



CHEMICALS AND HEALTH

THE SOCIAL COST OF CHEMICALS

The Cost and Benefits of Future Chemicals
Policy in the European Union

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A WWF CHEMICALS AND HEALTH CAMPAIGN REPORT

A REPORT FOR WWF-UK

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Disclaimer

This report has been prepared by Professor David W Pearce of University College London and Imperial College London, and Dr Phoebe Koundouri of Reading University and University College London. Both authors are acting in their capacities as private consultants. University College London, Imperial College London and Reading University are not responsible for the contents of this document.

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EXECUTIVE SUMMARY

New European Union legislation on chemicals regulation will involve an extensive system of registration, risk assessment and authorisation for both new chemicals and existing chemicals. The system is known as REACH. The benefits of REACH arise primarily from (a) greater and improved information about what chemicals are being produced and used, and (b) behavioural responses to the costs of complying with REACH such that some chemicals will be withdrawn from use and substituted for by other chemicals, and some may possibly be withdrawn totally.

All EU legislation is required to be subjected to some form of appraisal of costs and benefits. In the case of REACH, an ideal approach would involve:

- an assessment of current and future exposure to chemicals with and without the legislation;
- a behavioural model which would show how the industry and users will respond to the true costs of compliance;
- dose-response functions for health and for environmental effects, linking exposure levels to changes in health status and to changes in environmental impact; and
- a procedure for placing money values on the changes in exposure such that the resulting monetised benefits can be compared to the costs of compliance.

Unfortunately, the information and resources to implement such an approach are not available. In this Report we have therefore resorted to what we term an *n*th best approach. We make what we regard as reasonable assumptions about some of the key variables and parameters, and we then adopt three different models to assess the benefits of REACH. We assess *only* the health benefits since we judge that the environmental effects cannot be estimated without a detailed stated preference approach to valuing such effects, and without far better information about exposure-response functions. Accordingly, the benefits of REACH exceed the estimates shown here.

We also caution that other features of assessing REACH are not satisfactory. In Chapter 3 we argue that the cost estimates for complying with REACH may be under- or over-estimates. At the time of writing, reasonably detailed estimates of the direct costs of compliance exist: i.e. the costs of registration and securing risk assessments and authorisation. While estimates of wider indirect costs, substantially exceeding the direct costs, are in circulation, documentation showing how those costs have been estimated is unavailable. We also note that the assumed reduction in exposure from implementing REACH will strike some as a serious under-estimate, and others as an over-estimate. Accordingly, there remains considerable room for debate, which is what we would expect in the absence of any publicly available detailed model of the industry and supply and demand responses.

Our first two models are based on the notion of a Disability Adjusted Life Year (DALY), a procedure for estimating the burden of disease and premature mortality in a single unit. We have taken estimates of DALYs in industrialised countries and

computed DALYs-per-capita. We have made projections of these losses to 2020, and have adopted World Bank estimates of the fraction of DALYs arising from exposure to chemicals. We then make an assumption about the extent to which REACH will reduce those DALYs — the 10 per cent assumption. This assumption corresponds closely to that suggested by the European Commission's estimates of the reduction in the number of chemicals substances (8-12 per cent). But we caution that numbers of substances is not the same thing as level of exposure, and it is the latter that matters.

Finally, we value a DALY in two very different ways. The first approach — Model I — looks at health expenditure in the UK and EU. The intuition is that this expenditure is spent on avoiding and treating the causes of DALYs, so we can compute health expenditure per DALY. The number of DALYs saved from REACH can then be multiplied by this unit cost to estimate health expenditure saved by REACH.

The second approach — Model II — proceeds in the same way but notes that the value of a DALY is greater than the healthcare costs incurred and must include the willingness to pay (WTP) of individuals to avoid the health states in question. We follow a procedure adopted in a World Bank study of pollution control and assign a WTP value to a DALY based on an anchor estimate of the value of a statistical life (VOSL) and an implied value of a life year (VOLY).

The third approach — Model III — proceeds differently and attempts to estimate the medical costs and forgone productivity from specific diseases or health end-states. Here again, some assumption is required about the extent to which these health states are due to exposure to chemicals. We followed a US study which assumes 10-50 per cent of the resulting costs arise from exposure to chemicals. We then adopt our 10 per cent REACH effectiveness estimate to calculate the benefits of REACH.

A summary of the results from Models I-III is shown in Table E.1. (The results are shown for the EU only since wider compliance cost estimates are not available for the UK). We would confidently expect Model II to produce higher results than Model I (since willingness to pay must exceed healthcare costs), but we have no prior expectation about the relationship between Model III and Model II since they work on different bases. We find that Model I produces benefits less than costs; Model II produces a strong likelihood that benefits exceed costs; and Model III produces benefits significantly exceeding costs. Once again, however, we stress that a number of assumptions have been made to secure this conclusion and it is open to anyone to challenge the assumptions. If so, the methodologies are sufficiently transparent that anyone can generate their own estimates based on what they regard as superior assumptions.

Since our models exclude all environmental effects, we argue that our benefit estimates are understatements. **Overall, our own judgement is that we feel confident that REACH generates net benefits.**

Table E.1 Summary of costs and benefits of REACH

		10⁹ : Present Value, s = 3%			
		Benefits	Costs	Net Benefits	B/C Ratio
Model I	EU	4.8 - 20.1	23.6	-3.5 to -18.8	0.21 to 0.85
Model II	EU	22.4 - 93.3	23.6	-1.2 to +69.8	0.95 to 3.95
		12.3 - 51.3	23.6	-11.3 to +27.7	0.52 to 2.17
Model III	EU	56.7 - 283.5	23.6	+33.1 to +259.9	2.40 to 11.01

1 THE ISSUE

1.1 The aim of this report

The European Commission has published draft legislation which seeks to establish a new system of **Registration, Evaluation and Authorisation of Chemicals (REACH)** covering chemicals marketed in quantities over one tonne per annum (CEC, 2003a). Despite the requirements of Article 130R of the Treaty of Union concerning the need to establish the costs and benefits of any regulation, and the requirements of Directive 76/769 which requires risk assessment and cost-benefit appraisal of regulatory measures affecting the chemical industry (CEC, 2002, para 1.1), the Commission has offered only incomplete attempts to estimate the costs and benefits of REACH (see, for example, Annex I of the 2002 draft of the White Paper — CEC (2002)). The issue arises, then, whether the Regulation passes a cost-benefit test. The purpose of this report is to assemble what data are available, and to explore alternative methodologies for determining the costs and benefits of REACH. Section 1.2.2 discusses an alternative interpretation of the legislation as an information-gathering device.

It needs to be stressed at the outset that a full and rigorous cost-benefit appraisal of REACH is not possible. This is because:

- a) The ways in which REACH will work, and hence the costs of implementing REACH, are still not entirely clear (see Chapter 2). Additionally, the precise number of chemicals is not known, nor, more importantly, are the behavioural reactions of suppliers and users of chemicals. In short, the change in the dose of, or exposure to chemicals is not known with any degree of certainty.
- b) The epidemiological basis for determining the benefits of REACH in terms of improved public and occupational health is not known across the many chemicals affected. Thus, even if the change in exposure was known, the change in health and environmental responses needs to be estimated across the relevant chemicals.
- c) There are alternative approaches to placing an economic value on the health benefits of REACH. This results in a potentially wide range of damages.
- d) REACH will generate benefits in terms of reduced non-health environmental damage, but the relationship between the chemicals and environmental responses is not known. Procedures for placing economic values on many environmental change are available but do not yet cover some of the important impacts, e.g. on biological diversity.

These problems are discussed in more detail in Chapter 2. Nonetheless, this report shows that it is possible to obtain some broad ballpark estimates of gains and losses by making reasonable assumptions and by pursuing several different methodologies for estimating benefits. The report sets out the methodologies in a transparent manner so that anyone disagreeing with the assumptions can change them and test the sensitivity of the results to different assumptions. The results presented at the end of the report represent our own best guess estimates. It remains the case that a far more

detailed and rigorous approach is required for the analysis of the social costs and benefits of the new chemicals strategy. Such an analysis would, in our view, be costly by comparison with the norms for regulatory cost-benefit studies, but would be insignificantly expensive compared to the costs of being in error in devising such a policy. At the very least, we consider that there are powerful arguments for engaging in cost-benefit thinking, i.e. for setting out the relevant arguments and parameters for doing a cost-benefit study, even if we cannot engage in a full-blown analysis ourselves.

1.2 REACH

Registration, Evaluation and Authorisation of Chemicals (REACH) covers chemicals marketed in quantities over one tonne per annum. All chemicals manufactured in quantities greater than 1 tonne p.a. must be *registered*. Those manufactured in quantities greater than 100 tonnes p.a. must be *evaluated*, and chemicals giving rise to special concern must be *authorised*. The authorised category covers, *inter-alia*, carcinogens, mutagens and reproductive toxic substances (CMRs). The Commission is proposing to authorise PBT (persistent, bioaccumulative and toxic), vPvB (very persistent and very bioaccumulative) and EDCs (endocrine disrupting chemicals) on a case by case basis (Point 44 of CEC, 2003a, Volume I). A central feature of the new chemicals strategy is that it brings under one regulatory umbrella new and existing chemicals, the vast majority of chemicals in the market place being existing chemicals.

1.2.1 The baseline issue

REACH replaces a set of existing schemes, and the nature of those schemes varies slightly according to member state. Hence the relevant benefits and costs arising from REACH must always be seen as being relative to the existing systems of control, not to a do nothing situation. One immediate complexity is that existing laws may not be fully understood or, if understood, may still not be complied with (Warner and Thompson, 2002). If so, the baseline for comparison is open to debate. It could be argued that existing legislation with full compliance is the appropriate baseline, since all parties should comply. Equally, it could be argued that non-compliance might arise from the nature of the current legislation, in which case REACH would, hopefully, replace inefficient regulation with efficient regulation. If so, the relevant baseline for comparison is the existing situation inclusive of any non-compliance. Both arguments have validity. In the models developed in this report, we take the status quo, inclusive of any non-compliance, as being the baseline against which REACH is to be judged. The caveat to this approach, therefore, is that improvements to the baseline could be secured by enforcing compliance with existing regulations. If so, our estimates of benefits arising from REACH will be over-stated. If, on the other hand, REACH is regarded as being essential precisely *because* existing regulations have built-in risks of non-compliance, then the benefits will properly be reflected in the estimates we make.

1.2.2 REACH as an information device

Section 1.1 set out the basic reasons why it is not possible to engage in a full or rigorous cost-benefit analysis of REACH. Arguably, such a cost-benefit analysis

should have accompanied the drafts of the legislation. The European Commission is gradually releasing partial cost-benefit studies as the details of REACH are formulated. However, it could equally be argued that the *purpose* of REACH is to gather the information that is necessary to conduct a cost-benefit appraisal, i.e. the benefits of REACH lie in the information it generates. In turn, that information will help to reduce risks to health and ecosystems. On this view, cost benefit analysis could not be conducted on REACH because the information is not available to carry out such an evaluation and the purpose of REACH is to generate that information. Against this, it needs to be borne in mind that REACH has significant resource costs, so that consideration would need to be paid to all the alternatives of achieving a similar information goal without the costs of registration, evaluation and authorisation. Moreover, it remains the case that some attempt could still be made to evaluate gains and losses from the generation of such information. We take the view, therefore, that the cost-benefit approach is still of value.

1.2.3 The chemical domain

The EU has some 100,000 chemicals listed as being on the European market between 1971 and 1981 — the so-called existing chemicals. The chemicals industry has not had to generate safety data on these substances. All chemicals placed on the market since 1981, the new chemicals, have had to be notified to the regulator. It is believed that there are now around 30,000 existing chemicals being sold in volumes of greater than 1 tonne per annum, with 10,000 being sold in volumes greater than 10 tonnes. The total number of chemicals on the market may grow if new substances increase in number faster than existing substances are retired or replaced, but REACH is expected to result in some withdrawal. The predicted degree of product withdrawal is not known, but the Commission has quoted estimates of 8-12 per cent withdrawal of chemicals from production or use. However, there is an additional and unknown number of intermediates in use, with estimates ranging from 50,000 to 120,000 (RPA and Statistics Sweden, 2002; DEFRA, 2002). Whether intermediates are accounted for or not, affects the estimates of costs and benefits¹.

As we show in Chapter 2, the large number of chemicals involved precludes the adoption of the ideal approach to cost-benefit appraisal. This ideal approach would seek to identify exposure-response (or dose-response) functions for each chemical, both in terms of health and environmental effects, and would then estimate the effects of REACH in reducing exposure levels. Reduced exposure levels would then translate into reduced effect levels, and effects can be valued in monetary terms using standard economic valuation procedures. While this approach is feasible for a few chemicals, it is not feasible for 30,000 or more chemicals, particularly given the fact that basic safety data are not available for the majority of these chemicals — in fact only a very small number will have sufficient information for such an analysis. Chapter 2 discusses this issue in more detail.

Under REACH, all chemicals used in quantities above 1 tonne will be registered in a central database. This threshold represents an increase over the current regulation for

¹ Intermediates are defined as substances that are used exclusively for the synthesis of another substance or other substances. Since intermediates tend to have low exposures, concerns have been raised that the inclusion of intermediates will overwhelm REACH and inhibit attention being paid to high priority substances (e.g. see DEFRA, 2002).

new (post 1981) chemicals, which is 10 kg. The 1 tonne threshold in the REACH system applies to *new* substances and to *existing* substances. Given the large number of chemicals, priority will be given to those produced in the largest quantities (a proxy for exposure), and also those where there is some known cause for concern. Those produced or imported at over 1,000 tonnes per annum will have to be registered first, followed by those of lower tonnages. The chemical industry would have the responsibility of ensuring that chemicals are safe, i.e. the regulation takes the form of producer responsibility common in other forms of EU environmental regulation ². This responsibility extends to importers of chemicals from outside the EU. But downstream users will also have responsibility for downstream safety. Authorisation of high concern chemicals will only be given on evidence of negligible risk or acceptable risk (relating to the benefits of using the chemical in question). The Commission is of the view that such regulations should encourage (a) substitution of less damaging chemicals for more damaging chemicals and (b) innovation in the chemicals sector to reduce chemical usage or find less damaging alternatives. Both (a) and (b) affect any estimate of the costs of the regulation — see Chapter 3³

REACH involves the following stages of assessment of chemicals:

- (a) *Registration* of all substances produced in quantities above 1 tonne per annum. Registration only is likely to affect around 80 per cent of substances. Downstream users (formulators and industrial users) must also indicate if chemicals are being used for purposes other than those originally intended (unintended uses). Testing will generally be limited to in vitro methods for chemicals produced or imported at <10 tonnes per annum.
- (b) *Evaluation* of substances produced in quantities above 100 tonnes per annum (t/y) (perhaps 15 per cent of existing substances), and of any other substances giving rise to particular concern. A schedule of different levels of information requirements for chemicals produced/imported in quantities 10-100 t/y, 100-1000 t/y and above 1000 t/y is given in CEC (2002) Action 3B;
- (c) *Authorisation* of high concern substances, i.e. a system of special permission for specific uses of a given chemical. This is expected to relate particularly to persistent organic pollutants (POPs) and carcinogenic, mutagenic or reprotoxic (CMR) substances. This should affect the remaining 5 per cent of all substances. The key Commission departments (DG Environment and DG Enterprise) have proposed, in their legislative text for consultation, that persistent, bio-accumulative and toxic (PBT) substances, very bio-accumulative substances (vPvB) and substances of equivalent concern such as endocrine disrupters should be considered on a case-by-case basis. Others have argued that some respiratory sensitisers should also be subject to authorisation (DEFRA, 2002).

Overall, then, REACH is a procedure for registering, testing and authorising new and existing chemicals. Compared with existing practices, REACH would ensure that

² For example, packaging and packaging waste.

³ Terminology can be confusing. Throughout, reference to the costs of the regulation refers to the resource costs borne by industry and society as a whole in order to comply with the regulation — the compliance costs. The social cost of the chemical refers to the damage done by the chemical in terms of human health and the environment — damage costs. The benefit of REACH is the reduced damage costs arising from the regulation.

safety data are available for all chemicals on the market, rather than just for new chemicals, provide more comprehensive information about the hazards associated with each chemical, and, potentially, a reduction in the use of those chemicals with high social concern.

2 AN IDEAL APPROACH TO THE EVALUATION OF REACH

In order to show what is needed for a proper evaluation of REACH this chapter sets out an ideal approach, i.e. one based on a conceptually sound model, but which ignores the availability or otherwise of the relevant data. This permits us to judge the gap between what should be done and what can be done in practice.

2.1 Uncertainty and the ideal approach

One misconception needs to be dispelled at the outset. One of the criticisms of economic (cost-benefit) approaches to policy evaluation is that they add to the uncertainty associated with evaluation. As such, it is argued, the approaches are best not adopted in the first place. There are indeed uncertainties, and often significant uncertainties, in cost-benefit appraisal. The problem is that the uncertainty is not reduced by *not* adopting cost-benefit analysis (CBA). Invariably, uncertainty is actually *increased* by not adopting CBA. There are many reasons for this conclusion, but two will suffice.

First, what CBA does is to compare benefits in the same units (money) as costs⁴. This permits a decision of whether or not to adopt the policy at all. Adoption follows if benefits exceed costs and not otherwise. Failure to monetise benefits means that the choice context is one of *cost-effectiveness* in which costs are in money units but effectiveness is in a different unit, e.g., some notion of risk reduction (for example, lives saved). But cost-effectiveness can only rank alternative policies, it cannot say whether *anything* should be done. We may for example, choose policy A over B because A secures more risk reduction per euro than B (more bangs for the buck). But both A and B could still fail cost-benefit tests, indicating that neither should be undertaken. Thus, a failure to adopt CBA *increases* risk because a new risk emerges, namely that wrong policies are adopted.

Second, the risk reduction in question will show up in various ways. At the simplest level, these may comprise reduced mortality and reduced morbidity. Cost-effectiveness analysis cannot now be conducted unless we have some idea of the relative importance of reducing one form of risk over another form of risk — e.g. whether a reduction in chronic bronchitis is more important than a reduction in IQ impairment. Relative importance is measured by a set of weights, such that the ratio of the weights on any two forms of risk reduction reflects the relative importance of reducing one risk compared to another. Thus if the weight on IQ impairment is 3 and

⁴ This simple observation also explains why one cannot logically avoid monetisation. First, all policies have costs. If they did not have costs, there would be no need to consider whether or not they are good policies. Hence the acceptance of a policy implies that benefits must exceed costs, which sets a lower bound on the scale of monetary benefits. If the policy is rejected, the reverse applies. Second, costs are measured in money terms and few people have difficulty in agreeing that this is the correct way to measure costs. But costs are simply negative benefits, since all costs are properly measured by the forgone benefits of spending money on the chosen project rather than on something else. So, positive money costs are the same thing as negative money benefits. It follows that benefits must also be expressible in money units.

that on chronic bronchitis is 2, the relative weight is 1.5. If weights are not adopted, it is not possible to make any comparison between options, i.e. rational decision-making is not possible. All decision analysis involves one means or another of selecting weights: by implied political preference, overt expert judgements, or, in the case of CBA, individuals' willingness to pay (WTP) for one change compared to another. In short, CBA's weights are prices. Compared to a situation in which there is no knowledge of weights at all, CBA reduces uncertainty and does not increase it. It then becomes an issue of which set of weights is better. One advantage of CBA weights (prices) is that they reflect the preferences of those exposed to risk, and are hence more democratic than expert weights⁵.

2.2 An ideal model

The ideal approach to measuring the social benefits and costs of REACH would be as follows.

First, some assessment would need to be made of the extent to which REACH will reduce human and environmental exposure to chemicals. Refer to this change as $_X$ where X refers to exposure. In reality, $_X$ is a vector of many different chemicals. For those that are withdrawn completely from the market as a result of REACH, $_X = 100\%$. For others, $_X \neq 0$. This stage of the analysis would therefore produce the *policy effect on exposure*.

Second, we need an *exposure-response* relationship. Two effects can be identified. The first is the effect on human health, call this $_H$. Again, there will be many different health effects, ranging from reduced premature mortality through to changes in hospital admissions, days away from work, etc. So, $_H$ is also a vector. It is helpful to divide human health effects into reduced *occupational risks* ($_H_O$) and reduced *public health risks* ($_H_P$). This is because there may be differences in the way the two effects are to be valued in monetary terms. The second effect is the *environmental impact* on e.g. ecosystems and biodiversity. Call this $_E$. For any one chemical, then, the sum of the effects is $_H_O + _H_P + _E = _I$ where I is overall impact. (See Section 2.3 for a discussion of other possible candidates for benefits and costs).

Third, we need economic values for each impact since it is implicit in the equation for $_I$ that the effects are in the same units. We refer to these as the *shadow prices* because they are the prices that would be attached to the reduced risk if only there was an overt market in risk reduction. These shadow prices reflect individuals' willingness to pay for avoiding the ill-health or negative environmental impact associated with chemicals. Again there will be a whole set of shadow prices covering all of the impacts. We refer to these shadow prices as P and they are formally equivalent to the weights discussed in Section 2.1.1.

Fourth, we need to know *when in time* the changes in exposure will occur. This is because future changes in exposure will be valued less than near-term changes in exposure. The economic concept that reflects the different weights attached to time is

⁵ It could be argued that political weights are best of all since politicians are elected to make such decisions. Unfortunately, the political model underlying this view is naive, and assumes politicians always act in the best interests of voters. Moreover, techniques such as CBA are designed as checks on political decision-making: this is the purpose of policy analysis.

known as a *discount factor*. The process of attaching weights to time is known as *discounting*. The discount factor (DF) is linked to the discount rate (s) (expressed as an interest rate, i.e. in percentage terms) in the following manner:

$$DF = \frac{1}{(1+s)^t} \quad [2.1]$$

where t is time (years from the present)⁶. Timing is important since the EC White Paper assumes a very rapid start to REACH and the following deadlines for dossier registration:

Over 1000 t/y substances	end 2005
100-999 t/y substances	end 2008
1-99 t/y substances	end 2012

However, while these are deadlines for registration they are not the same as the points in time when exposure will be reduced. In the models developed in later chapters we experiment with several different assumptions about the start times for exposure reduction.

Fifth, we need to know *where* the exposure changes occur. For example, if they occur in heavily populated areas the benefits of risk reductions there will be higher. Environmental effects are even more location-specific. Unfortunately there appears to be no basis on which to determine the geographical variation in exposure reduction.

The ideal model can now be summarised as follows. The benefits that ensue from REACH are given by⁷:

$$PV(B) = \frac{\sum_{i,j,t} \Delta I_{i,t} (\Delta X_{j,t})}{(1+s)^t} \quad [2.2]$$

The notation I refers to the individual impacts and j to the individual chemicals. PV(B) refers to the *present value of benefits* from REACH and it is this sum that would be compared to the present value of costs (for example, 1.4 to 7.0 billion for the EU as a whole, with s = 3% — see Chapter 3). REACH would pass a cost-benefit test if PV(B) > PV(C), i.e:

$$PV(B - C) = \frac{\sum_{i,j,t} \Delta I_{i,t} (\Delta X_{j,t})}{(1+s)^t} - (\text{€}1.4 - 7.0 \text{ billion}) \geq 0 \quad [2.3]$$

⁶ Space precludes further discussion of discounting. It should be noted that it is not possible to avoid discounting. Not discounting is formally equivalent to discounting at 0%. Unfortunately, zero discounting has logical implications that make it undesirable, however reasonable it may at first appear. See Olsen and Bailey (1982).

⁷ Equation [2.2] ignores location for convenience of exposition, but it will be appreciated that benefits and costs vary by location.

Notice that [2.3] could be met overall in the EU but REACH in any one country might fail a cost-benefit test. Similarly, REACH could fail a cost-benefit test at the EU level, but pass it in any one country.

2.3 Other benefits and costs

Section 2.2 suggests that the relevant benefits from REACH are primarily human health and environmental effects. However, discussions of REACH have raised several other issues which appear relevant to a cost-benefit analysis.

2.3.1 Losses and gains to consumers

Chemicals exist for a purpose and benefits accrue from the use of those chemicals. REACH will change the number of chemicals available by inducing some withdrawals from the market, and will also change the prices of other chemicals that are marketed. Depending on the assumptions made about the behavioural reaction to REACH, prices might rise due to the costs of registration etc. or they may fall due to induced innovation. If prices rise, consumers will lose consumer surplus and this must be counted as a cost of REACH. Price reductions would appear as a benefit. If chemicals are withdrawn, then users of those chemicals will need to substitute other chemicals, and this process is not cost-free. In so far as the costs of REACH are estimated by the costs of registration etc. then some of those costs may be passed on to users, and the compliance cost estimates would capture at least some of the costs to users. But, in the absence of a publicly available full-scale economic model of the chemical industry we have no way of knowing what will happen to prices or to rates of substitution⁸. Accordingly, we have no way of knowing how serious these impacts are. We simply note that the benefits of REACH may be under- or over-estimated because of this consideration⁹.

2.3.2 Employment and competitiveness

It is widely argued that changes in employment resulting from a policy should factored into a cost-benefit analysis. If REACH has the effect of raising costs and prices, it is possible that it will induce unemployment. If it stimulates innovation and cost reductions, it is possible that it will stimulate employment. Those who believe that cost increases will prevail tend to emphasise the negative impacts of regulation of competitiveness. A further extreme is that those who believe in the cost-increasing scenario argue that industries will be tempted to relocate in response to high compliance costs — the migrating industries argument. Unsurprisingly, employment and competitiveness arguments are used by supporters and opponents of new regulations alike. We propose to take the view that neither employment effects nor competitiveness are likely to be serious issues in the context of REACH. There are several reasons for this. First, there is little evidence that environmental regulations have significant effects on employment. Second, it is unclear what competitiveness means in a European context in which the euro (and non-euro currencies) float against

⁸ In its Draft Impact Statement, the Commission refers to a DG Enterprise Note: *A Microeconomic Model to Assess the Economic Impact of the New Chemicals Policy*. But we have been unable to access this document. See CEC (2003c).

⁹ Commissioner Margot Wallström refers to REACH as producing a win-win situation for everybody (Wallström, 2003). This is not a credible point of view. All policies involve costs.

competing countries. Third, we know of no evidence to suggest that firms relocate in response to environmental regulation. Other costs are far more important to locational decisions than regulatory compliance costs. Fourth, in so far as regulation can be employment-enhancing this tends to be confined to those regulations that require significant labour or abatement equipment inputs, and that appears not to be relevant to REACH. While we do not rule out negative employment effects, our views are consistent with those reviews that have investigated these issues (e.g. Sprenger, 1997)¹⁰. Similarly, detailed econometric studies have generally failed to find significant negative or positive employment effects from regulation even in contexts where regulatory costs are regarded as being significant fractions of the value of output — see e.g. Morgenstern et al. (2002).

Finally, *if there are* employment losses it would be appropriate to consider the social cost of the loss of wellbeing associated with being unemployed (Markandya, 2000). However, it is important to define these costs correctly. They would include costs such as any associated ill-health due to longer term unemployment and forgone income, but transfer payments within society — unemployment benefit, for example — is not a net cost to society (one individual gains and taxpayers lose, but in equal money amounts). Any employment *gains* would need to be treated by lowering the compliance cost of the regulation. In the EU context, this is unlikely to be relevant to a CBA¹¹.

2.3.3 Reduced animal testing

We have not been able to obtain any estimates of the extent to which REACH will reduce the use of animals for chemicals testing. Some experts have told us the effect is not likely to be significant, but others figure expectations of reduced testing prominently in their support for REACH. We simply note that if animal testing is reduced *because of the legislation*, then it is quite correct to include the benefits of that reduced testing in a CBA. If it increases *because of the legislation* then there is a cost to be assigned to REACH. The way these costs or benefits can be estimated is to conduct a stated preference (questionnaire) study in which individuals are asked their willingness to pay to reduce animal testing by some specified fraction and their willingness to pay to avoid increases in animal testing. While it is likely that such an approach would elicit a significant number of protest responses from those who believe that animal testing is morally wrong (and hence not something they should pay to reduce), such questionnaires have been successful in other comparable contexts (e.g. reducing pesticides in food). The central point is that the gain (loss) in human wellbeing from knowing that animal testing is reduced (increased) is a legitimate benefit (cost) to be included in a CBA and constitutes a non-use value. As we understand it, the impacts of REACH on animal testing are very unpredictable and very dependent on the detail of the legislation (and its interpretation — e.g. how far

¹⁰ We are aware of the Arthur D Little report for Federation of German Industries (Arthur D Little, 2002) which estimates a *minimum* loss of 150,000 jobs and a *maximum* of 2,350,000 in Germany alone. A review of this study is beyond the scope of this report.

¹¹ This issue is not straightforward. Labour should be valued at its shadow wage, i.e. what it would secure if it was employed elsewhere. In economies with near full employment, this shadow wage would be close to the ruling wage, which means that the adjustment would make little difference. The adjustment would be relevant if the policy affected economies with high unemployment, e.g. some accession countries.

companies can go in saying one chemical is similar to another), and the extent of funding of alternatives (and success in developing them). Thus, whether testing will go up or down is not currently predictable.

2.4 Problems with the ideal model

Models of the kind shown in equation [2.3] have been used fairly extensively for conventional air pollutants such as SO_x, NO_x, PM and VOCs (e.g. see Olsthoorn et al. 1999; Krewitt et al. 1999). These models make use of long-established emission-diffusion-deposition models (such as RAINS Europe) which also contain measurable ecosystem impacts based on notions of critical loads¹². They also have established exposure-response relationships for human health. The policies that are simulated also have known, or reasonably known, time-schedules over which the pollutants are reduced. Finally, they utilise economic values per effect based on long-standing work under the ExternE programme of DGXII in the European Commission.

The contrast with what is known about REACH is a stark one. In the case of REACH we do *not* know:

- the effects of REACH on exposure ($_X$) since this is dependent on the behavioural reaction of producers, users, and regulators to (a) the changes in information generated by REACH, and (b) the costs of registration, evaluation and authorisation. Put another way, we have no economic model of the chemical industry — including users — with which to simulate the effects of any policy change such as REACH;
- the health and environmental exposure-response functions ($_I(_X)$) for the chemicals, of which, in any event there are many thousands¹³;
- the locations at which risks will change;
- the split between occupational and public health effects; and
- the time-schedule of $_X$ either, although some assumption could be made about this with respect to the registration deadlines.

We do have some economic values for health end-states based on the same ExternE work as used by existing air pollution cost-benefit studies¹⁴, but valuation of

¹² A critical load is the maximum level of deposition of airborne pollutants that produces no discernible change in the receiving ecosystem. Above this level, some form of ecological damage occurs. Note that critical loads relate solely to ecosystems and not to health effects. A critical level would be that ambient concentration that produced no discernible change in, say, human health, materials corrosion, crop loss, etc.

¹³ Substantial amounts of information do exist for some chemicals. Hence one more sophisticated approach would be to identify exposure-response functions for at least some chemicals of high concern, and evaluate those. Resources and time have not allowed us to do this in this report.

¹⁴ ExternE is a major programme of work funded by the European Commission and aimed at valuing the effects of air pollutants from energy and transport sources, primarily conventional pollutants such as SO_x, NO_x, PM etc. See European Commission (1999) and <http://externe.jrc.es>.

environmental effects would not be possible since we have no idea of the end states of the changes in e.g. biological diversity.

We conclude that it is not possible to approximate the ideal model in the case of REACH. The information is simply not available.

2.5 N-th best models

In the circumstances, we have to proceed in a far more *ad hoc* way.

The first thing to do is to invert equation [2.3] and find out just how large the benefits need to be for REACH to pass a cost-benefit test. We do this for health effects only since we cannot estimate environmental effects. This procedure gives us a benchmark. If health benefits exceed this level then we know that REACH passes a cost-benefit test comfortably. We also have a minimum estimate of benefits since environmental effects are not calculated and these unknown benefits would need to be added.

Second, we need some crude ways in which benefits can be estimated under certain assumptions. The alternatives are set out in Chapter 4 and range from looking at the probable averted health expenditure costs due to reduced incidence of disease, to more extensive willingness to pay approaches.

3 THE COSTS OF REACH

3.1 Approaches to cost measurement

A cost-benefit analysis requires that the costs of any policy be estimated. The relevant costs are those of the total with policy situation minus the total costs of the without policy situation, regardless of who bears those costs¹⁵. The without-policy context relates to a continuation of existing legislation. The relevant costs are those accruing to *any* agent affected by the policy, including manufacturers and importers of chemicals, downstream users, government and consumers. Cost also relates to any loss of human wellbeing, and not just to money costs (resource costs). For example, if one effect of the policy is to substitute less harmful but less efficient chemicals for existing chemicals, then the loss of beneficial use of the chemical constitutes a cost. Working out who bears the costs is immensely complex and would normally involve some general equilibrium model of the chemicals sector, i.e. an economic model that shows how costs would be apportioned between agents, given that some of the initial compliance costs may be passed on to users and consumers. Better still, a dynamic general equilibrium model would estimate the impacts of the legislation on issues such as innovation and substitution across the next 10 or 20 years. No such model exists, publicly anyway, by which to estimate these effects¹⁶. Accordingly, the only detailed estimates of costs are those based on fairly rudimentary multiplication of the number of chemicals to be registered and tested, and a unit cost of testing which varies according to the level of testing (RPA, 2001; RPA and Statistics Sweden, 2002; CEC, 2002). These are referred to as the direct costs of REACH. CEC (2003b) refers to additional estimates of indirect costs, which should encompass some of the general equilibrium effects. The sequence of costs estimates is discussed below.

3.2 Early CEC cost assessment

The European Commission included a primitive cost assessment in the draft White Paper of 2002 (CEC, 2002, Section 3.4). This suggested that REACH will impose costs of 2.1 billion over the period 2001-2012 for the EU 15 countries as a whole. Unfortunately, the basis for this cost estimate is not provided in the White Paper. The White Paper records the following estimates:

Base-set testing:	85,000 per substance
Level 1 testing:	250,000 per substance
Level 2 testing:	325,000 per substance

In general, testing costs rise as tonnages increase. The testing package at 1 tonne is termed the base set, at 100 tonnes it is Level 1, and at 1,000 tonnes it is Level 2. The rough estimates of quantities are

1-99 tonnes	80 per cent of production
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¹⁵ CBA is not indifferent to the distributional burden of the costs or the distribution of the benefits, contrary to popular criticism of CBA. For an extensive discussion, see Pearce (2003). The distributional effects of REACH appear to be extremely uncertain and the issue is not addressed here.

¹⁶ CEC (2003c) refers to something that may qualify as such a model: *A Microeconomic Model to Assess the Impacts of the New Chemicals Policy*, but we have been unable to access this document.

100-999 tonnes	15 per cent of production
1000+ tonnes	5 per cent of production

Multiplication of the unit cost figures by these quantities produces a very much larger aggregate cost figure than 2.1 billion (around 3.2 billion). One possibility is that the CEC figure is discounted, which would be the proper way to present it. However, there are no indications in the draft White Paper that this was done. It is more likely that there is some unstated assumption about the number of chemicals going out of production due to the costs of testing. As noted in Section 1.1, the behavioural reaction of chemicals suppliers is an essential piece of information for a cost-benefit appraisal. The cost estimates in the early White Paper are best ignored since their foundation is obscure. They have, in any event, been superseded by later estimates — see below.

3.3 The RPA-Statistics Sweden cost assessment for the EU

RPA-Statistics Sweden (2002) present an altogether more sophisticated set of compliance cost estimates, but which still fall short of a rigorous approach to cost estimation. First, they use industry-based unit testing cost figures. Second, they adopt a scenario approach to allow for different chances that many chemicals will be withdrawn from the market (rationalisation) and differing cost estimates. Third, they allow for the likelihood that intermediates may have to be included in the testing process. Fourth, they allow for the registration of unintended uses. Fifth, they correctly estimate the costs in present value terms. Sixth, they include some savings for <1 t/y substances which no longer need to be registered, and add in dossier preparation costs and authorisation costs for sensitive chemicals. However, the estimates ignore the wider general equilibrium effects, which are likely to be important — see below.

The unit cost testing estimates are significantly above those of CEC (2002) for Level 1 and Level 2 testing but vary from a low of 18,850 per substance to 155,000 per substance for base set testing, low and high figures corresponding to low and high tonnages, and to differing assumptions about the level of testing. The costs are set out in Table 3.1

Table 3.1 RPA-SS estimates of unit testing costs

Tonnage	Base set costs: per substance	Level 1 costs: per substance	Level 2 costs: per substance	Total: per substance
1-9	18,850 - 31,393	Not applicable	Not applicable	18,850 - 31,393
10-99	74,100 - 154,972	Not applicable	Not applicable	74,100 - 154,972
100-999	75,050 - 154,972	419,800	Not applicable	494,850 - 574,772
1000+	88,800 - 154,972	Not applicable	683,400	772,200 - 838,372

Source: RPA-Statistics Sweden (2002)

RPA-Statistics Sweden adopt a scenario approach to the impacts of REACH in withdrawals from the market and some other assumptions. The scenarios are:

Scenario 1 Low number of chemicals placed on the market, and no intermediates in need of registration and testing. Put another way, this scenario has a high withdrawal rate, but also has high grouping of chemicals and few repeat registrations.

Scenario 2 Low numbers of chemicals, low number of intermediates, high grouping and low repeat registrations.

Scenario 3 Middle range assumptions for numbers of marketed chemicals, intermediates, grouping and repeat registrations.

Scenario 4 High range assumptions for numbers of marketed chemicals, intermediates, low grouping and high repeat registrations.

The following additional costs then need to be added to the testing costs:

- a) costs of dossier preparation
- b) costs of any authorisations and accelerated risk management for high concern chemicals

Since the threshold notification is raised under REACH to 1 t/y, many smaller registrations will cease, saving the relevant corporations money. RPA-Statistics Sweden estimate that this will save around 42,000 per substance, or 69 million in PV terms over a 10 year period. This is deducted from the costs in Table 2.2.

The discount rate used by RPA-Statistics Sweden is 3 per cent and we judge that to be correct in light of recent adjustments to UK official discount rates (from 6.0 per cent to 3.5 per cent — see HM Treasury, 2003) and academic work (Pearce and Ulph, 1999).

The final cost estimates thus amount to:

Testing costs + Dossier Preparation costs + Authorisation costs — Savings on <1 t/y substances.

Table 3.2 shows the results of the RPA-Statistics Sweden analysis.

Table 3.2 RPA-SS Compliance cost estimates for REACH (PV with s = 3%)

Case	Numbers registered: Scenario	Total PV costs: million
Low test, low authorisation, low ARM	1	1378
	2	2531
	3	2689
	4	4743
Medium test, authorisation and ARM	1	2212
	2	3354
	3	3605
	4	5958
High test, high authorisation, high ARM	1	2989
	2	4122
	3	4448
	4	7045

Note: ARM = accelerated risk management

Taking the medium case and scenario 3, costs are of the order of 3.6 billion. This figure is confirmed as CEC s most likely estimate of the direct costs (CEC, 2003b). However, it is unclear if this most likely estimate has been adjusted for a time horizon of 2020 to make it comparable to estimates of the indirect costs (see Section 3.5).

Timing is important since the White Paper assumes a rapid start to REACH and the following deadlines for dossier registration, the timing being from the date of the legislation s entry into force¹⁷:

Over 1000 t/y substances and CMRs in lower volumes:	year 3
100-999 t/y substances	year 6
1-99 t/y substances	year 11

Expressed as an annuity, i.e. an annual sum of 3.6 billion over the period 2002-2012, the estimates in Table 3.2 would amount to some 422 million p.a.¹⁸: note that these are direct costs and exclude any indirect costs.

3.4 The RPA cost assessment for the UK

RPA (2001) estimate the costs of UK compliance with REACH¹⁹. Direct comparison with the RPA-SS study is difficult because RPA (2001) use a 6 per cent discount rate

¹⁷ The deadlines in the 2002 Draft of the White Paper were 2005, 2008 and 2012 respectively. CEC (2003a) thus adds an estimated 3-4 years to these dates if implemented in 2005.

¹⁸ For a discount rate of 3 per cent over 10 years, divide the PV by 8.53.

¹⁹ The RPA study is billed as a Regulatory Impact Assessment, but RIAs are mandated to include costs and benefit. There is only a very limited benefit assessment in the RPA study and this is limited to

(the rate then recommended by HM Treasury) and covers the period 2001-2020 rather than 2001-2012 as in the EU study. We make some crude adjustments to put the estimates on to a similar footing to the RPA study for the EU. The RPA cost estimates need to be compared to a baseline. The baseline could be the current legislation of the evolving Chemicals Strategy for the UK. In order to maintain comparison with the EU study, we choose current legislation as the baseline, although it can be argued that the Chemicals Strategy represents what the industry is committed to by way of costs. Using current legislation as the baseline case, the present value of compliance costs is £513 million or some 770 million. This is at a 6 per cent discount rate and over 20 years. If a 3 per cent discount rate had been used, the effect would be to raise costs by some 30 per cent to almost exactly 1 billion.

The exact status of these cost estimates is uncertain since a further RIA on the costs of REACH to the UK is to be prepared in 2003 (DEFRA, 2003). There is also to be a UK government position paper (DEFRA, 2003). Until this new work is completed, it is not possible to identify the precision of the RPA cost estimates for the UK. The limited coverage of these costs has already been noted and we do not anticipate that this coverage will improve much under the new proposed work, a reflection on the state of modelling of the chemicals sector.

3.5 Indirect costs estimates for the EU

CEC (2003c) — a Draft Impact Assessment - suggests *additional* costs for REACH of 14-26 billion across the period to 2020 and (presumably) in present value terms. Note that the time horizon for the cost estimates has changed from the one used for the direct cost estimates discussed above (2020 has been used for the indirect cost estimates compared to 2012 for the direct cost estimates). The procedure for estimating these indirect costs is only briefly outlined in CEC (2003c) so it is difficult to assess their validity²⁰. The cost estimate rests on a prediction that 8-12 per cent of chemical substances would be withdrawn from the market. Two effects follow. First, the price of chemicals will increase across the board, with consequent welfare losses for users. Second, users will have to switch into higher cost chemicals, where higher cost refers to cost-per-unit effectiveness²¹. An internal calibrated microeconomic model is then used to estimate the cost impacts, but we have been unable to identify what this model is. The model apparently does not account for capacity constraints (which would delay the adoption of substitutes, thus raising costs further), but does suggest that innovation effects will be moderate in the short-run.

If correct, a central estimate of the present value of the costs of REACH would appear to be:

3.6 billion direct costs +
20.0 billion indirect costs

making a total of 23.6 billion (taking the average of the range for indirect costs).

occupational effects of chemicals. How far occupational effects should be included in a cost-benefit appraisal is discussed in Chapters 4 and 5.

²⁰ The estimates are apparently contained in RPA (2003) which is the Phase II study of the RPA-Statistics Sweden (2002) study, but RPA (2003) was not available at the time of writing.

²¹ i.e. substitute chemicals may be just as effective as the original ones, but with higher unit cost; or be similarly expensive but with lower effectiveness.

3.6 Commentary on the cost estimates

Several issues arise with the cost estimates described in Sections 3.3 -3.5.

First, since RPA (2003c) is not available at the time of writing, it is hard to know if the indirect cost estimates for the EU are justified.

Second, the RPA-SS estimates for the *direct* costs for the EU cover a ten year time horizon. This is because of the original timetable suggested by the Commission, i.e. all registrations ending in 2012. However, a correct analysis would extend the time horizon to 20-30 years (as in the RPA UK study and, apparently, in the RPA (2003c) study for the EU) since the with/without principle of cost benefit analysis requires that effort be made to see if costs extend beyond the ten-year period. The rationale for stopping in year 10 is that all chemicals registered after that period will be new chemicals, and the costs of the system should not be significantly different to the current legislation. All existing chemicals should be registered by 2012 had the original timetable been adhered to. However, there may be additional costs in the second period, and we have no way of knowing the extent of any under-estimation of costs arising from this possibility. It is possible that RPA (2003c) has made this adjustment.

Third, as Korzinek et al (2003) note, the compliance costs of REACH may be partly offset by reduced liability provisions for complying parties. That is, companies are likely to face lower risks in terms of occupational and public liability as the risk profile of chemicals improves. This should eventually show up in reduced insurance premia. While this argument is correct as it stands, it must be used with considerable care in a cost-benefit framework. This is because the reduced risk liabilities will show up as *benefits* in a full cost-benefit study. If the benefits are properly recorded, adding in cost savings due to reduced liability would be double counting.

As implied above, we have no way of knowing if the wider costs have been estimated correctly. The obvious starting point for an analysis is to compare the correct notion of cost with what is reported in the statistics. What gets reported tends to be expenditures that industry regards as being due to environmental legislation. But these can obviously differ from true economic costs for various reasons. Economic cost would be measured by the change in the combined sum of producer's and consumer's surplus²². Any lost consumer surplus element is obviously omitted by industrial reporting. Changes in producer's surplus may also be problematic. If the expenditure takes the form of capital equipment there may be some negative effect on other capital investments. Private environmental expenditure could compete for limited capital funds at the corporate level. Hence some analysts regard the true cost of environmental expenditure as involving forgone long run profitability and economic growth due to these crowding out effects. Once again, there is little we can do in the current context to give some quantitative insight into the proper way of estimating compliance costs.

²² Producer's surplus is, roughly speaking, the excess of producer receipts over costs of production. Consumer's surplus is the excess of willingness to pay over the actual price. Both will change in response to a regulation and it is the change in the combined sum that measures the economist's concept of cost which, in turn, is the concept relevant to cost-benefit analysis.

The history of estimating legislative costs is that they can be very inaccurate (for a detailed analysis see Pearce and Palmer, 2001). Unfortunately, we have only a limited idea of the direction of the inaccuracy. There appear to be no exercises testing for the data reliability of environmental expenditure outside the US. The US studies are enabled by the collection of reasonably consistent and regular data on pollution control expenditures using the PACE system (Pollution Abatement and Control Expenditures) by the US Census Bureau. But even the US studies relate only to private, corporate expenditures and produce ambiguous answers. For example, Joshi et al. (2000) suggest that environmental expenditures in the US steel industry are grossly *underestimated* by a factor of around 10. On the other hand, Morgenstern *et al.* (1997; 2001) suggest that, for a wide range of manufacturing industries, reported costs are *overestimates* of true costs. The standard reason for supposing that recorded costs overstate true costs is innovation (Korzinek et al. 2003). The regulatory costs provide an incentive to innovate and innovation lowers costs. The mechanisms by which this happens are several. First, mandated expenditures may raise awareness within the corporation about ways of saving costs. This is more likely to be the case when regulations permit process changes rather than add-on abatement equipment (Morgenstern et al, 1997; 2001). In the case of REACH, we may surmise that process change is the more relevant scenario. Potentially more significant, and emphasised in the literature on corporate environmental management, is the complementarity between profit and environmental expenditure in contexts where firms are operating with some financial slack. The most famous example of this view is attributed to Porter (1990, 1991) and Porter and van der Linde (1995a, 1995b). The Porter hypothesis is not clear-cut, but is generally taken to imply that firms are not operating at full efficiency and that some form of regulation acts as a catalyst which makes firms realise more productive potential through resource efficiency. This is the familiar win-win argument in the corporate environmental literature. Economists have, however, been very sceptical of the Porter hypothesis. For example, whereas Porter and van der Linde (1995a) cite case studies to support their propositions, Palmer et al.(1995) surveyed the same corporations, and others, and found that they generally regarded the adopted clean technology as imposing a net cost on them, not a net benefit. The corporate environmental accounting literature has tended to suggest some balance of effects, i.e. the wider costs and the offsetting gains that may accrue (Schaltegger and Burritt, 2000). If so, we have no reason to suppose that recorded costs under or over-state true costs, but the issue is far from resolved.

Harrington et al. (2000) conduct a meta-analysis of regulatory costs in the US. They note that theory does not provide a clear indication of the expected direction of bias, i.e. there are reasonable explanations to expect both under- and over-estimation. Their approach is to compare *ex ante* estimates of costs with *ex post* estimates, with the former invariably being an estimate provided by the regulatory authorities and the latter being provided by independent experts. They found that, out of 28 regulations studied, 15 of the *ex ante* studies *over-estimated total* costs, and eight either *under-estimated or estimated correctly*. The remaining five were inconclusive. In the case of *unit* costs, 14 overestimated, six underestimated and eight were accurate. Overestimation of compliance costs was also found to be more likely the larger the regulation, i.e. the greater the total cost. Harrington et al. (2000) conclude that, on balance, over-estimation is more likely than under-estimation. However, translating these findings to the REACH context is problematic. First, all the regulations in the

Harrington et al. study are from the US. Second, some of the under-estimation arises from not achieving the intended regulatory target. Thus, the estimated effects of the legislation, in terms of some quantity reduction, were over-estimated for nine regulations. The significance of this latter point is that total costs could be under-estimated because of a failure to achieve the targeted reduction in quantity, i.e. effective non-compliance.

The debate surrounding REACH tends to divide industrial estimates of compliance costs (e.g. Arthur D Little, 2002, for Germany) which tend to emphasise high costs, and NGO/governmental estimates of costs which tend to emphasise low costs due to innovation (e.g. German Federal Environment Agency, 2003; Korzinek et al. 2003). While the Harrington et al work might be taken as evidence that compliance costs will be over-estimated, the reality is that there are no rigorous basis for this belief and the extent of over or under estimation is not known.

Overall, then, the only cost estimates available appear to be those of RPA-Statistics Sweden (2002) for the EU as a whole (direct costs), RPA (2001) for the UK (direct costs only) and RPA (2003c) for the EU (indirect costs, but with the document not being publicly available). Unfortunately we cannot determine the scale of any bias, nor, strictly, even its direction with any certainty. The issue is very much one of the uncertain additional costs that arise from general equilibrium effects versus uncertain additional cost savings arising from the stimulus to innovation²³. We are unclear how far the former have been captured in RPA (2003c). CEC (2003c) suggests the latter effects are modest. In the absence of better information, we adopt the RPA estimates.

²³ We have deliberately refrained from a discussion of other alleged cost impacts of regulation — e.g. that it reduces productivity and competitiveness and that it induces industrial out-migration. Effects on productivity are self-evidently negative, but entirely miss the point that productivity is itself being wrongly measured (Pearce, 2002). It is hard to know what is meant by competitiveness in an international trading regime with flexible exchange rates, and there is no evidence that environmental regulation induces out-migration. For a comprehensive survey see Pearce and Seccombe-Hett (2000).

4 MODEL I — DALYs AND HEALTHCARE COSTS

4.1 The concept of a DALY

The first model aimed at approximating the benefits of REACH makes use of the concept of a disability-adjusted life year or DALY. Annex 4.1 to this chapter explains in more detail how DALYs are calculated. DALYs measure the amount of healthy life lost and permit the aggregation of years lost due to premature mortality, and years of life spent suffering disease. While not free from criticism, the DALY concept has been widely used. The original estimates were sponsored by the World Health Organisation and detailed estimates and projections of DALYs for world regions (but not, unfortunately, for individual countries) are available in the publication *The Global Burden of Disease* (Murray and Lopez, 1996). These estimates can also be found on the World Bank website (www.worldbank.org).

4.2 Model I

We make use of the DALY concept as follows:

- (a) we take estimates of DALYs for Established Market Economies (EMEs) from the WHO/World Bank database;
- (b) we calculate the number of DALYs *per capita* for the EME region;
- (c) we apply the per capita DALY number to the UK and the EU and multiply by the relevant populations to secure total DALYs for the UK and EU;
- (d) we adopt World Bank estimates of the fraction of DALYs in EME countries judged to be due to agro-industrial pollution, with a low estimate of 0.6 per cent and a high estimate of 2.5 per cent (Lvovsky, 2001) - DALYs lost due to agro-industrial pollution are construed to be due to exposure to chemicals;
- (e) we make a judgement as to what fraction of this agro-industrial pollution exposure will be reduced by REACH, to give an estimate of DALYs reduced or avoided by REACH;
- (f) we then adopt differing procedures for valuing DALYs, the first of these — Model I — being health service costs.

Since the detailed workings are involved, we give an example for a single year only. Annex 4.2 sets out the relevant numbers. Accounting for future years involves estimating DALYs for those years and then valuing them at health service costs which themselves will be rising through time due to real cost increases in health service expenditures. Time also has to be allowed for through the process of discounting future cost savings, just as the costs of REACH were discounted (see Chapter 3). Consider the case of the UK. For 2003 we estimate that DALYs lost to males and females (DALYs are separately estimated for males and females in the WHO procedure) amount to 7.13 million. Hence (0.6 per cent - 2.5 per cent) of these DALYs are due to agro-industrial pollution, or approximately 43,000 to 178,000 DALYs. We estimate that the UK Health Service spends 5624 per DALY so that for each DALY reduced this sum would be saved. The unit cost of a DALY is estimated by taking total Health Service Expenditure and dividing it by total DALYs. This procedure is obviously crude because the relevant illnesses due to exposure to

chemicals need not cost the same as the average per DALY level of overall expenditure, but data limitations preclude a more sophisticated approach. The total cost due to chemical exposure in 2003 is thus around 250-1000 million.

This process is repeated for each year, allowing for escalating health service costs and on the assumption that the fraction of total DALYs due to chemical exposure remains the same over time. The estimates are then discounted at the same rate as costs (see Chapter 3) and summed to obtain a present value. Our estimate of the cost of *total exposure* to chemicals is then:

3,730 million - 15,550 million, or 3.73 billion - 15.55 billion.

Note that we adopt the fractions proposed by Lvovsky (2001) from Murray and Lopez (1996) for the burden of disease from agro-industrial pollution, i.e. 0.6 per cent to 2.5 per cent of all cause DALYs in established market economies. We refer to this as chemically induced DALYs, bearing in mind that the chemicals in question cover many sources of pollution and DALYs may include losses arising from cumulated stocks of chemicals in soils etc. and which cannot be affected by the REACH regulation. Since we have no particular basis to suggest that these fractions will change with time, we assume they apply in each year.

4.3 Estimating the change in exposure

Recall that the resulting social cost is (a) for health only, (b) for all chemicals exposure. Some assumption needs to be made about the effect of REACH on exposure to chemicals. This is complex. Some chemicals will go out of production because of REACH. RPA-Statistics Sweden (2002) make some estimates of these effects. However, there appear to be no attempt to estimate how chemical producers and users will react beyond this. For example, users may well switch into other chemicals if one is withdrawn. What matters for the cost-benefit analysis is the change in *exposure* (ΔX) rather than the change in the number of chemicals on the market. In the absence of data on behavioural response, we assume exposure change is proportional to the level of registrations. On the assumption that the regulation achieves high compliance, a low level of registrations of chemicals generally means that there are high levels of withdrawals (rationalisation in the language of RPA-Statistics Sweden). Conversely, high levels of registration mean low levels of withdrawal. The resulting scenarios are shown in Table 4.1.

On the basis of these estimates we could take as a maximum effect scenario, a range of say 30-50 per cent reduction in exposure due to REACH. Our judgement is that this is extremely high because it fails to account for absolute levels of chemical production and usage, simply being the change in the number of registrations. As other

Table 4.1 RPA implied estimates of change in exposure due to REACH

Scenario	RPA-Statistics Sweden Full registrations (including intermediates and unintended uses)	Change in exposure relative to low withdrawals
Low registration = high withdrawals	57,285	-54%
Mid range registration	85,059	-32%
High level registration = low withdrawals	125,735	0

Source: RPA-Statistics Sweden, Table 1. Note: the proportions are similar if intermediates and unintended uses are omitted, at -27 per cent and 48 per cent respectively.

(registered) chemicals are substituted for withdrawn chemicals, exposure would be affected only to a limited extent. Accordingly, we first consider what would happen with a 10 per cent reduction in exposure and then estimate the change in exposure level that would make costs equal benefits, i.e. a switchover point. The 10 per cent assumption can obviously be changed if it is judged that REACH will have less/greater effects.

Chapter 3 indicated that we cannot be certain of the estimates of the cost of compliance suggested by the Commission. Nonetheless, we have no way of obtaining comprehensive general equilibrium estimates so we adopt those estimates here. Table 4.2 summarises the resulting comparison of costs and benefits.

Table 4.2 Comparison of REACH costs and benefits using DALYs and healthcare costs only (present value, billion, $s = 3\%$)

	Total cost of chemical induced DALYs, 2006- 20 PV, 10^9	Benefit of REACH assuming 10% reduction in total exposure PV, 10^9	Cost of complying with REACH PV, 10^9	Benefit minus cost ¹	Worst case break - even level of reduced exposure ²
UK	3.73 — 15.55	0.37 - 1.56	>1.00	Less than -0.63 to 0.56	27.0%
EU	48.34 — 201.43	4.83 - 20.14	3.6 direct costs only 23.6 all costs	1.2 to 16.5 - 3.5 to -18.8	7.5% 49.1%

Notes: 1 — note that the range of estimates of benefits is due to differing assumptions about the overall importance of chemicals exposure as a source of DALYs, whereas the range of compliance costs is due to differing assumptions about chemicals withdrawal. We have assumed that high compliance costs are associated with high benefits, but it is possible to make alternative assumption. For example, high

compliance costs may reflect high registrations and hence low reductions in exposure and hence low benefits. The unknown substitution effects of registered for withdrawn chemicals makes it difficult to be more precise. 2 — i.e. what level of reduced exposure is required for benefits to be guaranteed to be equal to costs in the worst (high cost) case. For example, in the EU worst case, benefits are 4.8 and costs 3.6, so exposure needs to be reduced by $(3.6/4.8) \times 10\% = 7.5\%$.

4.5 Results for Model I

Table 4.2 suggests that REACH would not pass a cost-benefit test. Benefits exceed costs for the EU as a whole if direct costs only are considered. Once the wider costs are allowed for, there are systematic net costs to REACH. However, we would remain optimistic that this comparison understates the likelihood of net benefits. The reasons for optimism are:

- (a) the adoption of what we judge to be a fairly low assumption about reduced exposure, although figures of 8-11 per cent have been quoted in some sources but without substantiation;
- (b) the use of health care costs only, which we know significantly understate true *health* costs (which should be based on avoided care costs plus willingness to pay to avoid illness and premature mortality);
- (c) the omission of any environmental benefits.

The cautions are:

- (a) the use of highly aggregated data (which nonetheless come from detailed bottom up DALY estimates);
- (b) the possibility that costs are underestimated;
- (c) the unknown effects of REACH on exposure levels; and
- (d) the possibility that benefits are over-estimated because the DALY approach does not distinguish occupational from public health risks. The significance of this point is explained in Chapter 5, Section 5.1 where we note an argument that suggests occupational effects should be excluded from a cost-benefit appraisal.

Overall, our judgment is that, while on the basis of Model I, REACH fails a cost-benefit test, Model I is best treated as a worst case benchmark. We believe that the omitted benefits would very probably result in benefit being greater than costs.

Annex 4.1 : CALCULATING DALYs

DALYs are one of several indices that have been used over the years to compare health states. Taking the severest health state to be death and perfect health to be its other extreme a scale 0 1 can be established with which to weight those health states and compare them. The widespread use of DALYs owes most to the Global Burden of Disease (GBD) study which began in 1992. Seminal publications from this programme are Murray and Lopez (1996) and World Bank (1993). There are five components of a DALY.

1. Duration of time lost due to a death at a given age. Taking maximum life expectancies as 82.5 years for women and 80 years for men, a man dying at age x would then have a value for this time lost of $80-x$.
2. The disability (or quality of life) weights, D . Expert assessments are used to assign weights between 0 (perfect health) and 1 (death).
3. An age weighting function indicates the relative importance of healthy life at different ages. This function is again based on surveys of experts and others and produces the initially surprising result that respondents prefer to save young adults rather than children. One reason for this is that the weightings reflect valuations of others' lives, not necessarily the respondent's own life. Much depends on how the surveys were conducted. The relevant valuation in cost-benefit analysis, for example, would be the value of one's own life plus, to some extent, others' valuation of that life. The effect is a function that takes the form:

$$W = C \cdot x \cdot e^{-\beta x} \quad [A4.1]$$

Equation [1] produces a zero weight for age 0, rising to a maximum around 25 and declining thereafter. The ratio of weighting for the 25 age group is three times that of someone aged 80. C is a constant and equals 0.16243, $\beta = 0.04$ and x is age.

4. The discount function:

$$e^{-r(x-a)} \quad [A4.2]$$

where r is the discount rate and is set at 3% (0.03), and a is the year of the onset of disease.

5. The additivity assumption: health is added across individuals so that two people each losing 10 DALYs is the same as one person losing 20 DALYs.

Two equations emerge from these five elements:

$$DALY(x) = D \cdot (C \cdot x^{-\beta x}) \cdot (e^{-r(x-a)}) \quad [A4.3]$$

and

$$DALY = - \left[\frac{D \cdot (C \cdot e^{-\beta a})}{(\beta + r)^2} \right] \left[\{ e^{-(\beta+r)L} [1 + \beta + r](L + a) \} - [1 + (r + \beta)a] \right] \quad [A4.4]$$

The first equation is the number of DALYs lost due to a disability at age x. The second (rather formidable) equation gives the total number of DALYs lost from the onset of disability (x=a) to the age of death (x=a+L). Equation [A4.4] is the integral of equation [3] between x=a and L=a.

Consider an example of a female child that contracts a fatal disease at age five. She has five years of life, and there is a loss of 77.95 years which is equal to an assumed maximum life expectancy of 82.95 years minus the five years of life. The relevant figures are:

C = 0.16243, set by the fitted equation relating to weighting function agreed by experts

D = 1

r = 0.03, i.e. 3%

β = 0.04, set by experts

a = 5, the year of death

L = 77.95, the remaining years of life had death not occurred

Substituting in equation 4, gives:

$$DALY = - \left[\frac{0.16243}{0.0049} \right] \left[\{ e^{-5.4565} (6.8065) \} - 1.35 \right] \quad [A4.5]$$

of 35.85 years.

Annex 4.2 DATA AND ESTIMATES FOR MODEL I

TABLE 1: Chemically induced healthcare costs and avoided costs of REACH for the UK
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	NHS Expenditure ²⁴	CPI (base year 2000) ²⁵	Deflated NHS Expenditure	NHS Expenditure Discounted at 3% (from 2003)	Chemically Induced NHS Expenditure (assumed 0.6%)	Chemically Induced NHS Expenditure (assumed 2.5%)	NHS cost per DALY
2006	40264710.00	193.70	46814371.71	42841781.80889	257,050.69	1,071,044.55	6,015.76
2007	40319240.00	198.80	48112034.29	42746919.31173	256,481.52	1,068,672.98	6,004.63
2008	40373770.00	203.90	49413035.43	42624118.42688	255,744.71	1,065,602.96	5,989.57
2009	40428300.00	209.00	50717375.15	42475003.22849	254,850.02	1,061,875.08	5,970.80
2010	40482830.00	214.10	52025053.44	42301129.32874	253,806.78	1,057,528.23	5,948.54
2011	40537360.00	219.20	53336070.30	42103986.41693	252,623.92	1,052,599.66	5,922.99
2012	40591890.00	224.30	54650425.73	41885000.71093	251,310.00	1,047,125.02	5,894.34
2013	40646420.00	229.40	55968119.74	41645537.32369	249,873.22	1,041,138.43	5,862.79
2014	40700950.00	234.50	57289152.31	41386902.54772	248,321.42	1,034,672.56	5,828.51
2015	40755480.00	239.60	58613523.46	41110346.06023	246,662.08	1,027,758.65	5,791.69
2016	40810010.00	244.70	59941233.18	40817063.05154	244,902.38	1,020,426.58	5,752.48
2017	40864540.00	249.80	61272281.46	40508196.27937	243,049.18	1,012,704.91	5,711.05
2018	40919070.00	254.90	62606668.33	40184838.05132	241,109.03	1,004,620.95	5,667.54
2019	40973600.00	260.00	63944393.76	39848032.13815	239,088.19	996,200.80	5,622.10
2020	41028130.00	265.10	65285457.76	39498775.61997	236,992.65	987,469.39	5,574.88
Total for Years 2006-2020					3,731,865.78	15,549,440.76	

²⁴ Extrapolations of NHS expenditure are based on average annual percentage (%) change, using 70 per cent of 1996-1998 figures on gross NHS expenditure (Source: Department of Health; www.doh.gov.uk).

²⁵ UK Consumer Price Index (Source: www.statistics.gov.uk).

TABLE 2: Chemically induced healthcare costs and avoided costs of REACH for the EU
(Figures in thousands of euros. The benefits from the implementation of REACH start in 2006.)

Year	NHS Expenditure	CPI (base year 2000) ²⁶	Deflated NHS Expenditure	NHS Expenditure Discounted at 3% (from 2003)	Chemically Induced NHS Expenditure (assumed 0.6%)	Chemically Induced NHS Expenditure (assumed 2.5%)	NHS cost per DALY
2006	603970650	113.1	653675411.6	598205600.9	3,589,233.61	14,955,140.02	13,424.35
2007	604788600	114.6	663241852.2	589281795.4	3,535,690.77	14,732,044.88	13,242.52
2008	605606550	116.1	672831774.7	580390599.3	3,482,343.60	14,509,764.98	13,060.91
2009	606424500	117.6	682445178.9	571537093.4	3,429,222.56	14,288,427.34	12,879.65
2010	607242450	119.1	692082065	562726052.2	3,376,356.31	14,068,151.31	12,698.84
2011	608060400	120.6	701742432.9	553961956.7	3,323,771.74	13,849,048.92	12,518.58
2012	608878350	122.1	711426282.6	545249006.8	3,271,494.04	13,631,225.17	12,338.98
2013	609696300	123.6	721133614.2	536591134.1	3,219,546.80	13,414,778.35	12,160.12
2014	610514250	125.1	730864427.5	527992012.7	3,167,952.08	13,199,800.32	11,982.08
2015	611332200	126.6	740618722.7	519455071	3,116,730.43	12,986,376.77	11,804.96
2016	612150150	128.1	750396499.7	510983502	3,065,901.01	12,774,587.55	11,628.83
2017	612968100	129.6	760197758.5	502580274.1	3,015,481.64	12,564,506.85	11,453.76
2018	613786050	131.1	770022499.1	494248140.8	2,965,488.84	12,356,203.52	11,279.81
2019	614604000	132.6	779870721.5	485989650.5	2,915,937.90	12,149,741.26	11,107.05
2020	615421950	134.1	789742425.8	477807155.6	2,866,842.93	11,945,178.89	10,935.55
Total for Years 2006-2020					48,341,994.27	201,424,976.14	

²⁶ Harmonised Index of Consumer Prices; Source: www.statistics.gov.uk. The harmonised index of consumer prices (HICP) is an internationally comparable measure of inflation, calculated by each Member State of the European Union. HICPs are used to compare inflation rates across the European Union. Since January 1999, they have been used by the European Central Bank as the target measure of inflation for the Member States of the Eurozone. Increasingly, HICPs are being used for indexing contracts, which cover more than one EU Member States.

5 MODEL II — DALYs AND WILLINGNESS TO PAY

5.1 CBA and willingness to pay

Model I valued DALYs in terms of the healthcare costs of preventing DALYs. This approach seriously understates the true cost of premature mortality and ill-health. The reason for this is that healthcare costs refer only to the resources that are allocated by the state to curing, ameliorating or preventing ill-health. Cost-benefit analysis adopts as its basic value judgement the notion that what determines value is the preferences of the individual. In turn, those preferences show up in various ways. Voting would be one manifestation of preferences, but voting rarely takes place on the sufficiently widespread or detail basis that would be needed to evaluate individual decisions. One context in which votes are recorded every second of the day is the marketplace, and willingness to pay (WTP) is the means by which preferences are revealed. For any good costing X , a purchase signals that the purchaser has a WTP equal to or exceeding X , and a non-purchase signals that $WTP < X$. Those with a WTP in excess of X are actually getting something for nothing since, had the good been priced more highly, they would still have bought it. This excess of WTP over price is the consumer's surplus and is tantamount to a net benefit received by the consumer. The sum of these consumer's surpluses gives the measure of aggregate benefit of supplying the good²⁷.

In the context of chemicals (and many other environmental and risk reducing goods) the obvious problem is that health risks appear not to be traded in the market place, and environmental risks almost certainly are not traded. Hence the context is one of non-markets rather than markets. Environmental economists have developed an extensive range of techniques for eliciting WTP in non-market contexts. Space forbids any discussion here and the reader is referred to Freeman (2003). While the analysis assumes health and environmental risks are not traded in markets, one caveat is in order. REACH affects two forms of health risk: occupational and public. It is reasonable to assume that public health risks are genuinely non-marketed, but the same cannot be said conclusively about occupational health risks. One of the techniques used to place a money value on health risks is the hedonic wage model. Stripped of its complexities, the model argues that wage rates are a function of many variables relating to the characteristics of the wage-earner (age, skills etc.) and the characteristics of the workplace (degree of unionisation, nature of the job etc.). One of the workplace characteristics is occupational risk, and hedonic wage models have shown fairly conclusively that risk is embedded or internalised in wage rates. What this means is that workers are already at least partially compensated for being exposed to occupational risk. On this argument, adding the value of any reduction in these risks to other values (e.g. public health risks) amounts to double-counting. On

²⁷ There is a parallel notion for producers, namely the excess of received price over the price at which producers would have been willing to supply — this is producer's surplus and this too needs to be aggregated as a benefit of supplying the good. The sum of consumer's and producer's surplus is then the benefit recorded in a CBA.

this form of the argument, then, it would not be correct to regard reductions in occupational risk as a benefit to REACH. Only reductions in public health risks would matter.

How valid is this argument in the context of REACH? Again it is a moot point. Existing CBA studies of air pollution tend to work on the assumption that all pollution is external to the workplace, so that only public health risks are evaluated. The absence of occupational risk assessments thus tells us little or nothing about the validity of including occupational effects. Moreover, REACH is quite different in its intentions to standard policies of reducing air pollution: it is designed to affect occupational risk as well as public health risk. Some advance might be made if the relevant hedonic wage studies include risks from exposure to chemicals. For example, if they showed that wages were higher (other things being equal) in occupations with higher exposure to chemical risks, then the internalisation hypothesis would have some validity and it could be questioned as to whether occupational health benefits should be included in a CBA of REACH. Unfortunately, the hedonic wage studies tend to focus on fatality risks, i.e. injuries and accidents where it is fairly simple to relate occupational activity to the risk (Day, 1998). Few studies look at non-fatal risks, and it appears that none attributes fatalities to chronic exposure to the workplace environment²⁸. Accordingly, we cannot say whether exposure to workplace chemicals is a risk that is or is not internalised in the wage rate. We need to bear in mind the caution that the inclusion of occupational risks may overstate the true benefits of REACH.

Finally, we note that RPA (2003b), a report not publicly available at the time of writing, has estimated occupational benefits from REACH at 18-30 billion over a 30 year period, using a value of life saved approach²⁹. We are unable to comment on these estimates because of the non-availability of the document for public scrutiny. We note below some of the problems of valuing lives saved.

5.2 Valuing premature mortality

It is easy to become confused by the notion of valuing life. What in fact is valued is a change in the risk of fatality. Let this willingness to pay for a small change in risk (Δr) be given by WTP_i where i is the i th person. Then the value of a statistical life (VOSL) is given by:

$$VOSL = \frac{\sum_i WTP_i}{\Delta r \cdot N} \quad [5.1]$$

where N is population at risk. In other words, VOSL is convenient shorthand for an aggregate valuation of a change in risk affecting a given population.

²⁸ This does not mean chronic exposure has not been studied. It has, for example for asbestosis and for other risks. The point here is that the hedonic wage studies tend not to focus on these cases.

²⁹ As reported in CEC (2003c). We note that RPA (2003b) appears to cover a 30 year time horizon but RPA(2003a), on indirect compliance costs, covers a 20 year time period, and RPA (2001) on direct compliance costs covers a 10 year time period!

Estimates of VOSL vary. In the UK, an official VOSL of some £1 million = 1.67 million is used. Other European studies, e.g. Olsthoorn et al (1999) have used higher VOSLs for Europe of 3.2 million (at 1995 prices, which suggests a value of around 3.7 million at 2000 prices). We opt for the lower UK value here in order to be conservative in our estimates. Moreover, there is a continuing debate about the correct measure of risk valuation. In the case of chemicals, for example, the issue is one of some acute exposures and other chronic exposures. Valuing acute effects could involve the VOSL concept. For example, an acute death at, say, the age of 40 would involve some 40 forgone life-years and a value such as £1 million appears appropriate. There is (surprisingly) only limited evidence on how VOSL varies with age (i.e. how WTP to avoid risk varies with age) and it is currently thought that VOSL declines with age beyond a certain point (for a discussion see Pearce, 2000). For chronic exposure, however, the issue is one of morbidity if chemicals induce ill-health before death, and premature mortality. The epidemiology of chronic exposure to air pollutants is still weak, but it is thought that, in Europe, exposure reduces life expectancy by around 6 months (Kunzli *et al.* 2000). If so, the relevant valuation is the willingness to pay of those at risk to avoid this reduction in their life span. Unfortunately, few studies have attempted to estimate this. In so far as they have, the results imply very much more modest valuations than those observed from the literature estimating WTP to reduce contemporaneous risks³⁰.

5.3 Valuing morbidity

Chemicals also involve ill-health independently of any premature mortality. WTP studies exist for states of ill-health. However, the procedure adopted here is to forge the following links:

- (a) the WTP for a change in risk of fatality, i.e. the VOSL;
- (b) an equivalence between DALYs and premature life lost;
- (c) a corresponding value of a DALY.

Lvovsky et al (2000) conduct an analysis of this kind for six cities — Mumbai, Shanghai, Manila, Krakow, Bangkok and Santiago (Chile). They anchor their values on a VOSL of \$US 1.62 million (in 1990 prices, i.e. some \$2.4 million in 2003 prices). (The official VOSL in the UK is UK£1 million, i.e. about US\$ 1.6 million in 2003 prices, well below the Lvovksy estimate). We consider the UK value to be appropriate for Europe and hence retain the US\$1.6 million value but in 2003 prices rather than 1990 prices. Lvovsky et al. then calculate the number of DALYs lost per 10,000 cases for each of several health end-states: premature mortality. Chronic bronchitis etc. On the assumption that one premature death is equivalent to 10 DALYs, this permits them to derive WTP values for each health end-state. For example, chronic bronchitis is equivalent to 0.12 of a premature death (100,000 DALYs are lost per 10,000 cases, and 12,037 DALYs are lost per 10,000 cases). The resulting WTP to avoid all the (ill) health states is then derived. From this overall average WTP values per DALY can be inferred and the effect is shown in Table 5.1. We have then scaled these WTP values up by the ratio of per capita income in Europe and the UK to the countries shown.

³⁰ One reason for this is fairly obvious. Asking someone aged, say, 40 years their WTP to avoid a six months curtailment of their life at, say, age 80, will elicit much lower values than asking for the WTP to avoid a risk of fatality that could occur tomorrow.

Table 5.1 Values of DALYs based on WTP to avoid illness and fatality risks

	Value of DALY, 2003\$	Value of DALY, 2003	Income ratio UK to Country ¹	Income ratio EU to Country	Value of a DALY UK 2003	Value of a DALY EU 2003
Mumbai	3,345	3,040	25.5	47.1	77,520	143,311
Shanghai	7,285	6,622	7.2	13.3	47,628	88,143
Manila	10,594	9,630	11.1	20.5	106,893	197,613
Bangkok	24,000	21,818	4.6	8.7	100,363	188,767
Krakow	20,162	18,329	5.0	9.2	91,645	169,424
Santiago	25,924	23,567	3.1	5.7	73,058	135,062
AVERAGE	11,098	10,089	9.4		94,836	

Notes: 1 - We have adjusted the per capita incomes reported in Lvovsky et al. (2000) to allow for growth between 1990 (the year figures used in that report) and 2003.

Table 5.1 suggests that the appropriate value for a DALY in Europe/UK is around 90,000 per DALY. This can be compared to the health expenditure figure of some 5600 per DALY used in Chapter 4. The ratio is 16.

Recently, efforts have been made to value life years (VOLY) more directly. The ExternE programme has used estimates derived by adjusting VOSL according to the formula:

$$VOSL = VOLY(s) \cdot \sum_{i=a}^T \frac{aP_i}{(1+s)^{i-a}} \quad [5.2]$$

where aP_i is the conditional probability of surviving to year i given that the individual at risk has already survived to year a . a is then the age of the person at risk. Pearce (2000) shows that for a 40 year old, the VOLY would be around 40-50,000 for a VOSL of 1.5 million, or, say, 45-55,000 for a VOSL of 1.67 million. We take an average of 50,000. Pearce (2000) expresses some concerns about the procedure embodied in equation [5.2] but we include the resulting VOLY value derived by this approach and apply this to the notion of a DALY as well.

5.4 Results for Model II

Table 5.2 shows the results for Method II. At the higher value of a DALY, we see that, while there is a small chance of costs exceeding benefits, benefits are more likely to exceed costs and by a significant amount. At the lower value of a DALY the balance is more even between costs exceeding or falling short of benefits. **Overall, Model II suggests to us a more than even chance that benefits will exceed costs.** The caveats and cautions are the same as those listed under Method I.

Table 5.2 Comparison of REACH costs and benefits using DALYs and willingness to pay (present value, billion, $s = 3\%$)

	Total cost of chemical induced DALYs, 2006-20 PV, 10^9	Benefit of REACH assuming 10% reduction in total exposure PV, 10^9	Cost of complying with REACH PV, 10^9	Benefit minus cost ¹	Worst case break - even level of reduced exposure ²
At 90,000 per DALY					
UK	36.0 - 150.0	3.60 - 15.00	>1.00	2.60 - 14.00	n.a.
EU	223.9 - 932.7	22.39 - 93.27	23.6	-1.2 - 69.8	n.a.
At 50,000 per DALY					
UK	19.8 - 82.5	1.98 - 8.25	>1.00	0.98 - 7.25	n.a.
EU	123.1 - 513.0	12.31 - 51.30	23.6	-11.3 - 27.7	n.a.

Notes: 1 — see footnotes to Table 4.2.

One check on the reasonableness of the estimates in Table 5.2 is to consider the ratios of benefits to costs. These are approximately 0.5 to 2 for the lower value of the DALY and 1-4 for the higher value. These are fairly high but not unreasonably high. The US EPA *ex post* evaluation of the US Clean Air Act produced a benefit cost ratio of 44 (US EPA, 1997). EU studies of air pollution control produce benefit-cost ratios of between 3 and 6 (Pearce, 2000). The main cause for concern is, as discussed above, the use of the VOSL estimate of about 1.6 million. The VOSL estimates are the major factor driving the results of the US EPA study³¹ and they are similarly the major factor producing the 90,000 per DALY figure here. If, as many would argue, it is not legitimate to apply the VOSL figure in the context of chronic exposures to chemicals, then we might expect the substantial benefit-cost ratios shown here to be an exaggeration.

One other check is possible. The health expenditure cost estimates are, in our view, fairly robust. Chapter 4 showed that REACH could pass a cost-benefit test even on that basis alone. Yet we know WTP exceeds health expenditure costs. If we knew the ratio of WTP to health expenditure costs, we could cast some light on the ratio derived here of 16:1. Unfortunately, few studies estimate both WTP and health care costs. Rowe *et al.* (1995) adopt a value based on the US costs of treating cancers) and then multiply this by 1.5 on the basis that, where healthcare cost and WTP studies are available, WTP appears to be 1.5 times the COI. This procedure is clearly not satisfactory, as there are few studies that estimate COI and WTP. Moreover, the Rowe

³¹ The US EPA study used a VOSL of \$4.8 million.

et al. COI value dates from the mid-1970s. Clearly, if their ratio was applicable, Model II would yield only a 50 per cent increase in the benefits assessed in Model I. On that basis, however, benefits would stand more chance of exceeding costs for the EU.

Annex 5.1 DATA AND ESTIMATES FOR MODEL II

TABLE 1: Willingness to Pay estimates for the UK, where one DALY is equal to 90,000 euros
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Years	P_DALY	P_DALY Discounted at 3% (from 2003)	Health Benefits of REACH	
			0.6% Scenario	2.5% Scenario
2006	92,727.09	84,858.42	331,827.08	1,382,612.84
2007	93,654.36	83,210.69	315,791.13	1,315,796.37
2008	94,590.90	81,594.95	300,530.09	1,252,208.71
2009	95,536.81	80,010.58	286,006.53	1,191,693.86
2010	96,492.18	78,456.97	272,184.80	1,134,103.32
2011	97,457.10	76,933.54	259,030.99	1,079,295.80
2012	98,431.67	75,439.68	246,512.83	1,027,136.81
2013	99,415.99	73,974.83	234,599.61	977,498.36
2014	100,410.15	72,538.43	223,262.08	930,258.67
2015	101,414.25	71,129.92	212,472.44	885,301.82
2016	102,428.40	69,748.75	202,204.20	842,517.50
2017	103,452.68	68,394.41	192,432.17	801,800.72
2018	104,487.21	67,066.36	183,132.38	763,051.58
2019	105,532.08	65,764.10	174,282.00	726,175.00
2020	106,587.40	64,487.13	165,859.32	691,080.49
	°	°		
Total for Years 2006-2020			3,600,127.64	15,000,531.84

TABLE 1: Willingness to Pay estimates for the EU, where one DALY is equal to 90,000 euros
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Years	P_DALY	P_DALY Discounted at 3% (from 2003)	Health Benefits of REACH	
			0.6% Scenario	2.5% scenario
2006	92,727.09	84,858.42	2,076,308.12	8,651,283.85
2007	93,654.36	83,210.69	1,973,939.82	8,224,749.24
2008	94,590.90	81,594.95	1,876,614.94	7,819,228.90
2009	95,536.81	80,010.58	1,784,085.17	7,433,688.22
2010	96,492.18	78,456.97	1,696,114.44	7,067,143.48
2011	97,457.10	76,933.54	1,612,478.26	6,718,659.40
2012	98,431.67	75,439.68	1,532,963.20	6,387,346.68
2013	99,415.99	73,974.83	1,457,366.35	6,072,359.80
2014	100,410.15	72,538.43	1,385,494.76	5,772,894.84
2015	101,414.25	71,129.92	1,317,164.99	5,488,187.45
2016	102,428.40	69,748.75	1,252,202.61	5,217,510.89
2017	103,452.68	68,394.41	1,190,441.80	4,960,174.17
2018	104,487.21	67,066.36	1,131,724.88	4,715,520.32
2019	105,532.08	65,764.10	1,075,901.93	4,482,924.72
2020	106,587.40	64,487.13	1,022,830.43	4,261,793.46
Total for Years 2006-2020			22,385,631.70	93,273,465.41

6 MODEL III — DISEASE-SPECIFIC HEALTH COSTS

6.1 Methodology for Model III

Models I and II adopted the DALY approach, combined with various assumptions about the total fraction of DALYs that can be attributed to chemicals exposure, and an assumption about the extent to which REACH will reduce exposure to chemicals. Model III proceeds differently and follows the methodology of Muir and Zegarac (2001). They estimate social healthcare costs plus productivity effects of toxic substances in the USA for 1997. They cover:

Diabetes
Parkinson s disease
Neurodevelopmental effects + hyperthyroidism
Deficiencies in IQ

Two types of cost are estimated: medical costs, M , and forgone productivity, Q . Hence the total cost of any disease i , C_i is

$$C_i = M_i + Q_i \quad [6.1]$$

Table 6.1 shows these estimates for the US (adapted from Muir and Zegarac). Note these costs are for all the above diseases regardless of the fraction of them caused by exposure to chemicals. This suggests two approaches for the purposes of evaluating REACH:

- (a) estimate the cost per case implied by the Muir and Segarac figures, adjust for EU/UK incomes and multiply by the number of cases in the EU/UK, or
- (b) assume the incidence is the same in the EU/UK as in the US and adjust for EU incomes.

Approach (b) is easier because we do not have to find estimates of the number of cases in the EU/UK. Some estimates of UK and EU cases are independently available but time has not permitted a detailed search. Table 6.1 shows that there are problems in estimating per case expenditures which also makes it difficult to use approach (a). On approach (b) we use the total expenditure, differentiating between M and Q , and adjusting each of these for EU/UK conditions.

Table 6.1 Social costs of disease in the US, 1997 (1999 prices)

	M \$10 ⁹ 1999	Q \$10 ⁹ 1999	M+Q \$10 ⁹ 1999	Population affected = no of cases 10 ⁶	Cost per case \$ p.a (rounded)
Diabetes	51.2	62.6	113.9	15.8	7,200
Parkinson s	9.3 - 29.9	3.7 - 5.6	38.0 - 53.5	0.4 - 0.6	63,000 - 134,000 (1)
Hyperthyroidism and neuro- development	81.5 - 167.2	n.a.	81.5 - 167.2		Spec.education: 6,477-7,660 Brain disorders: 6,500 Ritalin costs: 250 Autism: 17,700
Loss of IQ due to 5 point decrement	0	275.0 - 327.0	275.0 - 327.0	4.02	68,000 - 81,300
Dynamic economic impacts	0	17.1 - 85.0	17.1 - 85.0	Whole population	
Social impacts	19.1	0	19.1	Whole population	
TOTAL	161.1 - 267.4	358.9 - 480.7	519.0 - 748.1		

Source: Appendix of Muir and Segarac (2001). Data in their appendix differ very slightly from some of the estimates in the main body of the text.

Notes: (1) includes \$25 billion other costs for PD, but Muir and Zegarac exclude this cost in their own final estimates

The procedure is as follows. First, we transfer medical (M) costs (shown for UK but same procedure can be used for EU):

$$M_{UK,j} = \left[\frac{N_{US,j}}{N_{US}} \cdot N_{UK} \right] \left[\frac{M_{US,j}}{N_{US,j}} \right] \left[\frac{(M/N)_{UK}}{(M/N)_{US}} \right] \cdot e \quad [6.2]$$

where M is medical expenditure, N is population, j is jth disease, N_j is no of cases of disease j, etc., e is the exchange rate \$/£ or £/ \$.

The first bracketed expression is the number of estimated cases in the UK and EU assuming the incidence is the same as in the US. The second bracketed expression is the cost per case in the US. The third bracketed expression is a scaling factor to allow for income differences in the two countries .

We then transfer productivity losses as follows:

$$Q_{UK,j} = \left[\frac{Q_{US,j}}{N_{US,j}} \right] \left[\frac{(Y/N)_{UK}}{(Y/N)_{US}} \right] \cdot N_{UK,j} \cdot e \quad [6.3]$$

where Q is output loss, Y is per capita income, N is population, j is the jth disease, e is the exchange rate. The first bracketed expression is output loss per case in the US. The second expression is a scaling factor to account for differences in income. The third expression is the number of estimated cases from [6.2].

These procedures give us total health damage across the relevant diseases and they can be summed to get a total. Muir and Zegarac say that 10-50 per cent of cases are due to environmental toxins, so *for any one year*

$$C_E = (0.1 \quad \text{to} \quad 0.5) \sum_j (M_{UK} + Q_{UK}) \quad [6.4]$$

The benefits from REACH are some fraction, k, of this. We take the same assumption as before, namely that REACH reduces cases by 10 per cent. Then, letting the range 0.1-0.5 in [6.4] equal E for environmentally-induced and allowing for discounting, we have:

$$C_E = k \cdot E \cdot \sum_{j,t} (M_{UK,j,t} + Q_{UK,j,t}) \cdot \delta_t \quad [6.5]$$

Note that the Muir and Zegerac assumptions about the fraction of exposure due to chemicals appear to be very different to that assumed for Models I and II. Models I and II took a range of 0.6 per cent to 2.5 per cent of DALYs due chemicals, whereas the range for the specific diseases covered by Muir and Zegerac is 10-50 per cent. A comparison is not possible because DALYs and their money valuations are quite differently conceptually to the health cost and forgone output approach in Muir and Zegerac. Moreover, we cannot be sure which diseases the World Bank identify as chemically induced in their approach using DALYs. Therefore we present Model III as a separate approach which cannot really be compared with Models I and II.

6.2 Results of Model III

Table 6.2 presents the results for Model III. The category social impacts in the Muir and Zegerac approach has been ignored here because the estimates are very small and do not affect the grand total. We have also taken the lower end of the output-effect assumptions, and the lower end of assumed affected population in order to be conservative. Even on this conservative basis, the benefits of REACH are in the range 15-75 billion for the UK alone and comfortably exceed the likely costs. At the EU level the benefits are in the range 65-283 billion, implying net benefits of some 23-260 billion.

Hence, Model III indicates that the benefits of REACH comfortably exceed the costs of REACH.

Table 6.2 Disease-specific health benefits from REACH

6.2A United Kingdom

10⁹	Diabetes	Parkinson s	IQ Loss	Dynamic economic effects
Total PV of health costs	221.8	13.4	963.5	297.8
Chemically induced health costs (10-50%)	22.2 - 110.9	0.1 - 6.7	96.4 - 481.8	29.8 - 148.9
Benefit of REACH at 10% reduction	2.2 - 11.1	0 - 0.7	9.6 - 48.2	3.0 - 14.9
Grand total of benefits of REACH	14.8 - 74.9			

6.2B The European Union

10⁹	Diabetes	Parkinson s	IQ Loss	Dynamic economic effects
Total PV of health costs	742.8	51.1	3947.9	928.8
Chemically induced health costs (10-50%)	74.3 - 371.4	5.1 - 25.6	394.8 - 1974.0	92.8 - 464.4
Benefit of REACH at 10% reduction	7.4 - 37.1	0.5 - 2.6	39.5 - 197.4	9.3 - 46.4
Grand total of benefits of REACH	56.7 - 283.5			

Annex 6.1 DATA AND ESTIMATES FOR MODEL III

TABLE 1: Medical costs (M) and Output loss (Q) for Diabetes in the UK
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Annual M+Q	Discounted at 3% from 2003	10% Reduction due to REACH	
			10% environmentally Induced	50% environmentally Induced
2006	19,458,195.67	17,807,005.48	178,070.05	890,350.27
2007	19,464,735.73	17,294,165.59	172,941.66	864,708.28
2008	19,478,036.90	16,801,925.73	168,019.26	840,096.29
2009	19,497,365.83	16,328,736.93	163,287.37	816,436.85
2010	19,521,859.78	15,873,058.47	158,730.58	793,652.92
2011	19,570,836.33	15,449,398.92	154,493.99	772,469.95
2012	19,625,459.40	15,041,280.46	150,412.80	752,064.02
2013	19,684,999.74	14,647,488.52	146,474.89	732,374.43
2014	19,748,857.87	14,266,995.11	142,669.95	713,349.76
2015	19,816,301.74	13,898,755.34	138,987.55	694,937.77
2016	19,886,898.91	13,542,010.46	135,420.10	677,100.52
2017	19,960,193.64	13,196,039.42	131,960.39	659,801.97
2018	20,035,764.45	12,860,194.79	128,601.95	643,009.74
2019	20,113,096.31	12,533,816.67	125,338.17	626,690.83
2020	20,191,947.08	12,216,460.06	122,164.60	610,823.00
Total	296,054,549.38	221,757,331.95	2,217,573.32	11,087,866.60

TABLE 2A: Medical costs (M) and Output loss (Q) for Parkinson s in the UK
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Affected Population 0.4		10% Reduction due to REACH	
	Annual M+Q	Discounted at 3% from 2003	10% environmentally induced	50% environmentally induced
2006	1,176,545.03	1,076,705.37	10,767.05	53,835.27
2007	1,176,931.58	1,045,688.46	10,456.88	52,284.42
2008	1,177,717.75	1,015,909.68	10,159.10	50,795.48
2009	1,178,860.19	987,276.85	9,872.77	49,363.84
2010	1,180,307.92	959,698.35	9,596.98	47,984.92
2011	1,183,202.70	934,031.14	9,340.31	46,701.56
2012	1,186,431.22	909,300.74	9,093.01	45,465.04
2013	1,189,950.38	885,434.84	8,854.35	44,271.74
2014	1,193,724.74	862,372.15	8,623.72	43,118.61
2015	1,197,711.04	840,050.43	8,400.50	42,002.52
2016	1,201,883.72	818,424.33	8,184.24	40,921.22
2017	1,206,215.83	797,450.76	7,974.51	39,872.54
2018	1,210,682.48	777,091.01	7,770.91	38,854.55
2019	1,215,253.21	757,305.62	7,573.06	37,865.28
2020	1,219,913.72	738,067.86	7,380.68	36,903.39
Total	17,895,331.50	13,404,807.59	134,048.08	670,240.38

TABLE 2B: Medical costs (M) and Output loss (Q) for Parkinson s in the UK
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Affected Population 0.6		10% Reduction due to REACH	
	Annual M+Q	Discounted at 3% from 2003	10% environmentally induced	50% environmentally induced
2006	1,797,325.97	1,644,807.87	16,448.08	82,240.39
2007	1,797,911.02	1,597,420.66	15,974.21	79,871.03
2008	1,799,100.90	1,551,920.24	15,519.20	77,596.01
2009	1,800,830.01	1,508,166.78	15,081.67	75,408.34
2010	1,803,021.16	1,466,021.20	14,660.21	73,301.06
2011	1,807,402.45	1,426,780.18	14,267.80	71,339.01
2012	1,812,288.86	1,388,968.51	13,889.69	69,448.43
2013	1,817,615.15	1,352,476.37	13,524.76	67,623.82
2014	1,823,327.70	1,317,210.72	13,172.11	65,860.54
2015	1,829,361.02	1,283,077.01	12,830.77	64,153.85
2016	1,835,676.42	1,250,006.32	12,500.06	62,500.32
2017	1,842,233.14	1,217,933.13	12,179.33	60,896.66
2018	1,848,993.46	1,186,798.55	11,867.99	59,339.93
2019	1,855,911.33	1,156,542.58	11,565.43	57,827.13
2020	1,862,965.07	1,127,124.51	11,271.25	56,356.23
Total	27,333,963.67	20,475,254.64	204,752.55	1,023,762.73

TABLE 3A: Output loss (Q) for Loss of IQ due to 5 point decrement in the UK
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss 275*10 ⁹		10% Reduction due to REACH	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	84,530,685.62	77,357,551.90	773,575.52	3,867,877.60
2007	84,559,415.90	75,129,945.81	751,299.46	3,756,497.29
2008	84,617,847.57	72,992,098.63	729,920.99	3,649,604.93
2009	84,702,759.00	70,937,227.16	709,372.27	3,546,861.36
2010	84,810,360.22	68,958,583.96	689,585.84	3,447,929.20
2011	85,025,512.80	67,119,924.95	671,199.25	3,355,996.25
2012	85,265,470.35	65,348,883.17	653,488.83	3,267,444.16
2013	85,527,029.35	63,640,142.10	636,401.42	3,182,007.11
2014	85,807,556.29	61,989,204.36	619,892.04	3,099,460.22
2015	86,103,835.30	60,391,497.69	603,914.98	3,019,574.88
2016	86,413,966.63	58,843,706.37	588,437.06	2,942,185.32
2017	86,735,948.24	57,342,679.79	573,426.80	2,867,133.99
2018	87,067,928.64	55,885,590.23	558,855.90	2,794,279.51
2019	87,407,645.30	54,469,554.79	544,695.55	2,723,477.74
2020	87,754,034.46	53,092,634.04	530,926.34	2,654,631.70
Total	1,286,329,995.68	963,499,224.95	9,634,992.25	48,174,961.25

TABLE 3B: Output loss (Q) for Loss of IQ due to 5 point decrement in the UK
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss 327*10 ⁹		Reduction due to REACH equal to 10%	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	100,514,669.81	91,985,161.72	919,851.62	4,599,258.09
2007	100,548,832.73	89,336,335.56	893,363.36	4,466,816.78
2008	100,618,313.29	86,794,240.91	867,942.41	4,339,712.05
2009	100,719,280.70	84,350,811.93	843,508.12	4,217,540.60
2010	100,847,228.33	81,998,025.30	819,980.25	4,099,901.26
2011	101,103,064.31	79,811,692.58	798,116.93	3,990,584.63
2012	101,388,395.65	77,705,762.89	777,057.63	3,885,288.14
2013	101,699,413.08	75,673,914.42	756,739.14	3,783,695.72
2014	102,032,985.12	73,710,799.37	737,107.99	3,685,539.97
2015	102,385,287.79	71,810,980.88	718,109.81	3,590,549.04
2016	102,754,062.14	69,970,516.30	699,705.16	3,498,525.82
2017	103,136,927.55	68,185,659.24	681,856.59	3,409,282.96
2018	103,531,682.42	66,453,047.29	664,530.47	3,322,652.36
2019	103,935,636.41	64,769,252.42	647,692.52	3,238,462.62
2020	104,347,524.62	63,131,968.48	631,319.68	3,156,598.42
Total	1,529,563,303.95	1,145,688,169.31	11,456,881.69	57,284,408.47

TABLE 4A: Dynamic Economic Impacts measured in Output loss (Q) in the UK
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss $17.1 \cdot 10^9$		10% Reduction due to REACH	
	Annual Q	Discounted at 3% From 2003	10% environmentally Induced	50% environmentally Induced
2006	5,256,271.72	4,810,233.23	48,102.33	240,511.66
2007	5,258,058.23	4,671,716.63	46,717.17	233,585.83
2008	5,261,691.61	4,538,781.41	45,387.81	226,939.07
2009	5,266,971.56	4,411,005.76	44,110.06	220,550.29
2010	5,273,662.40	4,287,970.13	42,879.70	214,398.51
2011	5,287,040.98	4,173,638.97	41,736.39	208,681.95
2012	5,301,961.97	4,063,512.37	40,635.12	203,175.62
2013	5,318,226.19	3,957,259.75	39,572.60	197,862.99
2014	5,335,669.86	3,854,601.43	38,546.01	192,730.07
2015	5,354,093.03	3,755,253.13	37,552.53	187,762.66
2016	5,373,377.56	3,659,008.65	36,590.09	182,950.43
2017	5,393,398.96	3,565,672.09	35,656.72	178,283.60
2018	5,414,042.11	3,475,067.61	34,750.68	173,753.38
2019	5,435,166.31	3,387,015.95	33,870.16	169,350.80
2020	5,456,705.42	3,301,396.52	33,013.97	165,069.83
Total	79,986,337.91	59,912,133.62	599,121.34	2,995,606.68

TABLE 4B: Dynamic Economic Impacts measured in Output loss (Q) in the UK
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss $85 \cdot 10^9$		10% Reduction due to REACH	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	26,127,666.46	23,910,516.04	239,105.16	1,195,525.80
2007	26,136,546.73	23,221,983.25	232,219.83	1,161,099.16
2008	26,154,607.43	22,561,194.12	225,611.94	1,128,059.71
2009	26,180,852.78	21,926,052.03	219,260.52	1,096,302.60
2010	26,214,111.34	21,314,471.41	213,144.71	1,065,723.57
2011	26,280,613.05	20,746,158.62	207,461.59	1,037,307.93
2012	26,354,781.74	20,198,745.71	201,987.46	1,009,937.29
2013	26,435,627.25	19,670,589.38	196,705.89	983,529.47
2014	26,522,335.58	19,160,299.53	191,603.00	958,014.98
2015	26,613,912.73	18,666,462.92	186,664.63	933,323.15
2016	26,709,771.50	18,188,054.70	181,880.55	909,402.73
2017	26,809,293.09	17,724,101.03	177,241.01	886,205.05
2018	26,911,905.22	17,273,727.89	172,737.28	863,686.39
2019	27,016,908.55	16,836,044.21	168,360.44	841,802.21
2020	27,123,974.29	16,410,450.52	164,104.51	820,522.53
Total	397,592,907.75	297,808,851.35	2,978,088.51	14,890,442.57

TABLE 5: Social Impacts measured in medical cost (M) and output loss (Q) in the UK
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Annual M+Q	Discounted 3% from 2003	10% Reduction due to REACH	
			10% environmentally Induced	50% environmentally Induced
2006	80,554.91	73,719.16	7,371.92	36,859.58
2007	80,554.91	71,572.00	7,157.20	35,786.00
2008	80,554.91	69,487.38	6,948.74	34,743.69
2009	80,554.91	67,463.47	6,746.35	33,731.74
2010	80,554.91	65,498.52	6,549.85	32,749.26
2011	80,554.91	63,590.79	6,359.08	31,795.40
2012	80,554.91	61,738.63	6,173.86	30,869.32
2013	80,554.91	59,940.42	5,994.04	29,970.21
2014	80,554.91	58,194.58	5,819.46	29,097.29
2015	80,554.91	56,499.60	5,649.96	28,249.80
2016	80,554.91	54,853.98	5,485.40	27,426.99
2017	80,554.91	53,256.29	5,325.63	26,628.14
2018	80,554.91	51,705.13	5,170.51	25,852.57
2019	80,554.91	50,199.16	5,019.92	25,099.58
2020	80,554.91	48,737.05	4,873.70	24,368.52
Total	1,208,323.72	906,456.16	90,645.62	453,228.08

TABLE 1: Medical costs (M) and Output loss (Q) for Diabetes in the EU
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Annual M+Q	Discounted at 3% from 2003	Reduction due to REACH equal to 10%	
			10% environmentally Induced	50% environmentally Induced
2006	78,164,185.33	71,531,302.27	715,313.02	3,576,565.11
2007	75,638,394.40	67,203,733.75	672,037.34	3,360,186.69
2008	73,318,649.39	63,245,311.02	632,453.11	3,162,265.55
2009	71,181,115.96	59,613,063.99	596,130.64	2,980,653.20
2010	69,204,711.13	56,269,763.17	562,697.63	2,813,488.16
2011	67,376,755.44	53,187,832.92	531,878.33	2,659,391.65
2012	65,681,997.64	50,339,782.01	503,397.82	2,516,989.10
2013	64,106,110.05	47,700,966.40	477,009.66	2,385,048.32
2014	62,636,831.26	45,250,179.60	452,501.80	2,262,508.98
2015	61,262,921.66	42,968,580.66	429,685.81	2,148,429.03
2016	59,975,220.65	40,840,206.87	408,402.07	2,042,010.34
2017	58,765,494.34	38,850,914.67	388,509.15	1,942,545.73
2018	57,626,464.53	36,988,234.74	369,882.35	1,849,411.74
2019	56,551,356.32	35,240,935.62	352,409.36	1,762,046.78
2020	55,534,755.95	33,599,440.66	335,994.41	1,679,972.03
Total	977,024,964.04	742,830,248.35	7,428,302.48	37,141,512.42

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 2A: Medical costs (M) and Output loss (Q) for Parkinson s in the EU
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Affected Population 0.4	Discounted at 3% from 2003	10% Reduction due to REACH	
	Annual M+Q		10% environmentally Induced	50% environmentally Induced
2006	5,259,853.65	4,813,511.19	48,135.11	240,675.56
2007	5,110,565.68	4,540,671.42	45,406.71	227,033.57
2008	4,973,456.16	4,290,146.97	42,901.47	214,507.35
2009	4,847,116.32	4,059,383.61	40,593.84	202,969.18
2010	4,730,300.06	3,846,166.83	38,461.67	192,308.34
2011	4,622,257.95	3,648,853.11	36,488.53	182,442.66
2012	4,522,088.56	3,465,804.34	34,658.04	173,290.22
2013	4,428,945.04	3,295,551.06	32,955.51	164,777.55
2014	4,342,102.69	3,136,827.37	31,368.27	156,841.37
2015	4,260,897.17	2,988,507.55	29,885.08	149,425.38
2016	4,184,787.05	2,849,636.35	28,496.36	142,481.82
2017	4,113,285.65	2,719,366.38	27,193.66	135,968.32
2018	4,045,962.80	2,596,949.56	25,969.50	129,847.48
2019	3,982,418.07	2,481,711.28	24,817.11	124,085.56
2020	3,922,331.46	2,373,075.04	23,730.75	118,653.75
Total	67,346,368.31	51,106,162.04	511,061.62	2,555,308.10

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 2B: Medical costs (M) and Output loss (Q) for Parkinson s in the EU
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Affected Population 0.6		10% Reduction due to REACH	
	Annual M+Q	Discounted at° 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	7,015,370.36	6,420,057.67	64,200.58	321,002.88
2007	6,790,438.27	6,033,216.45	60,332.16	301,660.82
2008	6,583,937.50	5,679,362.32	56,793.62	283,968.12
2009	6,393,735.91	5,354,653.16	53,546.53	267,732.66
2010	6,217,947.48	5,055,760.31	50,557.60	252,788.02
2011	6,055,820.02	4,780,520.25	47,805.20	239,026.01
2012	5,905,611.62	4,526,159.56	45,261.60	226,307.98
2013	5,766,040.56	4,290,475.69	42,904.76	214,523.78
2014	5,636,010.00	4,071,573.54	40,715.74	203,578.68
2015	5,514,514.49	3,867,769.51	38,677.70	193,388.48
2016	5,400,734.57	3,677,637.44	36,776.37	183,881.87
2017	5,293,933.67	3,499,913.81	34,999.14	174,995.69
2018	5,193,460.73	3,333,484.82	33,334.85	166,674.24
2019	5,098,709.68	3,177,347.31	31,773.47	158,867.37
2020	5,009,196.26	3,030,646.12	30,306.46	151,532.31
Total	87,875,461.12	66,798,577.98	667,985.78	3,339,928.90

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 3A: Output loss (Q) for Loss of IQ due to 5 point decrement in the EU
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss 275*10 ⁹		10% Reduction due to REACH	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	320,431,238.56	293,239,975.36	2,932,399.75	14,661,998.77
2007	309,335,511.61	274,840,595.52	2,748,405.96	13,742,029.78
2008	299,144,938.49	258,045,051.75	2,580,450.52	12,902,252.59
2009	289,754,815.59	242,665,096.36	2,426,650.96	12,133,254.82
2010	281,072,526.02	228,537,684.97	2,285,376.85	11,426,884.25
2011	273,042,369.21	215,542,167.62	2,155,421.68	10,777,108.38
2012	265,597,346.94	203,558,250.76	2,035,582.51	10,177,912.54
2013	258,674,518.06	192,478,134.83	1,924,781.35	9,623,906.74
2014	252,220,018.59	182,209,107.81	1,822,091.08	9,110,455.39
2015	246,184,473.24	172,668,836.35	1,726,688.36	8,633,441.82
2016	240,527,639.73	163,787,618.58	1,637,876.19	8,189,380.93
2017	235,213,346.81	155,503,731.74	1,555,037.32	7,775,186.59
2018	230,209,621.59	147,762,796.02	1,477,627.96	7,388,139.80
2019	225,486,702.15	140,515,858.01	1,405,158.58	7,025,792.90
2020	221,020,805.95	133,721,222.47	1,337,212.22	6,686,061.12
Total	3,947,915,872.54	3,005,076,128.16	30,050,761.28	150,253,806.41

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 3B: Output loss (Q) for Loss of IQ due to 5 point decrement in the EU
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss 327*10 ⁹		10% Reduction due to REACH	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	381,021,872.76	348,688,988.88	3,486,889.89	17,434,449.44
2007	367,828,044.71	326,810,453.59	3,268,104.54	16,340,522.68
2008	355,710,526.86	306,839,025.17	3,068,390.25	15,341,951.26
2009	344,544,817.09	288,550,860.03	2,885,508.60	14,427,543.00
2010	334,220,785.48	271,752,083.59	2,717,520.84	13,587,604.18
2011	324,672,199.03	256,299,232.04	2,562,992.32	12,814,961.60
2012	315,819,390.72	242,049,265.45	2,420,492.65	12,102,463.27
2013	307,587,517.84	228,874,000.33	2,288,740.00	11,443,700.02
2014	299,912,531.19	216,663,193.65	2,166,631.94	10,833,159.68
2015	292,735,719.09	205,318,943.58	2,053,189.44	10,265,947.18
2016	286,009,229.79	194,758,368.28	1,947,583.68	9,737,918.41
2017	279,690,052.38	184,908,073.74	1,849,080.74	9,245,403.69
2018	273,740,168.22	175,703,397.45	1,757,033.97	8,785,169.87
2019	268,124,187.65	167,086,129.35	1,670,861.29	8,354,306.47
2020	262,813,831.07	159,006,689.99	1,590,066.90	7,950,334.50
Total	4,694,430,873.89	3,573,308,705.12	35,733,087.05	178,665,435.26

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 4A: Dynamic Economic Impacts measured in Output loss (Q) in the EU
 (Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss $17.1 \cdot 10^9$		10% Reduction due to REACH	
	Annual Q	Discounted at 3% From 2003	10% environmentally Induced	50% environmentally Induced
2006	19,924,997.02	18,234,194.83	182,341.95	911,709.74
2007	19,235,044.54	17,090,087.94	170,900.88	854,504.40
2008	18,601,376.18	16,045,710.49	160,457.10	802,285.52
2009	18,017,481.26	15,089,356.90	150,893.57	754,467.85
2010	17,477,600.71	14,210,888.77	142,108.89	710,544.44
2011	16,978,270.96	13,402,803.88	134,028.04	670,140.19
2012	16,515,325.94	12,657,622.14	126,576.22	632,881.11
2013	16,084,851.85	11,968,640.38	119,686.40	598,432.02
2014	15,683,499.34	11,330,093.61	113,300.94	566,504.68
2015	15,308,198.15	10,736,862.19	107,368.62	536,843.11
2016	14,956,445.96	10,184,611.92	101,846.12	509,230.60
2017	14,625,993.57	9,669,504.77	96,695.05	483,475.24
2018	14,314,852.83	9,188,159.32	91,881.59	459,407.97
2019	14,021,173.12	8,737,531.53	87,375.32	436,876.58
2020	13,743,475.57	8,315,028.74	83,150.29	415,751.44
Total	245,488,586.98	186,861,097.42	1,868,610.97	9,343,054.87

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 4B: Dynamic Economic Impacts measured in Output loss (Q) in the EU
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Output Loss $85 \cdot 10^9$		10% Reduction due to REACH	
	Annual Q	Discounted at 3% from 2003	10% environmentally Induced	50% environmentally Induced
2006	99,042,382.83	90,637,810.57	906,378.11	4,531,890.53
2007	95,612,794.50	84,950,729.53	849,507.30	4,247,536.48
2008	92,462,980.99	79,759,379.63	797,593.80	3,987,968.98
2009	89,560,579.37	75,005,575.24	750,055.75	3,750,278.76
2010	86,876,962.59	70,638,920.81	706,389.21	3,531,946.04
2011	84,394,914.12	66,622,124.54	666,221.25	3,331,106.23
2012	82,093,725.42	62,918,004.78	629,180.05	3,145,900.24
2013	79,953,941.95	59,493,241.67	594,932.42	2,974,662.08
2014	77,958,914.84	56,319,178.78	563,191.79	2,815,958.94
2015	76,093,382.64	53,370,367.60	533,703.68	2,668,518.38
2016	74,344,906.83	50,625,263.92	506,252.64	2,531,263.20
2017	72,702,307.19	48,064,789.81	480,647.90	2,403,239.49
2018	71,155,701.22	45,672,136.95	456,721.37	2,283,606.85
2019	69,695,889.75	43,432,174.29	434,321.74	2,171,608.71
2020	68,315,521.84	41,332,014.22	413,320.14	2,066,600.71
Total	1,220,264,906.06	928,841,712.34	9,288,417.12	46,442,085.62

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

TABLE 5: Social Impacts measured in medical cost (M) and output loss (Q) in the EU
(Figures in thousands of euros. Benefits from the implementation of REACH start in 2006.)

Year	Annual M+Q	Discounted 3% from 2003	10% Reduction due to REACH	
			10% environmentally Induced	50% environmentally Induced
2006	1,948,193.80	1,782,873.31	178,287.33	891,436.66
2007	1,948,193.80	1,730,944.96	173,094.50	865,472.48
2008	1,948,193.80	1,680,529.09	168,052.91	840,264.54
2009	1,948,193.80	1,631,581.64	163,158.16	815,790.82
2010	1,948,193.80	1,584,059.84	158,405.98	792,029.92
2011	1,948,193.80	1,537,922.18	153,792.22	768,961.09
2012	1,948,193.80	1,493,128.33	149,312.83	746,564.16
2013	1,948,193.80	1,449,639.15	144,963.92	724,819.58
2014	1,948,193.80	1,407,416.66	140,741.67	703,708.33
2015	1,948,193.80	1,366,423.94	136,642.39	683,211.97
2016	1,948,193.80	1,326,625.18	132,662.52	663,312.59
2017	1,948,193.80	1,287,985.61	128,798.56	643,992.81
2018	1,948,193.80	1,250,471.47	125,047.15	625,235.73
2019	1,948,193.80	1,214,049.97	121,405.00	607,024.99
2020	1,948,193.80	1,178,689.29	117,868.93	589,344.65
Total	29,222,907.06	21,922,340.63	2,192,234.06	10,961,170.31

Notes: 1 EU per capita Income proxied by the per capita income of Germany. 2 EU/US exchange rate before 1999 proxied by German national currency/US dollar. 3 Institute for Fiscal Studies data used for population, per capita incomes and exchange rates.

7 CONCLUSIONS AND SUMMARY

REACH involves an extensive system of registration, risk assessment and authorisation for both new and existing chemicals. The benefits of REACH arise primarily from (a) greater and improved information about what chemicals are being produced and used, and (b) behavioural responses to the costs of complying with REACH such that some chemicals will be withdrawn from use and substituted for by other chemicals, or possibly by total withdrawal.

An ideal approach to appraising REACH would involve an assessment of the exposure to chemicals, a behavioural model which would show how the industry and users will respond to the true costs of compliance, dose-response functions for health and for environmental effects, and a procedure for placing money values on the changes in exposure. Unfortunately, the information and resources to implement such an approach are not available. In this Report we have therefore resorted to what we term an *n*th best approach. We make what we regard as reasonable assumptions about some of the key variables and parameters, and we then adopt three different models to assess the benefits of REACH. We assess *only* the health benefits since we judge that the environmental effects cannot be estimated without a detailed stated preference approach to valuing such effects, and without far better information about exposure-response functions. Accordingly, the benefits of REACH exceed the estimates shown here. However, we also caution that other features of assessing REACH are not satisfactory. In Chapter 3 we argue that the cost estimates for complying with REACH may be under- or over-estimates. One reason for the uncertainty is the absence (at the time of writing) of public information on the European Commission's estimates of the indirect costs of compliance. We also note that the assumed reduction in exposure from implementing REACH will strike some as a serious under-estimate, and others as an over-estimate, although there is *some* support for our 10 per cent rule in the Commission's estimates of 8-12 per cent reduction in chemical substances. Accordingly, there remains considerable room for debate, which is what we would expect in the absence of any publicly available detailed model of the industry and supply and demand responses.

Our first two models are based on the notion of a Disability Adjusted Life Year (DALY), a procedure for estimating the burden of disease and premature mortality in a single unit. We have taken estimates of DALYs in industrialised countries and computed DALYs-per-capita. We have made projections of these losses to 2020, and have adopted World Bank estimates of the fraction of DALYs arising from exposure to chemicals. We then make an assumption about the extent to which REACH will reduce those DALYs — the 10 per cent assumption. Finally, we value a DALY in two very different ways.

The first approach — Model I — looks at health expenditure in the UK and EU. The intuition is that this expenditure is spent on avoiding and treating the causes of DALYs, so we can compute a health expenditure per DALY. The number of DALYs saved from REACH can then be multiplied by this unit cost to estimate health expenditure saved by REACH.

The second approach — Model II — proceeds in the same way but notes that the value of a DALY is far greater than the healthcare costs incurred and must include the willingness to pay (WTP) of individuals to avoid the health states in question. We follow a procedure adopted in a World Bank study of pollution control and assign a WTP value to a DALY based on an anchor estimate of the value of a statistical life (VOSL) and an implied value of a life year (VOLY).

The third approach — Model III — proceeds differently and attempts to estimate the medical costs and forgone productivity from specific diseases or health end states. Here again, some assumption is required about the extent to which these health states are due to exposure to chemicals. We followed a US study which assumes 10-50 per cent of the resulting costs arise from exposure to chemicals. We then adopt our 10 per cent REACH effectiveness estimate to calculate the benefits of REACH.

A summary of the results from Models I-III is shown in Table 7.1. Results are shown for the EU only since we have no estimates for the wider compliance costs for the UK. We would expect Model II to produce higher results than Model I, but we have no prior expectation about the relationship between Model III and Model II since they work on different bases. On the basis of Models I and II, costs could be greater than benefits, but Model II encompasses a strong probability that benefits exceed costs. On Model III, benefits comfortably exceed costs. Once again, however, we stress that a number of assumptions have been made to secure this conclusion and it is open to anyone to challenge the assumptions. If so, the methodologies are sufficiently transparent that anyone can generate their own estimates based on what they regard as superior assumptions.

Our Models explicitly exclude any environmental benefits and hence we regard our benefit estimates as minima. **Overall, our own judgement is that we feel confident that REACH generates net benefits.**

Table 7.1 Summary of costs and benefits of REACH (EU only)

		10⁹ : Present Value, s = 3%			
		Benefits	Costs	Net Benefits	B/C Ratio
Model I	EU	4.8 — 20.1	23.6	-3.5 to — 18.8	0.21 to 0.85
Model II	EU	22.4 — 93.3	23.6	- 1.2 to + 69.8	0.95 to 3.95
		12.3 - 51.3	23.6	-11.3 to + 27.7	0.52 to 2.17
Model III	EU	56.7- 283.5	23.6	+33.1 to +259.9	2.40 to 11.01

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**TAKE ACTION: IF YOU WOULD LIKE TO SUPPORT WWF'S
CHEMICALS AND HEALTH CAMPAIGN AND TAKE ACTION
FOR A SAFER FUTURE FOR WILDLIFE AND PEOPLE, PLEASE
CALL 01483 860869 FOR A CAMPAIGN LEAFLET,
OR VISIT WWW.WWF.ORG.UK/CHEMICALS**

WWF'S CHEMICALS AND HEALTH CAMPAIGN

Along with wildlife around the world, we are being subjected to an uncontrolled and dangerous global experiment. Exposure to hazardous man-made chemicals is putting us all at risk. Our children and wildlife are especially vulnerable. WWF's Chemicals and Health campaign is seizing a once in a lifetime opportunity to put an end to this threat, by asking people to help us ensure forthcoming European chemicals legislation brings chemicals under control.

WWF is calling for hazardous man-made chemicals to be properly regulated – replaced where safer alternatives exist, or banned where necessary.

The mission of WWF – the global environment network – is to stop the degradation of the planet's natural environment and to build a future in which humans live in harmony with nature, by:

- conserving the world's biological diversity
- ensuring that the use of renewable resources is sustainable
- promoting the reduction of pollution and wasteful consumption

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Taking action for a living planet