CBA of Foot and Mouth Disease Control Strategies: Environmental Impacts

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1 Introduction

The Defra report on the economic costs of the 2001 outbreak[2] identified the environmental impacts of carcass disposal as a key area for which costs were not provided. These environmental costs are difficult to estimate because they are not captured through markets.

It is possible to estimate these environmental costs using the environmental economics literature. Indeed, this literature is well developed and accepted, and has widespread use in policy appraisal and cost-benefit analysis in UK appraisal[3].

We have used this literature to assess the environmental impacts and costs of the 2001 outbreak. This provides a guide to the significance of these costs, and allows us to compare them to the other categories identified in the Thompson report.

We have then assessed the direct environmental costs of different control strategies for dealing with outbreaks of foot and mouth disease.

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[2] Thompson et al (2001). Economic Costs of the Foot and Mouth Disease Outbreak in the United Kingdom in 2001. Published by Defra.

[3] For example, with guidance recommended in the Green Book. Appraisal and Evaluation in Central Government. HMT, 2004. http://www.hm-treasury.gov.uk/economic_data_and_tools/greenbook/data_greenbook_index.cfm

2 Impacts of the 2001 outbreak - Scoping Analysis

During the 2001 outbreak, over 4 million animals were culled for disease-control, and another 2 million animals were slaughtered for welfare reasons (Table 1). Around 29% of carcasses from disease-control slaughter were disposed of by burning (including in-situ burning, mass pyres and incineration), the remainder being disposed of by rendering, landfill or burial (see Table 2).

Table 1. Estimated number of animals slaughtered during 2001 outbreak

Animal	Disease control	Welfare	Total
Sheep*	3,487,000	1,587,364	5,074,364
Cattle	582,000	169,033	751,033
Pigs	146,000	286,943	432,943
Goats / Deer / Other	5,000	5,429	10,429
Total	**4,220,000**	**2,048,769**	**6,268,769**

Source: DEFRA[4]
*The figure for sheep does not include up to 3 million lambs at foot.

Table 2. Main disposal methods used during the 2001 outbreak for Disease Control Slaughter

Animal	Burn	Rendering	Landfill	Bury on farm	Other
Pigs	39%	32%	18%	11%	2%
Cattle	41%	35%	16%	6%	1%
Sheep	27%	27%	24%	20%	2%
All	29%	28%	22%	18%	2%
Tonnes		131,000	95,000 commercial landfill and 61,000 mass burial		
Number of sites	Over 950 pyre sites	7 plants	29 commercial and 4 mass burial	900 sites	

Source: NAO[5], Environment Agency[13]

Notes:
1. Excluding slaughter under the welfare scheme and in the three kilometre cull
2. Burn includes incineration, on-farm burning and mass pyres
3. Landfill includes mass burials

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[4] DEFRA (2003), Animal Health and Welfare web pages, Statistics on Foot and Mouth Disease, as of 3 June 2003, www.defra.gov.uk/footand mouth/cases/statistics/generalstats.htm and Livestock Welfare Disposal Scheme statistics, as of 3 March 2002, www.defra.gov.uk/footand mouth/cases/statistics/wdstats.htm

[5] National Audit Office, June 2002, *The 2001 Outbreak of Foot and Mouth Disease,* www.nao.gov.uk

The main potential environmental impacts of the outbreak were:

- Air pollution from burning carcasses on mass pyres, in-situ on farms and in incinerators.
- Contamination of land from deposition of emissions from burning carcasses (see above) and possible health impacts where agricultural produce from this land is consumed.
- Groundwater pollution from disposal of carcasses in landfill sites or mass burial sites, and disposal of ash in landfill sites.
- Spread of disease through release of fluids from carcasses, in smoke from burning, or through action of vermin.
- Surface water pollution from farming activities which were disrupted by the outbreak – mainly pollution from slurry disposal which was affected by movement restrictions.
- Water and land pollution from use and disposal of disinfectant.
- Loss of amenity due to new mass disposal sites – pyres or mass burial sites – which had a significant impact on local communities.
- Impacts on biodiversity arising from changes in livestock numbers and grazing patterns.

2.1 Air pollution costs from carcass burning

In order to assess the air pollution costs from carcass burning, the study has progressed through the following steps:

- Identify environmental burden;
- Quantify physical impacts;
- Value physical impacts in monetary terms using environmental economics literature.

2.1.1 Environmental burdens (emissions)

Carcasses were burnt on mass pyres, on localised sites on farms, and in high temperature incinerators. There is no data available from the outbreak that can be used to split the total number of carcasses burnt between these three methods, although it is reported that few carcasses were actually burnt in incinerators because available incinerator capacity was fully utilised for disposal of BSE-affected cattle carcasses[5]. This section concentrates on the impacts of burning carcasses on mass pyres.

Estimates of emission factors per animal carcass burnt on open pyres are reported in the 15th report of the UK National Atmospheric Emissions Inventory (NAEI)[6]. The text and tables below are taken from the NAEI report.

The burning of carcasses on mass pyres occurred between March and May 2001 and involved specified quantities of fuels, including wooden railway sleepers, kindling wood, straw, coal and diesel oil. Emissions of air pollutants occurred both due to the combustion of these fuels and the burning of the carcasses. Emissions of CO, NOx, SO_2, HCl, PM_{10}, PAH, and PCDD/F were considered to be most significant from pyres and were therefore estimated for inclusion in the NAEI. Estimates of quantities of fuels burnt on pyres were based on the following Government recommendations for fuel use for animal pyres:

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[6] National Atmospheric Emissions Inventory (2003), *UK Emissions of Air Pollutants 1970 to 2001,* www.naei.org.uk/reports.

- 1 railway sleeper (2.8 x 0.3 x 0.2 metres) per cow;
- 25 kg wood kindling per cow;
- 1 bale of straw per cow;
- 203 kg coal per cow;
- 1 gallon (or 3.21 kg) of diesel oil per metre length of pyre;
- 24.6 kg of Feedol 17 (an inorganic chemical accelerant) per tonne of coal

Some additional assumptions were required. The density of a wooden railway sleeper was estimated to be 500 kg/m^3, and the mass of a bale of straw was estimated to be 20 kg. Typical animal masses were taken from official UK agricultural statistics, and were 335 kg, 80 kg and 18.2 kg for a cow, pig and sheep respectively[7]. A 150 m pyre was assumed to contain 250 cattle or 800 pigs or 4600 sheep.

Emission factors for animal pyres do not exist in the literature, so emission factors for domestic combustion sources, straw burning or crematoria were used to compile the inventory. Towards the end of the outbreak, emission factors were calibrated (by using measurements made at actual pyres, and dispersion modelling). Modelling studies conducted by the Environment Agency (EA) suggested good agreement for most pollutants including NO_X, CO, SO_2, HCl, and PM_{10}. For PCDD/Fs and PAHs, studies of pollutant concentrations from the plume at several different pyre sites showed much lower concentrations than expected. Estimates of these emission factors were therefore reduced significantly. Table 3 shows the factors used and gives details of the rationale for their selection.

Table 3. Emission factors for pyres.

	NOx	CO	SO_2	HCl	PM_{10}	Dioxins	B[a]P	Fossil C
	kt/Mt	kt/Mt	kt/Mt	kt/Mt	kt/Mt	G/Mt	kg/Mt	kt/Mt
Coal	1.42[a]	45[a]	20[b]	2.35[a]	40.57		1500[a,l]	659.6
Wood (sleepers)	0.722[c]	99.3[c]	0.037[c]	1.175[d]	7.9[c]		1300[c,l]	
Wood (kindling)	0.722[c]	99.3[c]	0.037[c]	1.175[d]	7.9[c]		1300[c,l]	
Straw	2.32[e]	71.3[f]	0.037[c]		5		7200[e,l]	
Diesel oil	2.16[g]	0.24[g]	2.8[g]	0.01[h]	0.25[g]			857
Carcasses	4.63[i]	142.6[i]	1.4[j]	0.7[j]	10[i]		7200[e,l]	
Combined material						1000[k,l]		

Source: NAEI (2003)
a as for domestic coal combustion
b based on assumed sulphur content of 1%
c as for domestic wood combustion
d assumed to be 50% that of coal
e as for open burning of straw
f as for agricultural use of straw as a fuel (not open burning)
g as for domestic combustion of gas oil
h as for domestic fuel oil combustion
i assumed double that of straw
j assumed as for crematoria on mass basis
k Initial estimate from expert judgement- NAEI & EA (Coleman & Foan *pers. comm.* 2001)
l Correction factors derived from a comparison of modelled and measured deposition during animal pyres. Correction factors are 0.259 for B[a]P and 0.00334 for dioxins.

The estimated emissions from the 2001 outbreak have been estimated and compared to national emissions. Emissions were particularly significant compared with national totals in

[7] Animal weights were taken from IGER/SR. For the calculations later in the document, we have assumed weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep, following recommendations from the review.

the case of PM_{10}, B[a]P and CO. In the first two cases, the pyres contributed a few percent to national totals whereas the figure for CO was about half a percent.

Table 4 Emissions from Foot and Mouth Pyres 2001 Outbreak vs. Annual Emissions

	NOx	CO	SO_2	HCl	PM_{10}	Dioxins	B[a]P
	kt	kt	kt	kt	kt	g	kg
Cattle	0.48	16.78	1.1	0.2	3.05	0.547	302
Sheep	0.1	3.48	0.23	0.04	0.63	0.113	63
Pigs	0.03	0.93	0.06	0.01	0.17	0.03	17
Total	**0.6**	**21.19**	**1.39**	**0.26**	**3.84**	**0.691**	**383**
Other UK sources	**1680**	**3737**	**1125**	**79.7**	**178.5**	**357**	**10090**
% contribution from pyres to UK totals	0.04%	0.57%	0.12%	0.33%	2.15%	0.19%	3.80%

Source: NAEI, 2001.

Some of the uncertainties in these values are discussed in the box below.

Box 1. Uncertainties in Emissions

The emission estimates for animal pyres above are very uncertain. The emission factors used for estimating emissions have been derived for significantly different types of combustion process such as domestic fires, small industrial-scale combustion plant, crematoria, and open burning of straw. None of these types of combustion processes are as large as the animal pyres used for carcass disposal, none will have the same mixture of fuels as the pyres and none will also involve open burning of animal matter. Combustion in the pyres would not be expected to be as efficient as obtained in industrial combustion plant or crematoria and the efficiency achieved could vary significantly with local factors such as construction of the pyres and weather conditions.

Overall, the NAEI estimates that emission factors for toxic organic pollutants used in the emission calculations may be accurate to within a factor of five (i.e. the 'true' emission factor would be between 20% and 500% of the values used). For other pollutants, the accuracy is expected to be better, perhaps within a factor of two.

There are a number of potential air pollution issues which are not addressed by the emission estimates presented above. This is either because they have been assumed to be trivial or because no evidence exists to suggest that they occurred. These issues include:

- Emissions of other air pollutants from the combustion of fuels and animal carcasses including methane, volatile organic compounds (including benzene and 1,3-butadiene), nitrous oxide, ammonia, and metals. Emissions of these pollutants are considered to be trivial compared with other UK sources (however see the following point regarding metals);
- Emissions of arsenic, copper and chromium due to the presence of CCA (copper-chrome-arsenic) wood preservatives in treated wood, or emissions of toxic organic pollutants due to the presence of lindane or PCP in treated wood used in the pyres. It has been confirmed that UK-sourced railway sleepers would have been untreated or treated with creosote only, although any overseas-sourced railway sleepers might have been treated using other preservatives including CCA, lindane or PCP formulations. No data are available on the use of overseas-sourced wood but it has been assumed that none was used. The presence of creosote in wood is perhaps of concern because, since creosote contains polycyclic aromatic hydrocarbons, it has the potential to cause emissions of PAHs and dioxins to air. However, current information suggests that the presence of creosote in wood does not lead to increased emissions of PAHs and dioxins compared with untreated wood. Therefore, we have not attempted to estimate whether any creosote was present in wood used in pyres.
- Emissions of toxic organic pollutants due to the presence of plastics in the pyres. Plastic bags were used to cover the heads and feet of animals which had lesions, however guidance was issued that no PVC should be used. PVC bags would have provided a source of chlorine and might have increased the potential for dioxins to be formed. We have assumed that no PVC was present in the pyres.
- Emissions from Feedol are not included.
- In some cases, tyres and other waste containing plastics were burnt on pyres, against government guidance. This would have significantly increased emissions of dioxins and other pollutants locally.

The emission factors shown in Table 3 were used to calculate emissions per animal carcass burned.

Table 5. Emission factors for mass pyres, per carcass

	NOx	CO	SO_2	PM_{10}	Dioxins	B[a]P	Carbon*
	kg	kg	kg	kg	microg	g	Kg
Cattle	1.97	69.2	4.54	12.5	2.25	1.25	136.6
Sheep	0.107	3.76	0.247	0.682	0.122	0.0677	7.4
Pigs	0.473	16.6	1.09	3.01	0.541	0.299	32.6

Source: NAEI, 2001.

*note carbon from fossil fuel only. Carbon emissions from biomass (carcasses) are not included.

2.1.2 Environmental Impacts and Economic valuation

Air pollution has a number of important impacts on human health, as well as on the natural and man-made environment. These include impacts of short-term and long-term exposure to air pollution on human health, damage to building materials, effects on crops (reduced yield) and impacts on natural and semi-natural ecosystems (both terrestrial and aquatic). The impacts from air pollution on these receptors are described in the box below.

We stress that there is no 'approved' approach for valuing air pollution in UK policy appraisal, and the valuation of air pollution is the subject of a current review[8]. For the current study, we have based the quantification and valuation approach on a recent detailed analysis for Defra, on the quantification and valuation of air quality[9]. Health impacts dominate the air quality valuation, and so additional information on the quantification and valuation of health impacts are summarised in an appendix. In order to apply the analysis here, we have taken unit pollution costs from the Defra study (costs per tonne of pollutant), for specific pollutants, and applied to the emission estimates above.

In applying this approach, it is important to recognise that the location of air emissions is extremely important. This is because the health impacts of air pollution will vary with the local population exposed. Emissions in large, densely populated urban areas have order of magnitude greater impacts, per tonne of emissions released, than emissions in rural areas (see Appendix). Most of the pyres from the foot and mouth outbreak were in rural areas, and this is taken into account in the analysis below.

We also highlight that there is uncertainty with the different aspects of the approach used here, in addition to the emissions analysis. This includes the modelling of pollution concentrations, the health impacts assessed and concentration-response relationships, and the valuation of impacts. It is important to stress that this includes the statistical uncertainty for

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[8] A workshop was held on the' valuation of health benefits of reductions in air pollution and use of values in UK appraisal' on the 21st June 2004. This is likely to lead in time to new recommendations on valuation that would supersede the values used here (though we would not expect them to differ fundamentally to the approach or numbers presented).

9 Watkiss et al, forthcoming. Evaluation of the Air Quality. Study for DEFRA (Environmental Protection and Economics). AEA Technology Environment.

any given impact, but also uncertainty over the exact effects of different pollutants. The approach here does not fully take all this uncertainty into account, and uses a central 'restricted' range, i.e. a low and high, around a central 'best guess' value. The low and high 'restricted' range reflects the uncertainty in the quantification and valuation of key health impacts only.

Box 2. Analysis of Health and Non-Health Benefits of Air Pollution

Studies of air pollution episodes (such as the London smog episodes of the 1950s) have shown that very high levels of ambient air pollution are associated with strong increases in **adverse health effects**. Recent studies also reveal smaller increases in adverse health effects at the current levels of ambient air pollution typically present in urban areas. The health effects associated with short-term (acute) exposure include premature mortality (deaths brought forward), respiratory and cardio-vascular hospital admissions, exacerbation of asthma and other respiratory symptoms. The evidence for these effects is strongest for particles (usually characterized as PM_{10}) and for ozone. For these pollutants the relationships are widely accepted as causal. Recent studies also strongly suggest that long-term (chronic) exposure to particles may also damage health and that these effects (measured through changes in life expectancy) may be substantially greater than the effects of acute exposure described above. These health impacts have major economic costs because of the additional burden they impose on the health service, the lost time at work, and the pain and suffering of affected individuals. The approach adopted here uses concentration-response functions that link given changes in air pollution to health endpoints, which are then valued. Additional details on the approach used are included in the Appendix.

Air pollution also impacts on other receptors. The effects of atmospheric pollutants on **buildings and other materials** provide some of the clearest examples of air pollution damage. Air pollution is associated with a number of impacts including acid corrosion of stone, metals and paints in 'utilitarian' applications; acid impacts on materials of cultural merit (including stone, fine art, etc.); ozone damage to polymeric materials, particularly natural rubbers; and soiling of buildings. SO_2 is the primary pollutant of concern in building corrosion, primarily from dry deposition, but also from the secondary acidic species in the atmosphere. The approach for quantifying and valuing these impacts for 'utilitarian' buildings is based on previous impact pathway analysis in the EC's ExternE Project (1998: 2001), which links the 'stock at risk' of building materials to exposure-response functions. Impacts are monetised using repair and replacement costs, based on critical thickness loss. The key source of data for this part of the assessment is the UNECE ICP on Materials (UNECE ICP Materials (2003). While a similar approach could, in theory, be applied to historic and cultural buildings, there is a lack of data on the stock at risk, and also the relevant valuation of building damage. The analysis of building soiling centres on the deposition of particles on external surfaces and the dis-colouration of stone and other materials. Although soiling damage has an obvious cause and effect, the quantification of soiling damage is not straightforward. The approach here has been to quantify and value urban emissions of particles, based on a simplified approach using cleaning costs, with an upward adjustment for amenity loss. The analysis of ozone damage to materials has not been included in this study.

Ozone is recognised as the most serious regional air pollutant problem for the **agricultural sector** in Europe at the present time. Quantification of the direct impacts of ozone on agricultural yield has used an approach from the EC ExternE project. The valuation of impacts on agricultural production combines estimated yield loss by world market prices as published by the UN's FAO. Some air pollutants other than ozone have been linked in the literature to crop damages (e.g. SO_2, NO_2, NH_3), but generally at higher levels than are currently experienced.

Air pollution also can impact on **natural and semi-natural ecosystems.** The effects of SO_2 and secondary pollutants on ecosystems ranging from forests to freshwaters are well known, and have been the prime concern until recently in international negotiations. Emissions of NO_X are also known to be responsible for a range of impacts on ecosystems particularly through their contribution to acidification, eutrophication and the generation of tropospheric ozone. However, despite the large, well-documented literature available on these effects, it is not currently possible to conduct an economic analysis of the effects of SO_2 and related secondary pollutants (sulphates and acidity), nor euptrophication or ozone effects on forests, other terrestrial ecosystems and freshwaters, with any confidence. A robust economic analysis would require knowledge of specific effects over extended time scales and appropriate models are not available.

The analysis has also included carbon dioxide emissions. Note because animal carcasses are biomass in origin, burning them leads to no net increase in greenhouse gas emissions (GHG). However, there are GHG emissions from fuels added to increase burn efficiency. These have been valued using Defra guidance on the social cost of carbon,[10] which recommended an illustrative central value of £70/tC, with a range of £35 to £140/tC[11].

2.1.3 Air Emissions: Results

The emissions analysis above (from the NAEI) was combined with external cost estimates for air emissions in rural locations to give a cost per carcass. This has then been used to estimate the potential costs of the 2001 outbreak. The analysis assumes that most carcasses were burnt on-farm or in pyres, with very low incinerator use (due to lack of incinerator capacity), but it excludes any potential burning of animals killed for welfare reasons. The results are shown below, for a restricted low and high value, and a central 'best guess' value.

Table 6. Estimated Air Pollution Costs of the 2001 Outbreak from Disease Control Slaughter*

Carcasses burnt from Culling		**Environmental Cost**		
		Low	**Central**	**High**
Cows	242,720	3,899,200	14,368,500	27,754,000
Sheep	925,260	807,400	2,977,000	5,750,900
Pigs	55,770	214,800	792,600	1,530,800
TOTAL	**1223750**	**4,921,400**	**18,138,100**	**35,035,800**

Source: Numbers of carcasses burnt, NAEI, 2003, based on NAO.
*The numbers do not include additional burning of animals culled for welfare reasons (no data is available on these animals).
A number of important caveats are associated with these values:

- There are significant uncertainties in the emission estimates, see box 1 above.
- The numbers only include costs that occur in the UK. All trans-boundary pollution / impacts are excluded.
- Values for NO_X and SO_2 include secondary particulate (PM_{10}) formation (nitrates and sulphates). Values for VOC include ozone formation / effects. Values for NO_X do not include ozone formation / effects.
- The analysis assumes no threshold of effects and implements concentration-response functions linearly.
- The low value assumes £3100 for death brought forward and £31500 per life year lost, with future life years discounted (1.5%). The central value assumes £110000 for death brought forward and £65000 per life year lost, with future life years discounted (1.5%). The high value is double the central value (based on the use of the WHO recommended risk factor for chronic health effects).
- All chronic mortality impacts use original $PM_{2.5}$ functions to PM_{10} pollution metric.
- Values for dioxins and B[a]P are based on US EPA risk factors.
- The numbers exclude several categories of impacts, notably: impacts on ecosystems (acidification, eutrophication, etc), effects of NO_X on ozone formation, impacts on cultural or historic buildings from air pollution, mortality from PM_{10} on children, chronic morbidity health effects from PM_{10}, morbidity and mortality from chronic exposure to ozone, change in visibility (visual range), effects of ozone on materials.
- Environmental costs of air pollution vary according to a variety of environmental factors, including overall levels of pollution, geographic location of emission sources, height of emission source, local and regional population density, meteorology etc. These numbers take these issues into account to a certain degree only.
- The range of estimates is based around a 'restricted' range on certain key health values only. It does not include a range of uncertainty, nor consideration of uncertainty for other aspects of the analysis.

[10] GES (2002). Government Economic Service Working Paper. Estimating the Social Cost of Carbon Emissions. Defra-Treasury Paper. http://www.hm-treasury.gov.uk/documents/taxation_work_and_welfare/taxation_and_the_environment/tax_env_GESWP140.cfm

[11] Note the recommended value of the SCC is the subject of a current review.

- The values for B[a]P and dioxins are not based on primary studies. The value for B[a]P is based on earlier ExternE work looking at the transport sector. The value for dioxins is based on the COWI incinerator study (see main text). The applicability of these values to emissions, especially in relation to the location of the emissions, is a major source of uncertainty.

2.1.4 Deposition of pollutants from pyre smoke on land

The Food Standards Agency surveyed concentrations of pollutants such as dioxins in food produced from grassland close to burning sites. Their study did not find levels of pollutants that could be dangerous to human health[12]. We have therefore not considered deposition to land from pyres in this analysis.

2.2 Burial (Landfill, Mass Sites and On-Farm)

2.2.1 Landfilling and burial of carcasses

Landfilling or burial of carcasses can create high concentrations of organic and other pollutants in (liquid) leachate from the sites for 20 years or more[13]. It has been estimated that around 16m^3 of body fluids are released per thousand adult sheep and 17 m^3 per hundred adult cows within two months. The leachate may contain very high concentrations of ammonium (up to 2,000 mg/litre), have a high chemical oxygen demand (up to 100,000 mg/litre, a hundred times that of raw sewage) and contain potassium (up to 3,000 mg/litre). It may also contain sheep dip chemicals, disinfectant, barbiturates and pathogens. Most degradation will occur within 5 to 10 years but leachate may be released for up to 20 years or more[13]. Carcasses will produce methane emissions on decomposition (methane is a more potent greenhouse gas than carbon dioxide, and so this needs to be taken into account, as it leads to a net increase in GHG emissions). In addition to the potential emissions and leachate, there are additional effects from the disamenity associated with landfill sites. This includes traffic, odour, visual impact, vermin, seagulls, dust and windblown litter.

In 2001, landfilling was only permitted at sites with good leachate collection, treatment and disposal systems and adequate monitoring mechanisms. Gas collection and combustion, and odour and vermin control, were also considered in selecting sites. Twenty nine sites were eventually used. Monitoring of leachate, groundwater and surface water discharges showed some increase in leachate strength and volume, but no other impacts except at one site where remedial action was required, due possibly to damage to the landfill liner during excavation to bury carcasses.

The use of mass burial sites and on-farm burial, which do not have the control systems associated with landfill sites, can lead to much greater environmental problems.

Some problems were experienced for such sites, with necessitated remedial action. For example, at the Eppynt mass burial site in Wales all the carcasses had to be excavated and incinerated when leachate was detected in a monitoring borehole. Some smaller on-farm sites may have to be exhumed following further assessment, mainly where BSE-risk cattle or other unauthorised materials were also buried or where other guidelines were not followed. Continued monitoring by the Environment Agency around landfill and mass burial sites suggests that there have been few long lasting impacts on surface water or groundwater, on

[12] Food Standards Agency

the basis of the evidence gathered so far. However, continued monitoring is necessary to establish whether there will be any further effects in the long term.

The public reported over 200 water pollution incidents from the 2001 outbreak. Of these, three were Category 1 (major damage to the aquatic ecosystem) and 11 were Category 2 (significant damage to the aquatic ecosystem). The three major incidents were caused by slurry and/or disinfectant spills, and killed several thousand fish in each case. Over 300 complaints were received about odour from carcasses awaiting disposal and from excavation of new trenches at landfill sites for carcass disposal. The Environment Agency interim report into the environmental impacts of the foot and mouth disease outbreak[13] states that most impacts recorded so far are short term and localised, although monitoring will continue to assess any longer term impacts around disposal sites.

There are no estimates of the environmental costs of the burial of culled animals from 2001, and the data on the environmental burdens and impacts is less extensive than for air pollution above. This makes it difficult to provide accurate analyses for the outbreak. Nonetheless, it is possible to undertake some analysis. It is possible to estimate the environmental costs of landfill, from estimates in the environmental economics literature. For systems with good gas and leachate control, these mostly arise from amenity impacts. Further details are presented in a later section on landfill as a control option. This can be combined with the estimates of animals sent to landfill (shown in Table 2).

For mass and on-farm burial, the environmental costs will be higher. Firstly there are no leachate or gas collection systems. Secondly, the practices in place at landfill sites to control odour, vermin, etc will not be in place, and so the dis-amenity impacts will be higher. For leachate, we assume a low cost of £0.5 per tonne (see later section on landfill), with an upper estimate an order of magnitude higher than this to reflect the potential risk for uncontrolled sites. We have calculated the cost from methane emissions, as per landfill, but assumed that all methane is released, as there will be no collection systems. Applying these approaches leads to the following estimates for landfill and mass burial of carcasses from 2001. We stress the uncertainty with these values is high (though should be treated only as indicative).

Table 7. Estimated Environmental Costs of Landfill/Mass Burial Disposal in 2001

		Environmental Costs		
Disposal Route	**Tonnes**	**Low**	**Central**	**High**
Landfill	95,000	1,019,700	2,381,300	4,838,700
Mass Burial	61,000	3,304,100	6,904,200	13,826,700
On-farm	~54,000*	2,925,000	6,111,900	12,240,000
TOTAL		**7,248,800**	**15,397,400**	**30,905,400**

* estimate based on data in Table 2 and average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

The numbers do <u>not</u> include additional disposal of animals culled for welfare reasons (no data is available on the route of disposal of these animals).
All values in this table should <u>only</u> be treated as indicative.

[13] Environment Agency (2001), *The environmental impact of the foot and mouth disease outbreak: an interim assessment.* ISBN 1-85-7057856. www.environment-agency.gov.uk.

2.2.2 Ash disposal

For mass pyres and on-site burning, ash from pyres was sprayed with disinfectant. Ash and associated contaminated soil from pyre sites was either buried on-farm, or taken to selected landfill sites for burial. In some cases it was re-incinerated. Around 120,000 tonnes of ash was taken to landfill, mainly to four sites. Analysis of ash samples by the Environment Agency suggested that levels of toxic substances were not a major hazard, but high salt concentrations could cause failure of ammonium and potassium drinking water standards if burial was close to a groundwater source[13]. For this reason, ash from some unauthorised disposal sites has subsequently been removed.

The disposal of ash has a direct cost (the market price for disposal). The NAO report estimated 120,000 tonnes of ash that were sent to landfill had a direct cost of £38 million (equivalent to £317/tonne). There are also the additional environmental costs from the waste at landfill sites, i.e. from any emissions, leachate or disamenity effects (see above). Based on the environmental costs of landfill discussed in a later section, this would indicate an additional cost of £372,000 for ash disposal for the 2001 outbreak.

2.3 Rendering

A significant number of culled animals were rendered in 2001. More details of the process are presented in the later section looking at rendering as a control option. There were 7 plants used in 2001, and some 131,000 tonnes of culled animals were rendered. The environmental costs of rendering are also considered in the later section looking at control options. Multiplying the environmental costs by the 131,00 tonnes rendered in 2001 gives environmental costs of between £0.6 and £5.9 million, with a central estimate of £3.2 million.

2.4 Summary of the Environmental Costs of the 2001 Outbreak

The total environmental costs from air pollution from the 2001 outbreak are estimated at around £20 million for air pollution, £15 million for burial and landfilling, and £3 million for rendering. When totalled, they indicate a cost close to £40 million. It is clear that these costs are significant in relation to many of the cost categories identified in the Thompson report on the economic costs of the 2001 outbreak.

3 Analysis of Different Control Options

The different disease control strategies being considered will lead to different air emissions, waste arising, etc. They will therefore have different environmental impacts (and costs). The study has considered the environmental burdens and the environmental costs of different disease control options, for input into the cost-benefit model. The analysis has considered the difference between the main options:

- Culling (disposal); and
- Vaccination.

Where culling occurs, the EU Animal By-Products Regulation[14] states that foot and mouth carcasses must be disposed of either by incineration or rendering, except where there is:

1. A risk of further spread of disease due to transport of the carcasses; or
2. Where available capacity is insufficient; or
3. For remote areas (designated as only Lundy Island and the Isles of Scilly in the UK).

In these cases, burning or burial on site is permitted (except for BSE-risk carcasses).

These have been used by Government in the current hierarchy for disposal options, which is:

- High temperature incineration;
- Rendering;
- Landfill on approved sites;
- On-farm burning (not a preferred option);

Consistent with the lessons learnt from the 2001 outbreak, the use of mass pyres or burial is no longer permitted as an option. Indeed, any use of farm burial is now no longer supported. However, the use of on-site burning will be allowed in very rural or isolated areas, when it is impractical to ship carcasses to incineration/rendering sites. It is not clear whether the definitions of very rural or isolated areas are as the EU regulations above, or whether it could extend to other areas such as the Scottish Highlands. The environmental costs will vary for each of the four main control options, as well as for vaccination policies. The analysis below assesses the environmental costs per carcass, for each animal type, for these options. We exclude consideration of burial on the basis that it is no longer a permitted option. This output from this analysis provides input to the economic model framework to help in the overall assessment of different control strategies.

As noted above, the location of air emissions is extremely important in assessing health and environmental impacts. In practice, the values will actually vary on a site-by-site basis. For the analysis here, we have allocated specific options to different types of areas, to try to take some of these location factors into account.

The analysis below includes a low and high value, around a central best guess. In each case, the assumptions with the low and high value are stated.

[14] EU Animal By-products regulation 1774/2002.

Option 1 – culling and incineration

Incinerators were rarely used during the 2001 outbreak because available capacity was fully utilised by BSE-risk cattle. However, incineration is the first strategy in the disposal hierarchy for dealing with future outbreaks, although capacity is only sufficient for relatively small outbreaks. DEFRA is currently negotiating agreements with the operators of several large commercial animal-specific incinerators such as those used at pet crematoria[15].

Incinerators burning ***only*** animal carcasses are excluded from the EU Waste Incineration Directive[16] but are controlled by the EU Animal By-products Regulation. This lays down requirements for incinerator and rendering plant operating temperatures and conditions, but does not specify air emission standards, which are only covered by national level regulations. In the UK, incineration plants burning only animal carcasses fall into three categories[17]:

- **Over 1 tonne per hour**: Plants burning only animal carcases and with a throughput over 1 tonne per hour are regulated as an Integrated Pollution Control process by the Environment Agency or by SEPA in Scotland, under Part 1 of the Environmental Protection Act 1990. ***New*** incinerators with capacity over 1 tonne per hour are also covered by **Part A1** of the Pollution Prevention and Control regulations 2000 as amended by the Waste Incineration (England and Wales) regulations 2002.

- **Between 50 kg/hour and 1 tonne per hour**: Plants burning only animal carcases and with capacity between 50kg and 1 tonne per hour are regulated by the local authority in England and Wales or SEPA in Scotland as a Local Air Pollution Control process under Part 1 of the Environmental Protection Act 1990. ***New*** incinerators in this capacity range are also covered by **Part B** of the Pollution Prevention and Control regulations 2000 as amended by the Waste Incineration (England and Wales) regulations 2002. However, forthcoming amendments to this legislation may see animal incinerators with capacity of less than 1 tonne per hour but **more than 10 tonnes per day** regulated by Local Authorities as a **Part A2** process.

- **Less than 50kg per hour** throughput: small scale incinerators on farms are not covered by environmental legislation but may require planning consent.

The incinerators specified for potential use for carcass disposal by DEFRA have capacities between 60 and 400 tonnes per week, and thus would mostly fall into the first category above (plants with capacity over 1 tonnes per hour). These plants are regulated by the Environment Agency (or by SEPA in Scotland) as Part A1 processes. For smaller plants, which are licensed by local authorities, the emission controls and other environmental criteria will be set on an individual plant basis by these authorities, but in practice the emission limits will rarely exceed the minimum standards specified in DEFRA's guidance note.

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[15] DEFRA, Foot and Mouth Disease Contingency Plan, Version 3.0.

[16] Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the Incineration of Waste, Article 2 clause (vii).

[17] DEFRA, 2004, Draft Guidance (as at March 2004) Controls on High Capacity Animal Carcase Incineration Plants – Version 1.2.

The study team have been granted access to the RIO/LASSA list of approved incinerator operators. They include incinerators covering a capacity of 3005 tonnes per week in England (18 operators), 595 tonnes per week in Wales (4 operators), and 805 tonnes per week in Scotland (5 operators), totalling a total capacity 4405 tonnes per week. The study team has mapped the location of all operators, to investigate the location of these incinerators. Almost all of these incinerators are located in rural areas, i.e. they are either isolated or in extremely small built-up areas, and are some distance from major agglomerations. There are a few operators located just out of built-up areas (a couple of miles), and 3 operators located within built-up areas (i.e. within towns, albeit usually towards the edge of the urban area). We stress that this is in contrast to most municipal waste incinerators, which are predominately located near urban areas (to reduce transport distances).

The main impacts from incineration are:

- Greenhouse gas and air pollutant emissions from fuel used in combustion;
- Air emissions from incineration;
- Disamenity impacts of the incineration plant (traffic, visual impact, odour, stress).
- Environmental costs of landfilling of ash. Note ash from incinerators is a controlled waste and must be sent to a landfill site authorised to accept carcass ash.

Recent studies have indicated that the majority of the external costs from plants incinerating MSW are associated with air pollution and dis-amenity, with other costs being relatively insignificant[21, 22, 23]. There are no primary estimates of the dis-amenity cost of MSW incineration plants in the UK. Those that exist vary widely and are based on poor data, being generally extrapolated from landfill dis-amenity costs. The 2000 COWI study[21] estimated costs of 8 euros per tonne of waste (based on landfill costs); the ECOTEC study estimated between 31 and 219 euros per tonne of waste (based on landfill costs but adjusting for increased population density near urban MSW incinerators compared to rural landfill sites) [22] and the DEFRA research study recently published estimates of £21 per tonne of waste (but cautioned that this is not applicable to the UK)[23].

This study is concerned with disposal of foot and mouth carcasses in incinerators normally used for animal waste. It is probably fair to say that the impact of the incinerators would normally be considerably lower than that of a standard MSW incinerator or landfill site, which would be much larger and more "visible", and a MSW site is likely to be in an urban area (in contrast to many small–scale animal waste incinerators which are predominantly in rural sites). However, in the absence of better information, we have used the Defra study value of £21 per tonne of waste for this analysis. This is indicative of a population density of 500 inhabitants per square miles in the vicinity of the incinerator. As this is a probable over-estimate, we have considered a lower bound of zero. This gives the following values.

Table 8: Estimated external costs from amenity of incineration plant, per carcass

Incineration – Amenity		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		0	5.2	10.4
Sheep		0	0.5	1.0
Pigs		0	1.0	2.1

Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

Air pollution emission factors for incineration of animal waste have been assumed to be the same as emissions from MSW incinerators listed in the NAEI (in the absence of better data). These are presented in the table below. The exceptions were for SO_2, where the NAEI figure for human crematoria was used (as the sulphur content would be more similar to that from animal waste) and dioxins, where estimates from a COWI study were used[21]. We have cross-checked the estimates against actual emissions reported to the Environment Agency by four of the largest animal waste incinerators listed to accept foot and mouth carcasses, and the estimates seem reasonable in comparison. We have included an estimate of carbon emissions from fuel used in the incinerator, though this is only indicative. Current estimates for fuel use obtained from Wessex Incineration seem very low but this may be because an efficient incinerator can be largely self-sustaining.

Table 9 Estimated emission factors for combustion of animal waste in incinerators

	NOx	CO	SO_2	PM_{10}	Dioxins	C
	kg/t	kg/t	kg/t	kg/t	microg/t	kg/t
Animal waste	1.8	0.709	0.68	0.3	8.0	0.55
	kg/carcass	kg/carcass	kg/carcass	kg/carcass	microg/carcass	kg/carcass
Cattle	0.90	0.35	0.34	0.15	4.01	0.27
Sheep/goats	0.09	0.04	0.03	0.02	0.40	0.03
Pigs/deer	0.18	0.07	0.07	0.03	0.80	0.05

There are no specific emissions factors for carcass incineration. The values above are therefore only indicative.
Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

The emissions have been combined with our air pollution costs to estimate the environmental costs of incinerating carcasses. Remembering that the location of incinerator plants is important, we have assumed incinerators are in rural areas. This gives the following values.

Table 10. External environmental costs of air pollution from incineration, per carcass

Incineration – Air emissions		Environmental Cost (£) per carcass		
		Low	Central	High
Cows		0.56	2.83	5.47
Sheep		0.06	0.28	0.55
Pigs		0.11	0.57	1.09

See caveats listed under table 6. Note a rural location has been assumed for all emissions, which lowers the unit pollution costs significantly of some pollutants (see Appendix).
Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

Finally, there would be some additional environmental costs from the disposal of incinerator ash, which will go to landfill. Given the low levels of ash generated, and the lower costs of amenity impacts from landfilling (see later sections) we believe the additional costs from ash disposal would be insignificant to the above values. Note, there is also a direct market cost for ash disposal to landfill – this is included in the main analysis, but would be significant.

Option 2 – culling and rendering

There are 26 rendering plants in the UK. Seven of these were used in the 2001 outbreak. Rendering involves crushing and grinding of carcasses, followed by heat treatment in a sealed vessel to reduce the moisture content and to kill micro-organisms. Around 60% of the carcass weight is moisture, and this is lost as steam. Steam and gaseous emissions are collected, condensed and the condensate is either bio-filtered or incinerated in the boiler. A correctly controlled process therefore produces few direct emissions to air. Residue from the condensate treatment or incineration will be disposed of to landfill.

There will be potential emissions of carbon dioxide and air pollutants from the fuel used for heating. In the UK, local authorities individually license animal rendering plants, and these authorities will set any emission controls and other environmental criteria. However, if tallow is burnt as a fuel, this is considered to be waste and the plant will therefore have to comply with the Waste Incineration Directive.

The remaining 40% of the carcass weight consists of fat (15%) and protein (25%). Separation of the melted fat (tallow) from the solid protein is achieved through centrifuging (spinning) and pressing. The solid fraction is then ground into a powder, such as meat and bone meal[18]. Meat and bone meal from edible materials can be used for petfood or fertiliser (use as animal feed has been banned since 1996 because of the risk of transmission of BSE). However, meal from foot and mouth culled carcasses must be incinerated – there are currently three approved incineration plants in the UK[19]. Tallow, depending on quality, can be used for human food, animal feed, soap, cosmetics, pharmaceuticals, tyres, paints and other products. Tallow from BSE-risk cattle and foot and mouth carcasses can only be used as a fuel oil, often as a fuel for the rendering process, or incinerated at an approved plant.

In summary, the main environmental impacts from rendering would be:

- Greenhouse gas and air pollutant emissions from fuel used in combustion.
- Air emissions from incineration of meat and bone meal.
- Dis-amenity (traffic, odour, visual impact) of the rendering plant.
- Potential disposal costs and impacts from landfill of residues from condensate and gas treatment (but quantities expected to be small).

The first of these is potentially the most important. Information on specific energy use in rendering plants is collected through the Climate Change Levy (CCL) scheme. Energy use in 2002 was 853 kWh per tonne of waste rendered (primary energy), which was equivalent to 25.5 kg carbon per tonne of waste rendered. However, this includes carbon from burning of tallow, which is not considered to be a greenhouse gas as it is derived from a biological source. We obtained figures from Enviros, who were responsible for performing the evaluation of carbon emissions from the rendering sector, which showed that 21% of the carbon emissions derived from fossil fuels, the remainder being from tallow. The actual carbon emission factor excluding tallow is therefore 5.4 kg carbon per tonne of material rendered. The equivalent external costs for each animal type are shown in the table below. The low, best estimate and high costs are determined by the recommended range for the social cost of carbon (see earlier section). This is different to the full statistical range.

[18] UK Renderers Association website, www.ukra.co.uk

[19] Gordon Hickman, DEFRA, personal communication April 2004.

Table 11. Estimated external costs of carbon dioxide emissions from fuels used for rendering (carbon emission only).

Rendering – fuel used		Environmental Cost (£) per carcass		
		Low	Central	High
Cows		0.094	0.189	0.377
Sheep		0.009	0.019	0.038
Pigs		0.019	0.038	0.075

Of course, this only includes the carbon from the fuel. There are also other air pollutants. This includes air pollutants from the fuel used and from the burning of tallow as a fuel oil. We have estimated the potential emissions based on fuel oil emission factors from the NAEI, based on the fuel use data above. For the analysis, we have assumed that rendering plants are in rural locations. We highlight this as a major uncertainty.

Table 12. Estimated external costs of air pollution from fuels used for rendering, assuming use of tallow as fuel oil.

Rendering – fuel used		Environmental Cost (£) per carcass		
		Low	Central	High
Cows		0.43	2.04	4.08
Sheep		0.04	0.20	0.41
Pigs		0.09	0.41	0.82

The emissions factors used are factored on carbon use above, as there is no direct data. We have no data on emissions from tallow burning. The values above are therefore only indicative. Assumes rendering plant are in rural areas. See caveats listed under table 6.
Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

There are also emissions from the incineration of meal from foot and mouth culled carcasses (note this must be incinerated, it cannot be landfilled). On average, per tonne of carcass, around 23% goes to make protein meal[20]. Using this figure, it is possible to estimate the tonnes of protein meal per carcass. This has been used to estimate the relevant environmental costs. However, we assume that this protein meal would be incinerated in typical urban incinerators. This leads to the following values.

Table 13. Estimated external costs of incineration of protein meal from rendering.

Incineration of protein meal from rendering		Environmental Cost (£) per carcass		
		Low	Central	High
Cows	0.11 tonne meal/t	1.9	9.4	16.9
Sheep	0.01 tonne meal/t	0.2	0.8	1.5
Pigs	0.02 tonne meal/t	0.4	1.7	3.1

There are no specific emissions factors for protein meal incineration. The values are therefore only indicative. Note for disposal, we have assumed the use of typical urban incinerators, rather than specific animal disposal incinerators as for direct incineration above. See caveats listed under table 6. Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep. Assumes 23% of carcass to protein meal.

[20] Information from the UK Rendering Association indicates that the 1.75 million tonnes rendered in the UK each year, produces 250,000 tonnes of fat and 400,000 tonnes of protein meal.

There will be some additional effects from the landfilling of ash from incinerating protein meal (and associated amenity cost) from incineration. We believe the environmental costs from this will be minimal compared to the values above.

Finally, there may well be some dis-amenity costs from rendering plant itself. We have no data on the location of these plants, or the possible amenity impacts from them. Our initial assumption is that odour might be important from such plants. In the absence of any other data, we have assumed the amenity costs, per tonne of carcass, are the same as for incineration plants above (i.e. a lower bound of zero, and an upper of £21 per tonne).

Option 3 – culling and landfilling

Foot and mouth carcasses may be disposed of at "appropriately engineered" landfills, i.e. those with suitable leachate and gas collection and monitoring systems. During the 2001 outbreak, 111 suitable sites were identified in England and Wales of which 29 were eventually used. There are currently no plans to require any pre-treatment of carcasses prior to disposal at landfill sites, although this may change depending on future legislation under the EU animal by-products directive and landfill directive.

Assuming that well-engineered sites are chosen, with active landfill gas and leachate collection systems, environmental impacts will be relatively low. There will be some leakage of methane as no landfill gas collection system achieves more than around 80-90% collection efficiency. Other impacts should be negligible, if groundwater and surface water nearby is continually monitored to reduce the risk of undetected leaks, and if any remedial action required is undertaken promptly. There will inevitably be some additional public nuisance arising from extra lorry journeys to the site, odour problems and stress or worry, though this is minimised by choosing sites away from human habitation.

Direct financial costs will be incurred from landfilling operations. A landfill gate fee will be paid, which will be negotiated individually with the landfill operator but will probably be greater than the price paid to dispose of MSW. During the 2001 outbreak, the average price for disposal to landfill was £225 per tonne, although this included the costs of collection from the farm, cleansing and transport which probably inflated the figure by a factor of three to four[19]. The landfill component of the total disposal costs may therefore have been of the order of £55 to £80 per tonne on average. The gate fees are higher than those paid for MSW because the operator will incur additional costs related to leachate collection and clean-up, and possibly for odour and vermin control. The high prices paid during the 2001 outbreak also reflect the difficulty of finding disposal routes at short notice and in a crisis situation, as well as the intense public opposition to disposal of foot and mouth carcasses in their "back yard". There may also be costs for additional monitoring of nearby groundwater and surface water, and possibly further costs if any remedial action is required (such as repair of the landfill liner, which was necessary in one case during the 2001 outbreak). These costs would all be covered within the direct economic cost analysis of the model. We do not consider them further below, and only assess the environmental costs.

The main impacts from landfilling option would be:

- Methane emissions.
- Air emissions from combustion of landfill gas.
- Potential impacts from groundwater pollution

- Disamenity from traffic, odour, visual impact, vermin, seagulls, dust, windblown litter, from the landfill site

For methane emissions**,** we have estimated potential arisings using the following assumptions:

- Animal carcasses contain around 15% degradable organic carbon (DOC) – this was based on estimated of the DOC content of food waste, although food waste will typically contain a large proportion of vegetable matter as well as animal matter.
- Only two thirds of the DOC will actually degrade in the landfill site over a period of 100 years, giving an estimate of dissimilable DOC (DDOC) of 10% of the carcase weight.
- The carcase will degrade to form landfill gas which is 50% methane and 50% carbon dioxide. As this carbon dioxide originates from a biological source it is assumed to have no net greenhouse gas impact.
- 80% of this methane will be collected and combusted to form carbon dioxide. As this carbon dioxide originates from a biological source it is assumed to have no net greenhouse gas impact.
- At this stage of the analysis we have not made any allowance for carbon savings from displaced fossil fuels where landfill gas is used for energy recovery.
- We have also not made any allowance for methane generated from that portion of the animal which would ultimately have been disposed of to landfill anyway.

Air emissions from combustion of landfill gas have not been specifically evaluated during this phase of the study as the overall impact is likely to be small compared to other impacts (deduced from the results of similar studies such as COWI (2000)[21]).

Groundwater impacts should be small if the site has a well controlled leachate collection and treatment system. However, disposal of significant numbers of foot and mouth carcasses will give rise to additional leachate collection, treatment and monitoring costs. Recent studies have concluded that it is not currently possible to quantify the external costs of leachate pollution from landfill sites in the UK, due to a lack of relevant data[21,22,23]. However, the COWI study recommended that if it was essential to use a value, a figure which is equivalent to around £0.5 per tonne of waste could be used. This is based on the mean of two studies: the CSERGE study of 1993 which is based on clean-up costs, and a 1997 study based on a marginal damage cost approach. Whilst this is stated to be a highly uncertain estimate, it does illustrate that the order of magnitude of costs arising from leachate pollution is likely to be of less significance than the costs arising from methane emissions and disamenity (see below). However, it is important to bear in mind that while the external costs during the operating lifetime of a high quality landfill are likely to be low, there could well be more significant costs in the long term if the landfill liner leaks after the landfill site is closed.

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[21] COWI, 2000, *Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste,* report to the European Commissions, DG Environment.

[22] Eunomia and associates on behalf of ECOTEC, 2002, *Economic Analysis of Options for Managing Biodegradable Municipal Waste,* report for the European Commission.

[23] *Valuation of the external costs and benefits to health and the environment of waste management options,* draft report for DEFRA, 2004.

Disamenity impacts have been estimated in several recent studies[21,22,23]. A recent study for DEFRA based on a hedonic pricing approach recommends a value of £3.1 per tonne of waste landfilled[23]. However, other estimates from European studies are higher, equating to around £7 per tonne of waste[22]. We have used the Defra values here as the central value, and the European values for an upper estimate. However, it is possible that there might be greater impacts from the disposal of foot and mouth carcasses in relation to odour. We also highlight that in the event of large-scale disposal to landfill, there might be significant public opposition (locally).

The resulting external cost estimates for methane emissions and dis-amenity are shown in Table 14 below.

Table 14: Estimated costs of disposal of foot and mouth carcasses to landfill sites

Methane from landfilling		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		5.4	10.7	21.5
Sheep		0.5	1.1	2.1
Pigs		1.1	2.1	4.3
Dis-amenity from landfilling		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		0	1.6	3.5
Sheep		0	0.2	0.35
Pigs		0	0.3	0.7
Leachate from landfilling		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		0	0.25	0.5
Sheep		0	0.03	0.05
Pigs		0	0.05	0.1

Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

Note, there is the possibility of some double counting in that landfill gate price includes the landfill tax, which is set on the basis (partly) of environmental externalities.

Note a simplistic approach has been taken to value methane emissions. In reality, methane emissions would arise over a time profile consistent with the breakdown of the animal carcass (e.g. over the next 20+ years). In practice, we should discount these future costs, but we do not have enough information to fit an accurate time profile to these emissions. However, this effect is mitigated by the recommended social cost of carbon, which is increases at £1/tC per year, to reflect the increasing social costs in future years.

We stress that the above values do not include landfill gate prices. These would be likely to be greater than the costs above (e.g. £55 to £210 per tonne).

Option 4 – culling and localised burning

Emissions and other impacts from localised on-farm burning of carcasses are likely to resemble those of mass pyres (see Section 2). We therefore recommend the use of the earlier values for this option. The values are presented below, adjusted for carcass weights to be consistent with other options. It can be seen that the values are much higher than any of the other control options considered. The values are based on rural emissions. It is possible than impacts might actually be lower for these remote sites, because the population density will actually be potentially lower than for the UK rural average.

Table 15: Estimated costs of localised burning of foot and mouth carcasses

Localised burning		Environmental Cost (£) per carcass		
		Low	Central	High
Cows		24.0	88.4	170.7
Sheep		2.4	8.8	17.1
Pigs		4.8	17.8	34.3

See caveats listed under table 6.
Assuming average carcass weights of 500 kg for cattle, 100 kg for pigs, and 50 kg for sheep.

There might be some additional environmental impacts from localised burning. This would include the disposal of the ash from burning. The exact costs would depend on the disposal method. In 2001, ash from pyres was usually sprayed with disinfectant. The ash and associated contaminated soil from pyre sites was either buried on-farm, taken to selected landfill sites for burial, or re-incinerated. All of these options would lead to some additional direct and indirect costs.

The direct costs are likely to be most significant (e.g. the cost of ash that were sent to landfill in 2001 had an equivalent direct cost of £317/tonne). These will be considered in the main economic analysis.

Culling and on-site burial

We stress that the potential environmental costs from on-site burial are not considered, as this option is no longer included in the permitted disposal hierarchy. We highlight that should such a disposal option be allowed, the environmental costs would be much higher than with landfill disposal above.

Vaccination Policies

The environmental impacts of vaccinate-to-live are likely to be minimal. We therefore recommend a value of zero be assigned as an environmental cost for this option.

The impacts of vaccinate-to-kill policy, i.e. where animals are vaccinated to ease the burden on disposal, has not been considered separately here. This policy will ultimately lead to the same environmental costs as with the disposal options assessed above, merely with a small delay in timing.

4 Other Issues Identified

A number of issues were highlighted during the course of the study, and address comments raised from the economic peer review.

Methane emissions from animals

If animals are killed earlier than would otherwise have occurred, there might be lower methane emissions (methane is a powerful greenhouse gas).

Methane is produced in herbivores as a by-product of enteric fermentation, a digestive process by which carbohydrates are broken down by micro-organisms. Methane is also produced from the decomposition of manure under anaerobic conditions. When manure is stored or treated as a liquid in a lagoon, pond or tank it tends to decompose anaerobically and produce a significant quantity of methane. When manure is handled as a solid or when it is deposited on pastures, it tends to decompose aerobically and little or no methane is produced.

We have investigated the potential importance of this effect, based on methane emissions from livestock (from the NAEI emission factors) and applying the Government social cost of carbon. The results are shown below. They indicate that this effect might have important benefits, i.e. from culling reducing greenhouse gas emissions. Likewise, vaccination might increase the methane emissions from the overall UK head of animals.

However, the study team are extremely hesitant about including these values in the economic model. This is because of the issues with imports – if animals in the UK are culled, then other meat will be imported. Following product life-cycle guidance on boundary analysis, for a unit of useful product delivered (e.g. a kilogramme of beef), we would need to add the greenhouse gas emissions from the lifetime of the imported animal prior to export, the transportation emissions with the exported product from overseas, and the greenhouse gas emissions from the culled animal, i.e. prior to culling and disposal. This would be likely to lead to an increase in greenhouse gas emissions, i.e. an impact. We believe considering this level of detail would complicate the model unnecessarily. We therefore propose to exclude this issue from the analysis.

Table 16: Methane Emission Factors and GHG Externality for Livestock Emissions

	Enteric Methane kg CH_4/head/year	Methane from Wastes kg CH_4/head/year	Environmental Cost from methane emissions £/head/year
Dairy Breeding Herd	115	13.0	51.3
Beef Herd	48	2.74	20.3
Cattle: Others>1, Dairy Heiffers	48	6	21.6
Cattle: Others<1	32.8	2.96	14.3
Pigs	1.5	3	1.8
Breeding Sheep	8	0.19	3.3
Other Sheep	8	0.19	3.3
Lambs < 1year	3.2	0.076	1.3

Source: methane emissions from the NAEI, 2003.

Grazing

In some upland hill areas, there is a benefit in maintaining the current environment due to animal grazing. Without animals, i.e. with a disease outbreak, there may be some loss of the countryside environment. This could be protected from vaccination policy. It has not been possible to assess the potential biodiversity/visual amenity consequences from this effect, though relative to other categories we believe the benefits would be low.

The over thirty months scheme

The OTM Rule bans meat from most cattle aged over 30 months at slaughter from being sold for human consumption. This is to remove older animals, which are more likely to have developed a significant amount of BSE agent in any tissue, from the human food chain. It applies equally to home-produced and imported meat. At present, the current route for disposal of OTM animals is believed to be incineration.

There is a question on the size of the additional (marginal) environmental dis-benefit from disease control, given some animals would be incinerated anyway under the OTM scheme. However, the Food Standards Agency has recently completed a review of the OTM rule and have advised Ministers that it would be acceptable on health grounds to replace the rule with BSE testing of OTM cattle born after July 1996. Defra, partner organisations and key industry stakeholders are currently putting in place arrangements to facilitate the high-volume, high speed testing which will be needed to implement any change. For this reason, we have not considered this issue further.

Potential effects on deer

Foot and mouth can potentially affect other species, notably deer and goats. For any animals that are disposed of, the above values can be used after adjusting for carcass weight. It is our understanding that effects on these species is not considered in the economic model, and so we have not considered further.

5 Summary Numbers for Model

The values above are brought together, by option, below.

Table 17: Estimated external costs from disposal of foot and mouth carcasses

1) Incineration		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		0.6	8.0	15.9
Sheep		0.1	0.8	1.6
Pigs		0.1	1.6	3.2
2) Rendering		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		2.4	12.9	24.0
Sheep		0.2	1.2	2.2
Pigs		0.5	2.4	4.5
3) Landfilling		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		5.4	12.5	25.5
Sheep		0.5	1.3	2.5
Pigs		1.1	2.5	5.1
4) On-farm burning		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		24.0	88.4	170.7
Sheep		2.4	8.8	17.1
Pigs		4.8	17.8	34.3
5) Vaccination to live		**Environmental Cost (£) per carcass**		
		Low	**Central**	**High**
Cows		0	0	0
Sheep		0	0	0
Pigs		0	0	0

See main text and earlier tables for caveats.

Summary of Process and Environmental Impacts of Different Control Strategies.

	Rendering	Landfill	Mass burial	On-farm burial	Incineration	Mass pyres	On-farm burning
Process	Carcasses minced Heat and sometimes pressure treatment Produces blood and bone meal, tallow and condensate. Tallow may be used for fuel. Blood and bone meal incinerated, ash to landfill. Condensate treated or incinerated; ash to landfill.	No pre-treatment Approved landfill sites selected to have acceptable leachate collection, treatment, and monitoring systems; gas collection and combustion systems and odour/vermin control.	No pre-treatment Pits may or may not have liners. Selected to be on clay soils away from human habitation. No gas collection. Leachate may be collected and removed for treatment and disposal. No longer permitted	No pre-treatment Sites assessed briefly by Environment Agency taking account of groundwater levels and protection zones. No longer permitted	No pre-treatment Incineration in plants designed for animals – e.g. pet crematoria. Appropriate emission controls. Ash to landfill.	No pre-treatment No emission control. Carcasses burnt with specified amounts of fuel: wood, straw, diesel oil. Unauthorised waste co-burnt at some sites (e.g. tyres, plastic sheeting). Ash sprayed with disinfectant and buried on site, taken to landfill or re-incinerated. Some unauthorised disposal. No longer permitted	No pre-treatment No emission control. Ash buried on farm or to landfill. Some unauthorised disposal. Permitted only in exceptional circumstances
Impacts	Fuel use (but tallow may be re-used) Emissions from incineration of blood and bone meal and condensate. Landfill impacts Amenity from rendering plant	High levels of organic and other pollutants in leachate Risk of leaks into groundwater or surface water Additional methane generation – GHG impact – reduced through gas collection. Odour and nuisance	High levels of organic and other pollutants in leachate Risk of leaks into groundwater or surface water Methane generation – GHG impact Odour and nuisance	High levels of organic and other pollutants in leachate Risk of leaks into groundwater or surface water Methane generation – GHG impact Odour and nuisance	Direct emissions to air reduced through emission control. Impacts of ash disposal in landfill. Amenity impact of plant	Direct emissions to air. Impacts of ash disposal (see landfill).	Direct emissions to air Impacts of ash disposal (See landfill)

Glossary

1,3 butadiene. A potentially carcinogen.

Benzene. A carcinogen

B[a]P. benzo[a]pyrene. A potentially carcinogen.

CO. Carbon monoxide. An air pollutant primarily associated with transport.

CO_2. Carbon dioxide. A greenhouse gas.

COMEAP. UK Department of Health's Committee on the Medical Effects of Air Pollutants.

DEFRA. Department for Environment, Food and Rural Affairs.

EAHEAP. Economic Appraisal of the Health Effects of Air Pollution, Department of Health Ad-Hoc Group on the Economic Appraisal of the Health Effects of Air Pollution

EC. European Commission.

IGCB. Interdepartmental Group on Costs and Benefits.

IOM, generally referring in the text to the work by the Institute of Occupational Medicine for the Scottish Executive, which used a higher exposure-response function for the analysis of chronic mortality impacts from air pollution.

NO. Nitric oxide.

NOx. Oxides of nitrogen (includes NO and NO_2). Precursor species for ozone.

NO_2. Nitrogen dioxide.

PM_{10}. Particulate matter less than 10µm aerodynamic diameter.

SO_2. Sulphur dioxide.

VOC. Volatile Organic Compounds. Precursor species for ozone.

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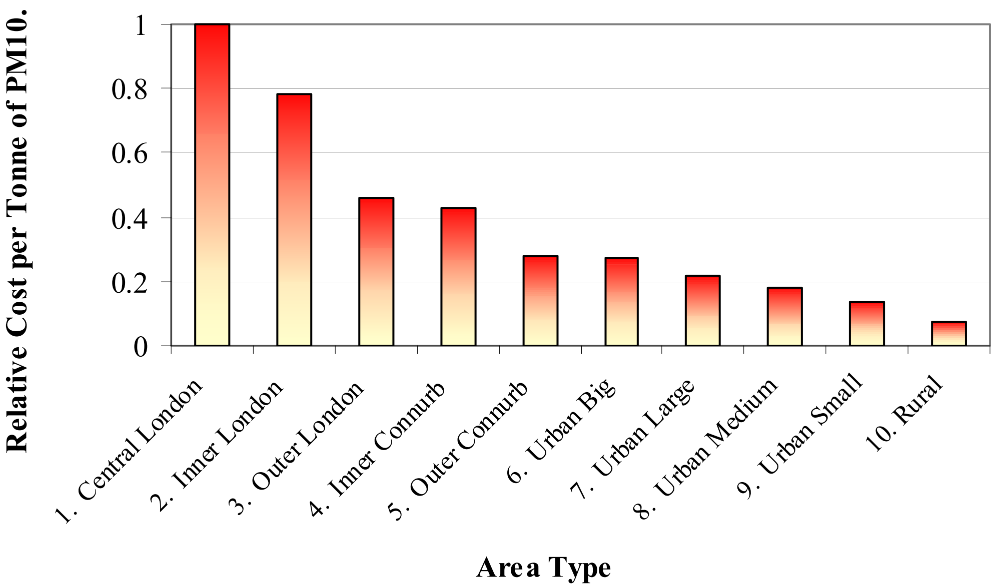
Appendix

Box A1 below provides the main background on the quantification of the health impacts of air pollutants used in the study. The main health outcomes quantified in the study are:

- Short-term (acute) pollution effects - deaths brought forward and respiratory hospital admissions; and
- Long-term (chronic) effects - changes in life expectancy, known as chronic mortality.

The approach to valuation of the main health impacts is provided in Box A2.

The location of air emissions is extremely important in the health impacts, because of the local population density exposed to pollution. Emissions in large, densely populated urban areas have order of magnitude higher impacts, per unit tonne of emissions, than rural areas. This can have a major impact on the results, as shown in the figure below.



Source: Watkiss et al, forthcoming. Air Quality Evaluation.

Box A1. Quantification of Health Effects

Two types of epidemiological study are relevant to the quantification of mortality impacts from health pollution:

- Time series studies, available for assessing the mortality and morbidity impacts of the short-term (acute effects) exposures to PM, SO_2, O_3 etc., which examine associations between daily pollution levels and daily numbers of deaths or respiratory hospital admissions.
- Cohort studies which examine age-specific death rates (technically mortality hazards) in study groups of individuals followed up over prolonged periods. Having adjusted for other mortality risk factors measured for individuals (gender, race, smoking habit, educational status, etc.), differences in age-specific death rates between cities are assessed in relation to average pollution concentrations over periods of several years (chronic effects).

We have based the quantification of health effects on reports of the UK Department of Health's *Committee on the Medical Effects of Air Pollutants (COMEAP 1998: 2001)*). These recommend quantification of deaths brought forward, respiratory hospital admissions, and chronic mortality for particulates (including secondary particulates) and deaths brought forward and respiratory hospital admissions for ozone and SO_2. The IGCB approach (which we adopt here) treats the mortality effects from short-term and long-term exposure as additive. For deaths brought forward and respiratory hospital admissions we have quantified health impacts using the functions from time series studies recommended by COMEAP (1998). For particulates, this uses PM_{10} concentration-response functions. Functions have been implemented linearly, without threshold, consistent with COMEAP and IGCB guidance.

Pollutant	Impact Category	% change in rate per µg/m^3
PM_{10}	Deaths brought forward	0.075%
SO_2	Deaths brought forward	0.060%
Ozone	Deaths brought forward	0.060%
PM_{10}	Respiratory Hospital Admissions	0.080%
SO_2	Respiratory Hospital Admissions	0.050%
Ozone	Respiratory Hospital Admissions	0.070%
NO_2	*Sensitivity Only*	
	Respiratory Hospital Admissions	*0.050%*

For mortality and long-term exposure to PM, the risk estimates are based on analyses of the American Cancer Society (ACS) cohort by Pope *et al* (1995,) and updated in 2002. The main results give a lower bound estimate of increase in death rates of 0.3%/µgm^{-3} $PM_{2.5}$. The IGCB (2001) used a lower risk estimate (0.1% per µgm^{-3} $PM_{2.5}$), based on the preferred estimate selected by COMEAP (2001) from the HEI reanalysis estimates adjusted for further possible confounders. This is a third of the lower bound risk estimate derived by Pope *et al,* (1995, 2002). We have applied the 0.1% risk estimate here, and referred to it as the ***low*** analysis. However, we have also applied the original lower bound risk estimate from the Pope study (0.3%/µgm^{-3} $PM_{2.5}$), as used by the ExternE project in European cost-benefit analysis (EC 1998: 2001) and by the Institute of Occupational Medicine (2003). Where used, this is referred to as the ***central*** analysis. There is therefore a factor of 3 between the risk factors applied. Note the central high factor is still within the sensitivity range recommended by COMEAP. Use of the central high value is supported by Pope *et al* (2002). Indeed, recent European work by WHO under the CAFE project (Clean Air for Europe) and Long-Range Transboundary Air Pollution is now recommending 0.6%/µgm^{-3} $PM_{2.5}$, based on Pope *et al* (2002), i.e. double the central high estimate used here. This is referred to as our ***high*** estimate.

Following IGCB (2001), we have applied the risk estimates for $PM_{2.5}$ directly to the PM_{10} concentrations assessed here. For implementation, we have used a life-table approach to quantify the change in life years. In order to examine the effect of individual policies in individual years, we have had to use a different approach to the IGCB analysis (2001). The IGCB analysis assessed the benefits of achieving a given air quality objective, and assumed this level was maintained thereafter (e.g. looking at benefits in the population through to 2110). This is not appropriate for assessing marginal changes from specific policies, against a changing background of air quality concentration. Therefore we have used an approach based on the IOM's work, which assesses the net benefits of incremental pollution emissions for a single year and follows the effects of the single-year increment on death rates and then on life expectancy through the population over time. We have also undertaken some sensitivity analysis, using additional endpoints not recommended for quantification by COMEAP, based on functions that have been used in European cost-benefit studies (e.g. in ExternE) for the pollutants benzene, 1,3-butadiene and benzo[a]pryene.

Box A2. Valuation of Health Effects

The valuation of time-series health endpoints was discussed by the Ad-Hoc Group on the Economic Appraisal of the Health Effects of Air Pollution. (EAHEAP, 1999). The EAHEAP report noted that there were no direct studies of people's valuation of reducing the risk of a death brought forward by air pollution. The valuation estimates were therefore inferred by adjusting a baseline figure obtained from other contexts. Adjustments were based on the expectation that those at risk would take account of their own prognosis, age and health in assessing the values they attached to further reductions in air pollution. The uncertainties in this process resulted in a wide range of estimates from 2,600 to 1.4 million to avoid a death being brought forward by air pollution. It was highlighted that the deaths associated with increases in air pollution are thought to occur in the elderly and among those with pre-existing serious cardio-respiratory disease, and so that life is shortened typically by weeks or months but not years (note the loss of life expectancy is not known precisely). The low value (£2600) has been used here for valuation of deaths brought forward in the ***low*** analysis. The report also presented some adjusted values based on the assumption – see earlier – that those at risk of earlier death following days of higher air pollution have on average a lower life expectancy (say, in the order of 1 month to 1 year) than average for elderly population (12 years). The value of £1.4 million was therefore adjusted to £120,000 (for one year), and by 0.7 to reflect a lower quality of life (0.2 to 0.7) than average for elderly population (0.76). The use of upper quality of life adjustment gave a value of £110,000. Other studies (e.g. the ExternE study – EC, 1998:2001) have provided a single value, based on an assumption of the period of life lost. This has been used in European cost-benefit analysis, assuming a period of 6 months of life lost, with a value of £110,000. This value has been used here in ***central*** analysis.

Note the upper value (£1.4 million) from EAHEAP has not been used here for time series studies (acute effects). Time series studies provide results in terms of changes in the number of daily deaths associated with air pollution. Aggregated over days, these results can be represented as the number of deaths per annum whose immediate life shortening was attributable to air pollution in the preceding days. These are described as the number of deaths brought forward, to indicate that in at least some of these cases, the actual loss of life is likely to be small – the death might in any case have occurred within the same year. There is an issue whether these effects can be added to the results of the cohort studies, i.e. the mortality effects of long-term exposure. In principle, cohort studies should capture the full mortality effects of PM. On that basis, it would involve double counting to add the PM-related mortality effects as estimated from time series studies. In practice, it may be that some aspects of the PM-attributable mortality identified by time series studies are *not* incorporated into the relative risk estimates of the cohort studies. In particular, this may apply to deaths brought forward by only a few days or weeks. Omission of time series estimates would therefore lead to some under-estimation of the total mortality impact. In this report, we have added the time-series and cohort studies, however, in selecting monetary values for the former, we do not believe it appropriate to use the unadjusted value for a death brought forward, as to do so would imply (in our view) a longer period of life lost, and would double count the benefits captured from the cohort studies.

We highlight that there are a number of specific valuation studies on mortality from air pollution that have recently been published (The Defra study 'Valuation of Health Benefits Associated with Reductions in Air Pollution', published June 2004, and the NewExt Study 'The Willingness to Pay for Mortality Risk Reductions: An EU 3-Country Survey', to be published by the EC 2004.). These studies provide values for a life year lost. As an interim position, the indicative value from one of these studies for a life year lost (£31,500) has been used in the ***low*** analysis. The previous value in use in European cost-benefit analysis (£65,000) has been used in the ***central*** analysis (e.g. see ExternE 1998:2001). Note the analysis of life years saved includes benefits that happen in the future, from current pollution reductions. As we have progressed through to valuation in this analysis, it is appropriate to discount these future benefits over time. The study has used a 1.5 % discount rate as used in the IGCB work (An Economic Analysis to Inform the Review of the Air Quality Strategy Objectives for Particles, 2001), on the basis of the following statement (reproduced from IGCB, 2001) taken from the Green Book:

'Some costs and benefits, such as for example risk of death or change in health state, might be seen as having a broadly constant utility value over time, regardless of changes in income. If so, then such future costs or benefits could be valued in 'today's' values and discounted at [the pure time preference rate], so avoiding the need to calculate separately a rate of increase in their value over time.' (Page 85, paragraph 17, The Green Book, HM Treasury, 1997). If health effects are measured in quantities (e.g. life years saved) and the value of health effects is increasing over time, discounting the volume of health effects at a lower rate than costs is an acceptable method of taking into account the future value of health effects. The Department of Health recommendation is that health effects are discounted at 1.5%. A rate of 1.5% is used because it is a measure of the pure time preference rate (including allowance for catastrophic risk). This is consistent with guidance from the Treasury Green Book. For the purposes of the analysis presented in this report, future health effects are discounted at 1.5%.'

Costs per carcass for the 2001 outbreak

The emissions were combined with external cost estimates for air emissions in rural locations to give a cost per carcass. The results are shown below, for a restricted low and high value, and best guess values, based on the NAEI analysis.

Environmental costs of air pollution from mass pyres, per carcass

	NOx	CO	SO2	HCl	PM10	Dioxins	B[a]P	C	Total
Low	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**
Cattle	0.30	0.05	2.92	0.00	2.01	3.60	2.40	4.78	16.06
Sheep	0.02	0.00	0.16	0.00	0.11	0.20	0.13	0.26	0.87
Pigs	0.07	0.01	0.70	0.00	0.48	0.87	0.57	1.14	3.85
Best guess	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**
Cattle	1.92	0.10	13.35	0.00	12.81	16.65	4.80	9.57	59.20
Sheep	0.10	0.01	0.73	0.00	0.70	0.90	0.26	0.52	3.22
Pigs	0.46	0.02	3.21	0.00	3.09	4.00	1.15	2.28	14.21
High	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**	**£/carcass**
Cattle	3.85	0.19	26.70	0.00	25.63	29.25	9.59	19.13	114.35
Sheep	0.21	0.01	1.45	0.00	1.40	1.59	0.52	1.04	6.22
Pigs	0.92	0.05	6.41	0.00	6.17	7.03	2.29	4.57	27.45

A number of important caveats are associated with these values:

- The numbers only include costs that occur in the UK. All trans-boundary pollution / impacts are excluded.
- The values for NO_X and SO_2 include secondary particulate (PM_{10}) formation (nitrates and sulphates).
- The values for VOC include ozone formation and effects.
- The values for NO_X do not include ozone formation and effects.
- The analysis assumes no threshold of effects and implements concentration-response functions linearly.
- Future life years lost have been discounted using agreed 1.5% discount rate.
- The low value assumes £3100 for death brought forward and £31500 per life year lost, with future life years discounted (1.5%). The central value assumes £110000 for death brought forward and £65000 per life year lost, with future life years discounted (1.5%). The high value is double the central value (based on the use of the WHO recommended risk factor for chronic health effects).
- All chronic mortality impacts use original $PM_{2.5}$ functions to PM_{10} pollution metric.
- Values for dioxins and B[a]P are based on US EPA risk factors.
- The numbers exclude several categories of impacts, notably: impacts on ecosystems (acidification, eutrophication, etc), effects of NO_X on ozone formation, impacts on cultural or historic buildings from air pollution, mortality from PM_{10} on children, chronic morbidity health effects from PM_{10}, morbidity and mortality from chronic exposure to ozone, change in visibility (visual range), effects of ozone on materials.
- Environmental costs of air pollution vary according to a variety of environmental factors, including overall levels of pollution, geographic location of emission sources, height of emission source, local and regional population density, meteorology etc. These numbers take these issues into account to a certain degree only.
- The range of estimates is based around a 'restricted' range on certain key health values only. It does not include a range of uncertainty, nor consideration of uncertainty for other aspects of the analysis.
- The values for B[a]P and dioxins are not based on primary studies. The value for B[a]P is based on earlier ExternE work looking at the transport sector. The value for dioxins is based on the COWI incinerator study (see main text). The applicability of these values to emissions, especially in relation to the location of the emissions, is a major source of uncertainty. Note for the analysis of future disease control, we have used an adjusted value for dioxins, which takes into account the rural location of likely incinerators.