assumed a 50% share of HBCDD used in XPS and EPS based on IOM, 2008. In 2006, the ratio of XPS versus EPS was about 52:48. *Applicability of measure:* Mineral wools and PUR/PIR foams are considered to be the most likely alternatives to EPS containing HBCDD based on conclusions of the Klif (2011) study. See section 3.5 on technical and price feasibility of the alternatives considered. Chemical alternatives will probably also be adopted in practice, but no cost/price information is available at present. *Specific measures taken forward:* HBCDD_EPS_EW_1: Replacement of HBCDD in EPS used for external wall insulation with mineral wool (100% of use); HBCDD_EPS_F_1: Replacement of HBCDD in EPS used for floor insulation with PUR/PIR (100% of use); HBCDD_EPS_R_1: Replacement of HBCDD in EPS used for roof insulation with PUR/PIR (100% of use); Replacement of EPS-based roofing insulation with PUR/PIR foams costs €25,000/t; of EPS-based external wall insulation with mineral wools €175, 200/t and of EPS-based floor insulation with PUR/PIR foams €100,120/t. In total these adds up to €300,000/t of HBCDD. *Main uncertainties and limitations* • Data on one-off costs are insufficient for their inclusion in the analysis. • Data on costs of alternatives are based on limited information (only German market prices) and subject to uncertainty. No data on prices of PIR/PUR regarding external wall insulation and of mineral wools regarding floor insulation.

Measure	**Details of key elements of measure**
Replacement of HBCDD in XPS foams	*One-off costs:* Sufficient information on total R&D costs for replacement of HBCDD based XPS is not available. However, switching to other construction alternatives will imply plant pilot trials, process start up, and product qualification. Also there will be costs associated with rebranding of new products, as well as loss of the residual value of capital equipment of EPS manufacturers *Recurring annual costs:* Operational costs have been calculated based on the price of alternatives in the applications considered (external wall, roofing and floor insulations). These have been extracted from the report prepared by COWI for the Norwegian Climate and Pollution Agency (Klif) in 2011. Prices correspond to 2011/10 and have not been adjusted for inflation. Also, the report does not indicate whether their price estimates include VAT – but we have assumed that they have not. Prices have been normalised at a functional unit that achieves the same level of thermal resistance. The functional unit corresponds to the insulation needed for a thermal resistance of 2.857 m2·K/W corresponding to 10 cm insulation at a thermal conductivity 0.035 W/(m*K) like a typical EPs board. The cost of raw materials; costs of other inputs such as energy and water and costs of worker health and safety have not been considered in the calculations. According to Klif (2011), XPS consumption share by application in construction industry is 38% for external walls, 38% for floor construction, 18% and 1% for flat roof and pitched roof respectively and the remaining 5% for other applications. The 5% use in other applications has been prorated among the main applications. The least expensive alternative has been considered for each of these applications. *Emissions reduced: 100% per cent substitution of EPS containing HBCDD is assumed. Total emission reduction = 5.6 t of HBCDD. It has been assumed a 50% share of HBCDD used in XPS and EPS based on IOM, 2008. In 2006, the ratio of XPS versus EPS was about 52:48.* *Applicability of measure:* Mineral wools and PUR/PIR foams are considered to be the most likely alternatives to EPS containing HBCDD based on conclusions of the Klif (2011) study. It is noted that these will also be suitable for XPS. See section 3.5 on technical and price feasibility of the alternatives considered. Chemical alternatives will probably also be adopted in practice, but no cost/price information is available at present. *Specific measures taken forward:* HBCDD_XPS_EW_1: Replacement of HBCDD in XPS used for external wall insulation with mineral wool (100% of use); HBCDD_XPS_F_1: Replacement of HBCDD in XPS used for floor insulation with PUR/PIR (100% of use); HBCDD_XPS_R_1: Replacement of HBCDD in XPS used for roof insulation with PUR/PIR (100% of use); Replacement of EPS-based roofing insulation with PUR/PIR foams costs €-186, 670/t; of EPS-based external wall insulation with mineral wools €-106,000/t and of EPS-based floor insulation with PUR/PIR foams €-67670/t. In total these adds up to €-36000/t of HBCDD. *Main uncertainties and limitations* • Data on one-off costs are insufficient for their inclusion in the analysis. • The applicability of alternatives to XPS uses is based on limited information. In this sense the Klif (2011) report focuses on EPS boards and it is not clear the extent to which these solutions could substitute XPS foams. • Data on European consumption of XPS include Russia and Turkey. • Data on prices of XPS and alternatives are based on limited information (only German market prices) and subject to uncertainty. In particular, the price of XPS boards regarding external wall insulation has been derived from the average price reported for other applications (floor and roofing). • No data on prices of PIR/PUR regarding external wall insulation and of mineral wools regarding floor insulation.

European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

MDA - Final Report



AMEC Environment & Infrastructure UK Limited

December 2012

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3	Revised final report taking into account ECHA comments	27 Sept 2012
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European Chemicals Agency

Abatement Costs of Certain Hazardous Chemicals

MDA - Final Report

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Executive Summary

This report provides the results of work on collection of data on abatement costs of reducing the use of MDA. The work involved developing cost curves based on information readily available in the literature in combination with consultation with industry and relevant associations. It has been undertaken on behalf of ECHA by AMEC.

MDA was identified as a Substance of Very High Concern (SVHC) pursuant to Article 57(a) as it is classified as Carcinogenic, Category 2 and was therefore included in the candidate list for authorisation following ECHA's decision ED/67/2008 on 28 October 2008. MDA is now included on Annex XIV of the REACH Regulation with a sunset date of 21 August 2014 and a latest application date of 21 February 2013. Therefore the current study has concentrated on assessing abatement costs of direct alternatives in all applications, excluding the use as an intermediate.

Three uses were identified as potential non-intermediate uses for further examination: hardeners in epoxy resins, hardener in adhesives, and use in polyimides (PMR-15).

During the course of this study, no companies were identified that are currently using 4,4-MDA as either a hardener in epoxy resins or in adhesives. It is suggested that since 2008 the use of MDA has been phased out in both of these applications. It is suggested this could be, in part, as a result of the classification of the substance as a carcinogen and the inclusion of the substance on the Candidate List and then the authorisation list.

The principal identified use relates to use in polyimides (PMR-15). Information available for this study suggests that PMR-15 is currently being used in Europe. It is believed that whilst there may be other (unidentified) companies using PMR-15 in Europe, the total number of companies using the substance in polyimides is relatively limited. The available data on abatement potential and costs for 4,4-MDA in polyimides is limited. It appears that there are functionally alternative products available that can be used in place of PMR-15 for some applications, though given the high performance nature of the articles it is used in, it is not likely to be straightforward to simply replace PMR-15 with the lowest-price available alternative. It has not been possible to estimate the quantity of 4,4-MDA used in PMR-15 in the EU. As a result, it has not been possible to develop cost curves for substituting MDA for use in polyimides.

Contents

1. Project Understanding

1.1 The Project

The European Chemicals Agency (ECHA) has commissioned a project to provide information on "abatement costs for certain hazardous chemicals" (contract number ECHA/2011/140). The work is being undertaken by AMEC Environment & Infrastructure UK Limited ("AMEC").

The present report is intended to provide a summary of the data collected on abatement costs of reducing the use of the substance 4,4'-Diaminodiphenylmethane (MDA) (CAS number 101-77-9, EC number 202-974-4).

The data collected is intended to be used for:

- Supporting the Agency in assessing the most appropriate risk management options for the substances addressed;
- Furthering the understanding of the usefulness of data on use/emissions abatement costs in risk management decision-making; and
- Supporting the Agency in the preparation of restriction dossiers.

1.2 Project Context

This project follows on from a 2010 project on "Abatement cost curves for substances of concern" conducted by AMEC Environment & Infrastructure UK (previously Entec UK) for the Environment Agency, ECHA, the Health and Safety Executive (HSE) and RIVM. The main aim of that project was to develop a suitable method for estimating abatement costs to reduce emissions of chemicals and to apply and test the method with three selected case study substances. That study provided a first illustration of the benefits of being able to compare unit abatement costs amongst different substances and different uses.

The objective of this project is to assist ECHA in establishing capability to assess the abatement costs of reducing the use or emissions of hazardous substances. Under this lot ("Lot 5"), abatement potential and costs for MDA have been assessed.

MDA was identified as a Substance of Very High Concern (SVHC) pursuant to Article 57(a) as it is classified as Carcinogenic, Category 2 and was therefore included in the candidate list for authorisation following ECHA's decision ED/67/2008 on 28 October 2008. MDA is now included on Annex XIV of the REACH Regulation with a sunset date of 21 August 2014 and a latest application date of 21 February 2013. Therefore the current study has concentrated on assessing abatement costs of direct alternatives (i.e. alternative substances or techniques) in all applications, excluding the use as an intermediate.

The main outputs of the work, for this substance and for the other substances being assessed under different lots, are expected to be as follows:

- Data on abatement costs of reducing the use or emissions of the chemical. Different applications of the same substance may/will introduce different abatement costs.

- An overview on the functioning of the markets for the substances in question, including information on prices, amounts of the substance on the markets (including import and export), the relative shares of the substance used for different applications, number of actors involved in the business, as well as possible trends in the relevant market.

1.3 Structure of the Report

This report is structured as follows:

- Section 2 of this report provides an overview of the process of data gathering for this study and presents an overview of the data received;

- Section 3 sets out an overview of the market for MDA;

- Section 4 presents the data analysis and resulting abatement cost information;

- Section 5 presents conclusions.

The appendices to this report include various other background data. Some of the information in these appendices is confidential and should not be distributed outside ECHA.

2. Data Collection

2.1 Overview

The aim of the study was to gather abatement costs data on the use of alternative substances or techniques to MDA in all applications (with the exception of use as an intermediate). This chapter briefly summarises the process of data gathering for this study and presents an overview of the data received.

2.2 Review of Existing Data Sources

The starting point for this study was an Entec (2008) report for ECHA which sought to provide information (including data on uses, releases and potential alternatives) on substances of very high concern including MDA, as well as the risk assessment report for the substance (EC, 2001).

This information presented here has been collected by using the sources mentioned above, supplemented with further relevant sources identified during the course of the study and consultation with relevant industry organisations and trade associations.

2.3 Stakeholder Consultation

There was targeted consultation with stakeholders by telephone and email; in total, 18 organisations were contacted (see confidential appendix). These stakeholders included the REACH registrants for the substance, relevant trade associations for key uses (see Section 3.3) and downstream users identified as *potentially* using the substance in non-intermediate uses.

3. General Market Overview

3.1 Introduction

MDA is synthesised by reaction of formaldehyde and aniline in the presence of hydrochloric acid. This reaction is carried out either in a batch reactor or in a continuous process (EC, 2001). The reaction does not lead to a single product, but to polymeric MDA (PMDA) consisting of mixtures of 4,4'-, 2,4'-, and 2,2'-isomers and oligomeric MDAs[1]. The amounts of MDA isomers and oligomers formed depend on the ratios of aniline, formaldehyde, and acid used, as well as the reaction temperature and time (Kirk Othmer, 2000a).

3.2 Manufacture and import/export of MDA

3.2.1 Historical Information

In 1989, 10 MDA-producing sites were reported in Europe (EC, 2001). In 2008, ISOPA (the European trade association for producers of diisocyanates and polyols) indicated that there were up to six companies producing MDA in the EU, but the locations of the sites were unknown (Entec, 2008).

In 1993, the production capacity of MDA was estimated at around 540,000t (EC, 2001). The Entec (2008) study estimated production of MDA to be in the region of 1.4m tonnes in 2008.

3.2.2 Current best estimates

The registration dossiers submitted for MDA suggest that currently between 10,000 and 50,000t of MDA per year are used in the EU[2]. See confidential appendix for more detailed information.

3.3 Overview of Uses

The vast majority of MDA is used as a precursor to methylene diphenyldiisocyanate (MDI). In 2008, it was estimated that around 98% of MDA is used to produce MDI (e.g. 1.38M tonnes). Other uses of MDA, identified through the registration information on ECHA's website and/or through further investigation for the current study, include:

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[1] The term MDA is sometimes used for pure 4,4'-MDA as well as the oligomeric mixture PMDA. Similar inconsistencies are encountered for the isocyanate derivatives (MDI and PMDI).

[2] It should be noted that several MDAs have been registered under REACH, including a full restriction dossier for 4,4'-MDA and an intermediate dossier for oligomeric MDA. The quantities of oligomeric MDA [CAS No 25214-70-4, EC No 500-036-1] are believed to be much greater than those for 4,4-MDA.

- As a hardener in epoxy resins and adhesives,
- In the production of high performance (PEEK) polymers,
- In the production of polyimides (e.g. PMR-15),
- In the production of polyamide-imide (PAI) polymers, and
- Processing to 4-4'methylenebis(cyclohexaneamine) (H_{12}MDA or PACM).

These uses are addressed in turn below.

Table 3.1 summarises the volumes of MDA used in different applications in 2008.

Table 3.1 MDA summary use volumes (Entec, 2008)

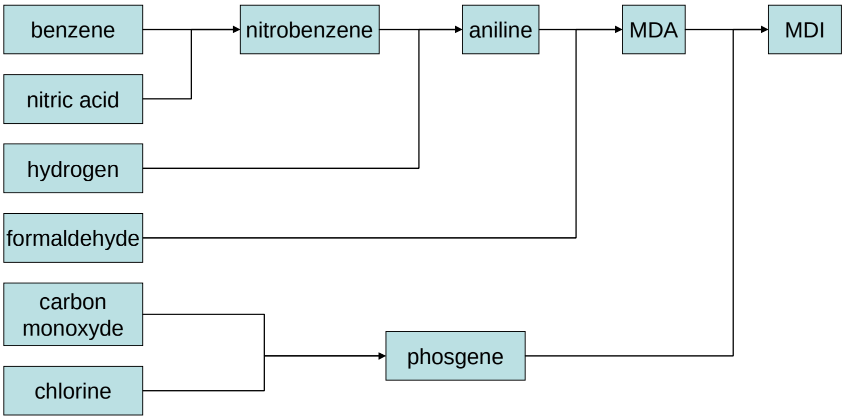
Use	Volume	Potential intermediate use?
Production of MDI	1,380,000	Y
Hardener in epoxy resins	~200	N
Hardener in adhesives	No data	N
Intermediate for high performance polymers	<5,000	Y
Production of methylenebis (cyclohexaneamine)	No data	Y

3.3.1 Manufacture of MDI

More than 99% of the manufactured PMDA products are used in reactions with phosgene to produce the corresponding isocyanates for use in polyurethanes (see Figure 3.1). The resultant polymeric isocyanates (PMDI) are either sold commercially or are purified to isolate 4,4'-methylenediphenyldiisocyanate (MDI) (CAS number 101-68-8) (Kirk-Othmer, 2000a). MDI is an important intermediate in the manufacture of spandex fibres, thermoplastic resins, and coatings and is used in reaction injection moulding (RIM) for automotive applications. The primary use of PMDI products is in rigid polyurethane foam insulation, but they are also used in semi-flexible foams, foundry core binders, and particle board manufacture.

MDA when used in the manufacture of MDI is considered by industry to be an on-site isolated intermediate, used under strictly controlled conditions (SCC) (Entec, 2008; ECHA, 2009).

Figure 3.1 **Steps in the manufacture of MDI (BASF, 2007)**



3.3.2 High performance / PEEK polymers

MDA is reportedly used as a chemical building block in the synthesis of the high-performance polymer, polyether ether ketone (PEEK) (BASF, 2012). PEEK can reportedly be moulded and is used in applications such as liquid chromatography fittings, for coatings, in electrical insulation for high temperature service and in composites (Kirk-Othmer, 2000b; Victrex, n.d).

PEEK is a thermoplastic polymer. Examples of the way in which PEEK is used include (Kirk-Othmer, 2000c):

- A commercial prepreg of PEEK and carbon fibres made by the hot melt process. This is reportedly quite stiff and boardy and makes it difficult to form structures with complex shapes.
- Fine (0.5-5μm) particles of PEEK for use in powder prepregging. PEEK resin is too tough to be ground to a fine powder suitable for powder prepregging.
- PEEK can also be spun into fibres, which are commingled with a reinforcing fibre to form a yarn. When heated above its melting point, the PEEK flows around the reinforcing fibres (typically carbon) and forms the resin matrix. The commingled yarn is woven into the shape desired and consolidated.

PEEK polymers can be supplied commercially as granules, coarse or fine powders, or glass-fibre/carbon-fibre filled.

Given that PEEK manufactured from MDA is a thermoplastic which can be subsequently melted and re-moulded. It seems to be generally supplied in a form (e.g. granules or powder) intended for subsequent use in manufacture of articles. Given that these other polymer substances are subsequently placed on the market, use in manufacture of high-performance PEEK polymers could be an intermediate use.

As indicated in Table 3.1, in 2008, up to 5,000t of MDA were used in the production of high performance polymers.

Information from consultation is provided in a confidential appendix.

3.3.3 Polyamide-imide polymers

Polyamide-imides are thermoplastic amorphous polymers that have valuable mechanical, thermal and chemical resistant properties. Polyamide-imide polymers can be processed into a wide variety of forms – from injection or compression moulded parts and ingots – to coatings, films, fibres and adhesives. For example, polyamideimide coatings are used on magnet wire (used in the winding of motors and other electrical devices) to improve abrasion resistance and toughness. The thermal properties of polyamideimide coatings are reported to be outstanding when compared to the cost of polyimide resins (Elantas, 2006).

One production route is through the condensation of an aromatic diamine, such as MDA and trimellitic acid chloride (TMAC), to form a polyamide-imide (PAI) polymer, which is then mixed with further monomer components and a solvent. It is understood that MDA is present at <0.05% in the mixture and therefore this use is considered to be exempt from authorisation. See appendix for further information.

3.3.4 Hardener

Hardener in epoxy resins

MDA is understood to be used as hardener in epoxy resin curing agents for coatings. The curing agent is mixed with epoxy resin to form a cured coating system which reacts chemically to form the final coating. Such coatings are used where high chemical resistance is required, such as coatings for tank linings. According to industry, MDA offers a number of important properties:

1. Long working life,
2. Specific mechanical properties, including the Glass Transition Temperature (T_g),
3. A high degree of chemical resistance.

In addition, MDA is used as a hardener for the manufacture of pipes using filament winding process. In 2008, Entec estimated that at least 200t/y of MDA are used as hardener in epoxy resin curing agents and at least 150 t/y of MDA is used exclusively in the filament would pipe application (Entec, 2008). However, based on information collected during the current study, it is thought that this is only oMDA, not 4,4’-MDA.

Hardener in adhesives

The information available on the use of MDA as hardener in adhesives suggests that industry has made significant efforts to phase out MDA.

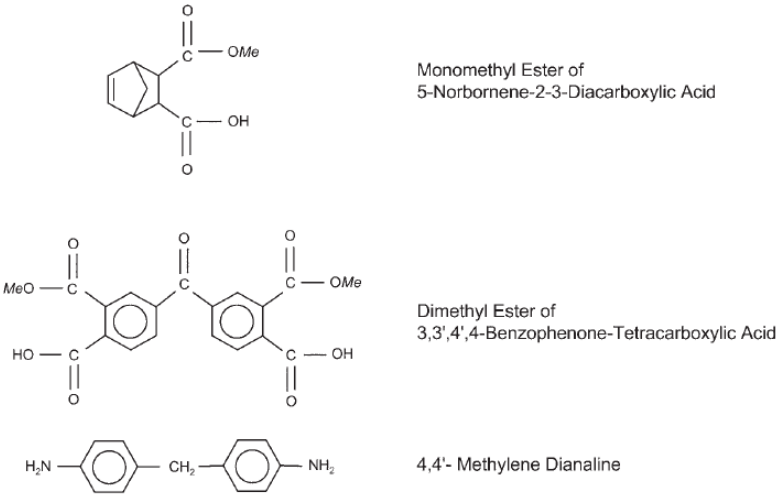
Summary

As stated in the background document by ECHA (2009), the use of MDA as hardener in epoxy resins and adhesives is not expected to be considered as a use of an intermediate in a manufacturing process of another substance but as an end use of the substance since it does not result in another substance which is manufactured/imported or placed on the market as such or in a mixture.

3.3.5 Polyimides (PMR-15)

Reportedly the best known of the addition-curing polyimide materials is PMR-15, in which three types of monomers (see below) are mixed together along with a solvent, usually methyl or ethyl alcohol. One of the monomers is 4,4-MDA.

Figure 3.2 Production of PMR-15



Several non-MDA polyimides have been formulated and sold, however their elevated temperature performance is reportedly not as good as that of the original formulation. PMR-15 undergoes condensation reactions early in the cure cycle which means that the volatile impregnation solvent must be removed before the resin gels, to avoid voids and porosity (Campbell, 2010).

Polyimides can be used at high temperatures (up to 250ºC wet/300ºC dry) and typical applications include missile and aero-engine components. They tend to be hard to process due to their condensation reaction emitting water during cure, and are relatively brittle when cured.

According to the literature PRM-15 prepegs are typically low in price compared to other PMR resins systems (Alston and Scheiman, 2000). They range from €100 – 260/kg, depending upon the type of reinforcement and quantity purchased (Kantz, 1990).

PMR-15 refers to “in situ polymerization of monomer reactants” and was developed by NASA. PMR-15 and other polyimides are thermosetting rather than thermoplastic. Applications of PMR-15-carbon fibre composites reportedly include jet-engine cowlings, ducts, compressor blades, and flaps and fairing (Kirk-Othmer, 2000c).

Given that polyimides are thermosetting, it is assumed that the polymerisation reaction needs to take place in-situ and that this results in creation of a finished article (i.e. through the combination of MDA, the other monomers, solvent and a carbon-fibre or other reinforcement[3], with subsequent curing).

One description[4] of PMR-15 includes the following: "PMR [...] technology used an alcohol solution of polyimide monomers to make "prepreg," graphite or glass fibre bundles impregnated with polyimide resins, which could be thermally cured into composites with low voids, eliminating the difficulty of removing high-boiling solvents often used for condensation (step-polymerization) polyimides"[5].

Overall, the use in polyimides seems to occur through a curing reaction involving MDA as a monomer and other starting monomers. PMR-15 in particular seems to involve the impregnation of these starting monomers, along with a solvent, into a glass-fibre or carbon-fibre reinforcement.

Following the workflow in the guidance on substances in articles, it would appear that MDA is present as part of a mixture, sometimes in combination with an article (e.g. glass-fibre or carbon-fibre). It would appear, therefore, that this use may not be an intermediate use.

3.3.6 Processing to 4-4'methylenebis(cyclohexaneamine) (H_{12}MDA or PACM)

MDA can be hydrogenated to 4-4'methylenebis(cyclohexaneamine) (H_{12}MDA or PACM). PACM can be subsequently used for the manufacture of the corresponding aliphatic diisocyanate (H_{12}MDI). H_{12}MDI is reportedly used for the manufacture of:

- Binders or hardeners for coating materials or adhesives (60%);
- Prepolymers (20%);
- Others e.g. for the production of elastomers (20%) (Entec, 2008).

It is also understood that PACM can be used itself as a hardener in epoxy resins.

3.4 Scope of uses covered in the current study

The scope of the current study covers uses of MDA that are not expected to be intermediate uses and that are hence potentially subject to a requirement for authorisation. The following uses were identified as potential non-intermediate uses to be further examined:

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[3] We presume that the impregnated tissue referred to could be based on such a fibre, although we have insufficient details to draw definitive conclusions.

[4] Polyimide Boosts High-Temperature Performance (Originating Technology/NASA Contribution), NASA Tech-Briefs, 1 May 2009.

[5] http://spinoff.nasa.gov/Spinoff2008/ip_5.html

- Hardeners in epoxy resins;
- Hardener in adhesives; and
- Use in fibre-tissue PMR-15.

The remainder of the study focuses on these three uses.

4. Data Analysis

4.1 Uses

4.1.1 Overview

Much of the information presented in this section is based on the Entec (2008) study and is supplemented with the information received during stakeholder consultation for the current study.

4.1.2 Hardener in epoxy resins

As indicated in Table 3.1, in 2008 it was estimated that 200t/year of MDA was used in this application. This was based on information provided during consultation from a small number of companies who use oligomeric MDA (containing both 4,4'-MDA and higher molecular weight species, hereby referred to as oMDA) as a component in epoxy curing agents (hardeners). A comparable quantity of MDA was found to be used exclusively in the filament wound pipe application.

Consultation has been conducted with relevant stakeholders to obtain an update on the situation. Please refer to confidential appendix.

No companies have been identified that are currently using 4,4-MDA as a hardener in epoxy resins. The evidence suggests that since 2008 the use of MDA has been phased out.

4.1.3 Hardener in adhesives

In 2008, only one adhesive producer was identified through FEICA as using MDA. Their customers were expected to be in the automotive sector using the hardener in sand forms (moulds) to cast engine parts. According to FEICA other members indicated a phase out of the substance over the preceding 8 to 15 years.

Consultation has been conducted with relevant stakeholders to obtain an update on the situation. FEICA was contacted, who in turn contacted their members. No input was received from any members. Please refer to confidential appendix for further information.

No companies have been identified that are currently using MDA as a hardener in adhesives. The evidence suggests that since 2008 the use of MDA has been phased out. The information obtained from consultation with previous users of MDA suggests that companies have moved away from this substance because (amongst other factors) the manufacturers of the substance have not supported this use in their REACH registration dossiers.

4.1.4 Use in polyimides (PMR-15)

This use was not considered in the 2008 (Entec) report.

One company has been identified as supplying various high temperature resins, including PMR resins, for aerospace applications. The resins are known to be produced in the USA. No information was available on the quantity supplied to EU companies.

Information available for this study suggests that PMR-15 is currently being used in Europe. It is believed that whilst there may be other (unidentified) companies using PMR-15 in Europe, the number of companies using the substance in polyimides is relatively limited. No further evidence of companies using it has been identified.

NASA (1996) estimated that the worldwide market for PMR-15 was in the order of 20t/year. With total sales of around $5-10 million, this suggests a price of $220-440/kg. Since then concerns over MDA have led to the elimination of PMR-15 from engines of certain airline fleets (NASA, 1996). In the absence of further information, it is assumed that up to 20t of MDA is used each year to make PMR-15. This is considered to be an upper limit.

According to the available literature, there are a wide range of PMR resins available. PMR-15 was the first generation of PMR resins to be developed in the mid-1970s. Later, second-generation PMR resin technologies, such as PMR II–50 and VCAP–75 offered improvements in the upper-use temperature (to 700°F) and in the useful life at temperature without major compromises in processing and property retention, but with significant increases in resin cost. Newer versions of MDA-free PMR resins, such as BAX PMR–15, reportedly offer similar advantages as originally found for PMR-15 but also with significant increases in resin cost (Alston and Scheiman, 2000).

DMBZ-15 (based on 2,2'-dumethylbenzidine), was developed as a suitable replacement for PMR-15 (NASA, 1996). Its formula replaces MDA with 2,2'-dimethylbenzidine (DMBZ), and it can endure service temperatures in carbon fiber composites as high as 635°F/335°C (McConnell, 2009). However, consultation with a US polyimide component producer found that this is no longer a commercial product as it was not possible to obtain the diamine needed in the synthesis commercially.

Bismaleimide resins (BMI) are also understood to be feasible alternatives to PMR-15. BMI resins are polyimides used in high-performance structural composites that require superior toughness and high-temperature resistance, such as Formula One cars and fighter aircraft (Stezenberger, n.d.). They have processing characteristics similar to epoxy resins, and are used as laminating resins, prepregs, and adhesives. Epoxy blends of BMI resins can withstand use-temperatures as high as 245°C without a decrease in thermal stability. Consultation with a non-EU supplier found that they have recently developed a a BMI resin, BR-2412, that is comparable to PMR-15. It is expected to have a comparable price to PMR-15 but is not yet commercially available and no information is available on technical performance.

In the USA, the FreeForm-14 product family (such as MVK-14) has been designed specifically to replace PMR-15 polyimide in high-temperature airframe and propulsion applications for military, commercial and general aviation structures requiring 375°F to 475°F wet and 500°F to 550°F dry operating environments. This product is expected to be more expensive than PMR-15. Further information on prices of alternatives is included in the confidential appendix.

4.2 Current and Planned Abatement Measures

Occupational exposure limits exist in a number of EU Member States, typically requiring workplace concentrations, typically requiring compliance with concentrations in the range 0.009 mg/m^3 (The Netherlands) to 0.82 mg/m^3 (Belgium and Spain) as an 8-hour time-weighted-average. Not all Member States have an OEL in place and there is currently no EU-wide OEL.

4,4-MDA is now on the list of substances subject to authorisation in Annex XIV of the REACH Regulation. It is clear that the classification of the substance as a carcinogen and the inclusion of the substance on the Candidate List and then the authorisation list have been a driver for companies to move away from use of the substance.

This is also supported by consultation undertaken for the current study, with a number of companies indicating that they have moved away (or are in the process of moving away) from use of the substance. The fact that key historical non-intermediate uses of the substance were not registered by the main suppliers has been a driver in some companies' reduced use.

As a result, consultation for the current study has not identified any remaining users of 4,4-MDA as a hardener in epoxy resins or adhesives. There are also indications that the costs of applying for authorisation were a driver in discontinuing use in some cases.

Whilst replacement of 4,4-MDA by alternatives such as aliphatic polyamines has been possible in some cases, a number of companies consulted for the current study have indicated that they have been unable to find suitable replacements in all cases, and the associated uses have therefore effectively been 'lost'. The result is presumably that either their customers have found alternative products, or that the end-products (articles) are now produced outside the EU, potentially with the articles being imported into the EU.

There remains some use of oligomeric MDA in hardeners for epoxy resins. This was registered separately from 4,4-MDA as "Formaldehyde, oligomeric reaction products with aniline" (CAS No 25214-70-4, EC No 500-036-1). This substance reportedly contains 47% to <65% 4,4-MDA[6] and was included on the Candidate List on 19 December 2011. An Annex XV report was also produced for this substance, by the German authority[7].

4.3 Possible Future Abatement Measures

The main possible future abatement measure under consideration for the current study is replacement of the substance, as a result of not applying for, or not being granted an authorisation. The only uses of interest for the current study are non-intermediate uses.

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[6] ECHA (2011): Support document for identification of formaldehyde, oligomeric reaction products with aniline as a substance of very high concern because of its CMR properties: (http://echa.europa.eu/documents/10162/13638/svhc_supdoc_technical_mda_en.pdf).

[7] http://echa.europa.eu/documents/10162/b8e4fd2e-54b7-4743-9b31-d858277a1b6a.

As noted above, companies that have previously used the substance appear to have largely moved away from use of the substance. The principal identified use relates to use in polyimides (PMR-15).

Use as a hardener in epoxy resins and/or adhesives has taken place in the past but no current use has been identified in the present study. Remaining uses cannot be ruled out, although it is expected that these would not be covered by the scope of the current registrations for the substance.

Furthermore, another potential means of reducing use of 4,4-MDA is through possible controls on use of ‘oligomeric MDA’, in which substantial concentrations of 4,4-MDA may be contained. As indicated previously, some remaining use of this substance has a hardener has been identified, although the companies in question seem to have significantly reduced (or eliminated) their use of the substance.

4.4 Available Abatement Cost Data for MDA

The available data on abatement potential and abatement costs for 4,4-MDA is limited. For use in polyimides, data on the relative price of using alternative resins is available (see **Error! Reference source not found.**). It appears that there are functionally alternative products available that can be used in place of PMR-15 for some applications, though given the high performance nature of the articles it is used in, it is not likely to be straightforward to simply replace PMR-15 with the lowest-price available alternative. It has not been possible to estimate the quantity of 4,4-MDA used in PMR-15 in the EU.

In addition, some data is available on the quantities of oMDA (containing 4,4-MDA) currently used in hardeners for epoxy resins, as set out in the confidential appendix. No quantitative information has been identified on the costs of replacing MDA in these applications, although it seems that some companies that have moved away from use of the substance have not been able to find suitable alternatives for all of their products. Furthermore, the costs of toxicological testing for some of the potential alternatives (as required for REACH registration) have also been indicated to be a barrier to substitution in some cases.

5. Conclusions

During the course of this study, no companies were identified that are currently using 4,4-MDA as either a hardener in epoxy resins or as a hardener in adhesives. It is suggested that for both of these applications, since 2008 the use of MDA has been phased out. This could be, in part, as a result of the classification of the substance as a carcinogen and the inclusion of the substance on the Candidate List and then the authorisation list.

The principal identified use relates to use in polyimides (PMR-15). Information available for this study suggests that PMR-15 is currently being used in Europe. It is believed that whilst there may be other (unidentified) companies using PMR-15 in Europe, the number of companies using the substance in polyimides is relatively limited. The available data on abatement potential and costs for 4,4-MDA in polyimides is limited. Some data on the relative price of using alternative resins is available. It appears that there are functionally alternative products available that can be used in place of PMR-15 for some applications, though given the high performance nature of the articles it is used in, it is not likely to be straightforward to simply replace PMR-15 with the lowest-price available alternative. It has not been possible to estimate the quantity of 4,4-MDA used in PMR-15 in the EU. As a result, it has not been possible to develop cost curves for substituting MDA for use in polyimides.

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