Regulatory assessment for chemicals: a rapid appraisal cost–benefit approach

David Pearce[a,b,*], Phoebe Koundouri[a,c]

[a]*Department of Economics/CSERGE, University College London, Gower St., London WC1E 6BT, UK*
[b]*Department of Environmental Science and Technology, Imperial College London, UK*
[c]*Department of Economics, School of Business, University of Reading, UK*

Abstract

The purpose of this paper is to explore alternative 'rapid appraisal' methodologies for determining the costs and benefits of environmental legislation, the focus being the new Chemicals Policy in the European Union (EU) known as REACH (Registration, Evaluation and Authorisation of Chemicals). We show that a full and rigorous cost–benefit appraisal of this legislation is not possible because of informational deficiencies. Hence, some 'second best' approach is required. In addition, full cost–benefit appraisal is likely to be expensive and impossible to conduct in the near-term. We argue that it is possible to obtain some broad estimates of gains and losses by making reasonable assumptions and by pursuing different methodologies for estimating benefits. Two methodologies, both based on the notion of a disability-adjusted life year (DALY), are adopted. A DALYis a measure of health loss, enabling different forms of morbidity to be compared with premature mortality. We seek to 'monetise' DALYs in order to make a direct comparison with the costs of the policy measures. The first approach addresses health expenditure in the UK and EU, based on the presumption that this expenditure is incurred in order to avoid and treat the causes of DALYs. Health expenditure per DALY is thus a measure of the value of a DALYand this expenditure is avoided by reductions in DALYs due to environmental control. The second approach assigns a willingness-to-pay 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). On the basis of these models we show that while the costs of REACH could be greater than benefits, the second approach reveals a strong probability that benefits exceed costs. Since our models explicitly exclude any environmental benefits, we regard our benefit estimates as minima. Overall, our own judgement is that we feel confident that REACH generates net benefits, a result consistent with other partial studies that have been carried out to date.

Keywords: Cost–benefit analysis; REACH; Disability-adjusted life years (DALYs); Value of a statistical life (VOSL); Value of a life year (VOLY)

1. Introduction

In October 2003, The European Commission published draft legislation aimed at controlling exposure to both new and existing chemicals in the European Union (EU) (http://europe.eu.int/comm/enteprise/chemicals/chempol/whitepaper/consultation.htm). This New European Chemicals Strategy (NECS) involves the establishment of a new system of Registration, Evaluation and Authorisation of Chemicals (REACH) covering chemicals marketed in quantities over 1 tonne per annum (t/y) (CEC, 2003a). The legislation follows on from an earlier White Paper on Chemicals issued in 2001 (CEC, 2001). In essence, the legislation requires all new and existing chemicals to be registered and authorised. 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, 2001, para 1.1), the Commission initially produced 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, 2001; RPA, 2003). At the time of writing (July 2004), the Commission has initiated a consultancy study to evaluate REACH. Results from this study are expected in 2005.

Countries such as the United Kingdom, which have established Regulatory Impact Assessment procedures, have sought to identify the costs of REACH for the UK, and some attempt has also been made to estimate some of the benefits (RPA, 2001; ERM, 2003). The fact remains that full cost–benefit appraisals of REACH are not possible. This is because of informational deficiencies mainly relating to (a) the dose–response relationships linking chemical exposure to human health effects; (b) the absence of any detailed behavioural model of the chemical industry, which means that there is considerable uncertainty about the effects of REACH on behaviour in the chemical industry and among users with respect to substitutes; and (c) virtually zero knowledge relating exposure to ecosystem effects.

The issue arises, then, whether the NECS passes a cost–benefit test. The purpose of this paper is to set out a 'rapid appraisal' approach to benefit estimation. The methodology is transparent and it is open to anyone to substitute their own judgements on the reasonableness of the relevant parameter values. In the conclusion, we offer our own judgement, which, so far, is consistent with the findings of available alternative studies. We also draw attention to the limitations of our own approach.

The information and resources to implement an ideal approach are not available. Firstly, the ways in which REACH will work, and hence the full costs of implementing REACH, are still not entirely clear. Additionally, the precise number of chemicals is not known, nor, more importantly, are the behavioural reactions of suppliers and users of chemicals to changes in cost and availability. Secondly, the epidemiological basis for determining the benefits of REACH in terms of improved public and occupational health is not known for the many chemicals affected. Thus, even if the change in exposure was known, the change in health and environmental responses remains to be estimated across the relevant chemicals. Currently, that information is available for only a limited number of chemicals and only then with considerable degrees of uncertainty. Thirdly, REACH will generate benefits in terms of reduced non-health environmental damage, but the relationship between the chemicals and environmental responses is not known. Additionally, while procedures for placing economic values on health effects have improved considerably, economic values for such impacts as changes in biological diversity remain elusive and under-researched.

In this paper, we have therefore resorted to what we term a 'second best' approach. We make what we regard as reasonable assumptions about some of the key variables and parameters, and we then adopt two 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. Neither piece of information is currently available. Accordingly, we judge that the benefits of REACH exceed the estimates shown here.

Our 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—basically that REACH will reduce health exposure by 10%. This assumption corresponds closely to that suggested by the European Commission's estimates of the reduction in the number of chemicals substances (8–12%). But we caution that the number of substances is not the same thing as level of exposure, and it is the latter that matters. Accordingly, the 10% figure should be taken as a working hypothesis. As we see later, at least one recent study for the UK suggests that benefits to the UK exceed costs even if exposure is reduced by a mere fraction of 1%.

Finally, we place an economic value on a DALY in two very different ways. The first approach—Model I—looks at health expenditure in the UK and EU. We assume 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). Each of the steps in these procedures faces methodological and data problems. We list these after the results are presented.

For the sake of brevity, the results are shown for the European Union only (and, indeed, for the EU excluding recent Accession countries). We would confidently expect Model II to produce results more favourable to the legislation than Model I, since willingness-to-pay must exceed healthcare costs. In fact, Model II produces a very strong likelihood that benefits exceed costs.

1.1. REACH legislation

The New European Chemicals Strategy involves the establishment of Registration, Evaluation and Authorisation of Chemicals (REACH) and which covers chemicals marketed in quantities over 1 t/y. REACH replaces the

Existing Substances Regulation (793/73/EEC (ESR)), which has proven to be frustratingly slow in terms of formalising risk assessments for chemicals. Only a handful of chemicals, out of many thousands, have been evaluated. Under REACH, all chemicals manufactured in quantities greater than 1 t/y (per company) must be registered. Those manufactured in quantities greater than 100 t/y must be evaluated, and chemicals giving rise to 'special concern' must be authorised. It is argued that REACH will ensure more rapid registration and assessment, not least because registrations will be public and hence subject to public pressure for the disclosure of full information, and because the onus of registration and assessment is on industry rather than any government agency.

The 'authorised' category covers, inter alia, carcinogens, mutagens and reproductive toxic substances (CMRs), PBTs (persistent, bioaccumulative and toxic chemicals), and vPvBs (very persistent and very bioaccumulative chemicals). 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. 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 paper, 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 overstated. 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. Unfortunately, information is not available on any systematic basis for us to determine the size nor even the direction of this potential bias.

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 t/y, 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% 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). 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). Whether intermediates are accounted for, affects the estimates of costs and benefits.

The very 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 them.

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 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 1000 t/y 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—e.g. packaging waste. 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 European 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 as we show later.[1]

REACH involves the following stages of assessment of chemicals:

(a) *Registration* of all substances produced in quantities above 1 t/y. 'Registration only' is likely to affect around 80% of such 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 t/y.
(b) *Evaluation* of substances produced in quantities above 100 t/y (perhaps 15% 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 (2001) 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% of all substances. Also included are persistent, bio-accumulative and toxic (PBT) substances, and very persistent and very bio-accumulative substances (vPvB).

Overall, then, REACH is a procedure for registering, testing and authorising new and existing chemicals. Compared to existing practices, REACH would, if enacted fully, ensure that safety data are available for all chemicals on the market (rather than just new chemicals), provide more comprehensive information about the hazards associated with each chemical, and, potentially, bring about 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 section 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. The 'ideal' model can be summarised as follows. The benefits that ensue from REACH are given by:[2]

$$PV(B) = \frac{\sum_{i;j;t} DI_{i;t}(DX_{j;t})}{(1+s)^t} \quad (1)$$

The notation '*i*' refers to the individual impacts, '*j*' to the individual chemicals, and *t* to time. 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. A present value expresses the stream of benefits or costs with due allowance being made for the fact that future costs and benefits will be less 'important' than current costs and benefits. Essentially, society 'discounts' the future. The rate at which the future is discounted is given by $1/(1+s)^t$ where *s* is the discount rate. DI refers to the change in impact (exposure) from a reduction the quantity of chemicals (DX). REACH would pass a cost–benefit test if PV(*B*) $>$ PV(*C*), i.e.:

$$PV(B-C) = \frac{\sum_{i;j;t} DI_{i;t}(DX_{j;t})}{(1+s)^t} - PV(C) > 0 \quad (2)$$

Notice that (2) could be met overall in the EU as a whole, 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.

At this point it is important to note 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. 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 ('consumer surplus' refers to the fact that at least some people purchasing a product are willing to pay more for the product than the price paid. The measure of their benefit from the product is their true willingness to pay for it, rather than the price they actually pay. Hence, in cost–benefit analysis efforts are usually made to estimate this change in consumer surplus, the excess of willingness-to-pay over price). 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. Insofar 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

[1] 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.

[2] Eq. (1) ignores location for convenience of exposition, but it will be appreciated that benefits and costs vary by location.

compliance cost estimates would capture at least some of the costs to users. While a microeconomic model of the chemical industry does exist (Canton and Allen, 2003) its use so far appears to be restricted to the registration phase of the policy. As such, the 'true' costs of complying with REACH remain difficult to assess.

A second issue concerns the impacts on employment. It is widely argued that changes in employment resulting from a policy should be 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 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 to support the view that regulation benefits competitiveness and employment. This effect, consistent with the 'Porter hypothesis' after Porter (1991), allegedly arises because corporations are stimulated by regulation to seek new technologies which are not just cleaner, but cost-reducing as well. There is, however, little or no evidence that the Porter hypothesis is true—see the extensive review of the evidence in Roediger-Schluga (2004). Insofar 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 the NECS.

Second, evidence that employment is negatively affected by environmental regulation is also difficult to find. It is true that there may be initial unemployment impacts within the regulated industry, in this case chemicals, but flexible labour markets, whereby labour moves from the affected industry to other industries, are likely to absorb these effects. The available empirical studies support this finding—for a review see Roediger-Schluga (2004, Chapter 2) and Sprenger (1997). Detailed econometric studies have also 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, for example, Morgenstern et al. (2002).

Third, much the same goes for the suggestion that regulation induces relocation of firms to 'pollution havens'—again see the review in Roediger-Schluga (2004, Chapter 2). By and large, pollution control costs are not large relative to labour and capital costs, which tend to dominate cost structures.

Finally, *if there are* employment *losses* it would be appropriate to consider the social cost of the loss of well-being 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 cost–benefit analysis.

A related issue concerns estimates of the extent to which REACH will reduce the use of animals for chemicals testing. Some experts have claimed that this 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 well-being 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 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.

Models of the kind shown in Eq. (2) 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 loads'—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. 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. Firstly, we have no sufficiently detailed economic model of the chemical industry—including users—with which to simulate the effects of any policy change such as NECS. A valuable beginning has been made by Canton and Allen (2003), using a Dixit-Stiglitz (1977) framework. This model has been used by the European Commission to determine what they regard as the likely costs of compliance for REACH (CEC, 2003d). Nonetheless, the model is unlikely to capture all the general equilibrium and dynamic effects on cost, e.g. technological stimulation, and costs incurred beyond those on the industry and downstream users. Secondly, we do not know the health and environmental exposure–response functions ($\mathrm{D}I(\mathrm{D}X)$) for the chemicals, of which, in any event there are many thousands. Thirdly, there is considerable uncertainty as to the locations at which risks will change. Fourthly, we do not know the split between occupational and public health effects. Finally, we do not know the time-schedule of $\mathrm{D}X$ either, although some assumption could be made about this with respect to the registration deadlines. Although we have some economic values for health end-states based on the same ExternE work as used by existing air pollution cost–benefit studies (CEC, 1999 and http://externe.jrc.es/), valuation of 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. Accordingly, we proceed with 'second best' approaches.

It should be noted that one study has attempted to estimate occupational health benefits via dose–response functions—RPA (2003). This study takes various health end points—skin disease, respiratory diseases, eye disorders, central nervous system diseases and cancers—and makes assumptions about the reductions in these cases that would arise under REACH. The RPA analysis is confined to occupational effects only, suggesting that allowing for exposure reductions to the general public would magnify the benefits. However, the relevance of occupational benefits can be questioned since wage rates may already internalise some of the risks associated with occupational exposure, i.e. workers may already be compensated to some extent. We return to the RPA study later.

3. The costs of REACH

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 costs. 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 well-being, 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. 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.

During the evolution of the REACH proposals numerous compliance cost estimates were cited. Most involved the 'direct' costs, i.e. the costs to the chemical industry of registering and testing chemicals (RPA, 2001; RPA and Statistics Sweden, 2002; CEC, 2001; CEC, 2003b, 2003c). In 2003 much broader estimates of compliance costs, exceeding s20 billion circulated. However, the European Commission's 'consensus' figure is s5.2 billion (CEC, 2003d). This corresponds to the upper end of the range of cost estimates produced by Canton and Allen (2003). Assuming substantial substitution between chemicals, so that withdrawn chemicals are easily replaced by others, the range of costs is s2.8–3.6 billion. Assuming lower substitutability, costs range from s4.0 to 5.2 billion. In keeping with the Commission's view, and because we have not investigated the compliance cost issue in any detail, we adopt the s5.2 billion figure here.

4. Model I—DALYs and healthcare costs

4.1. The model

The first model aimed at approximating the benefits of REACH makes use of the concept of a 'disability-adjusted life year' or DALY. 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. First, the duration of time lost due to a death at a give age. Taking maximum life expectancies as 82.5 years for women and 80 years for men, a man dying at age x has lost $80-x$ years. Second, the disability (or 'quality of life') weights, D. Expert assessments are used to assign weights between 0 (perfect health) and 1 (death).

Third, 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 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:

$$W = Cxe^{-bx} \quad (3)$$

Eq. (3) 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, $b = 0.04$ and x is age.

The third element is the discount function:

$$e^{-r(x-a)} \quad (4)$$

where r is the discount rate and is set at 3% (0.03) and a the year of the onset of disease. The fifth element is the additivity assumption, that is, health is added across individuals so that two people each losing 10 DALYs is same as one person losing 20 DALYs. Two equations emerge from these five elements:

$$DALY(x) = DCx^{-bx}e^{-r(x-a)} \quad (5)$$

$$DALY = \left[\frac{DCe^{-ba}}{(b+r)^2}\right]\left[e^{-(b+r)L}\{-(1+b+r)(L+a)-1\} - \{-(1+r+ba)a-1\}\right] \quad (6)$$

Eq. (5) is the number of DALYs lost due to a disability at age x. Eq. (6) is the integral of Eq. (5) between $x = a$ and $L = a$.

In our model, we make use of the DALY concept as follows. First, we take estimates of DALYs for 'Established Market Economies' (EMEs) from the WHO/World Bank database. Second, we calculate the number of DALYs-per-capita for the EME region. Third, 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. Fourth, 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% and a high estimate of 2.5% (Lvovsky, 2001). DALYs lost due to agro-industrial pollution are construed to be due to exposure to chemicals. Fifth, 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. Finally, 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 (full details can be found in Pearce and Koundouri, 2003). 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. 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 (0.6–2.5%) of these DALYs are due to agro-industrial pollution, or approximately 43,000–178,000 DALYs. We estimate that the UK Health Service spends s5624 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 s250–1000 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 and summed to obtain a present value. Our estimate of the cost of *total exposure* to chemicals is then: s3,731–15,550 million, or s3.73–15.55 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. 0.6–2.5% 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.

4.2. 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 and Statistics Sweden (2002) make some estimates of these effects. However, there appear to be no detailed 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. Some preliminary estimates are given in Canton and Allen (2003). What matters for the cost–benefit analysis

Table 1
RPA implied estimates of change in 'exposure' due to REACH

Scenarios	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 and Statistics Sweden (2002), Table 1. *Note*: The proportions are similar if intermediates and unintended uses are omitted, at —27% and 48%, respectively.

is the change in *exposure* (D*X*) 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 1.

On the basis of these estimates, we could take as a 'maximum effect' scenario, a range of say 30–50% 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 (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% reduction in exposure and then estimate the change in exposure level that would make costs equal benefits, i.e. a 'switchover' point. The 10% assumption can obviously be changed if it is judged that REACH will have less/greater effects. Table 2 summarises the resulting comparison of costs and benefits.

4.3. Conclusions on Model I

Table 2 suggests that REACH would probably pass a cost–benefit test at the EU level but possibly not at the UK level. 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% 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); and (c) the omission of any environmental benefits.

Cautions in the other direction 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. Occupational benefits may not be a 'legitimate' benefit if it can be argued that workers are already compensated for risk exposure through wage rates that contain a risk-compensation element. Overall, our judgment is that 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.

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

Table 2
Comparison of REACH costs and benefits using DALYs and healthcare costs only (present value, s billion, *s* = 3%)

	UK	EU
Total cost of 'chemical' induced DALYs, 2006–2020 PV, s10^9	3.73–15.55	48.34–201.43
Benefit of REACH assuming 10% reduction in total exposure PV, s10^9	0.37–1.56	4.83–20.14
Cost of complying with REACH PV, s10^9	>1.0	5.2
Benefit minus cost[a]	Less than —0.63 to 0.56	—0.37 to 14.94
Worst case break-even level of exposure[a] (%)	27.0	10.8

[a] Benefits are 4.83 and compliance costs are 5.2, so exposure needs to be reduced by (5.2/4.83) × 10% = 10.77%.

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 good.

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 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 environment. 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 (2003) has estimated *occupational* benefits from REACH at $18–30 billion over a 30-year period, using a 'value of life saved' approach and with assumptions about the effects of REACH on human exposure. These estimates are revisited later as a check on our own estimates.

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:

$$\text{VOSL} = \frac{\sum_i \text{WTP}_i}{\Delta r N} \quad (7)$$

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.

Estimates of VOSL vary. In the UK, an official VOSL of some £1.2 million = $2.0 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). The European Commission has been using a benchmark figure of $1 million. We opt for a figure of £1 million = $1.67 million 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. Thus, for chronic exposure one would be seeking a measure of future

Table 3
Values of DALYs based on WTP to avoid illness and fatality risks

	Value of DALY, 2003$	Value of DALY, 2003s	Income ratio UK to Country[a]	Income ratio EU to Country	Value of a DALY UK 2003s	Value of a DALY EU 2003s
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	

[a] 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.

gains in life expectancy, and the notion of a 'value of a life year' (VOLY) seems more appropriate. The epidemiology of chronic exposure to air pollutants is still weak, but it is thought that, in Europe, exposure to current levels reduces life expectancy by around 6 months (Künzli et al., 2000). Unfortunately, few studies have attempted to estimate this magnitude, nor do we have any real idea of how a policy such as REACH would affect life expectancy.

5.3. Valuing DALYs

Chemicals also involve ill-health independently of any premature mortality. WTP studies exist for states of ill-health. However, rather than estimate VOSLs and morbidity values separately, 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; and (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 $2.4 million in 2003 prices). We consider £1 million to be appropriate for Europe and hence retain the US$ 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 3. We have then scaled these WTP values up by the ratio of per capita income in Europe and the UK to the countries shown. Table 3 suggests that the appropriate value for a DALY in Europe/UK is around s90,000 per DALY. This can be compared to the health expenditure figure of some s5600 per DALYused in Section 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:

$$\mathrm{VOSL} = \mathrm{VOLY}(s) \sum_{i=a} \frac{aPi}{(1+s)^{i-a}} \quad (8)$$

where aPi 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 s40–50,000 for a VOSL of s1.5 million, or, say, s45–55,000 for a VOSL of s1.67 million. Such values fits neatly with the UK Department of Health procedure of valuing a 'QALY' (Quality Adjusted Life Year, which is akin to a DALY) at £30,000, and reasonably with very recent work in the UK on VOLYs which provides a range of £27,000 (Chilton et al., 2004) to £42,000 (Markandya et al., 2004).

5.4. Results for Model II

Table 4 shows the results for Model II. At the higher value of a DALY (s90,000), we see that, while there is a small chance of costs exceeding benefits, benefits are more likely

Table 4
Comparison of REACH costs and benefits using DALYs and willingness-to-pay (present value, s billion, s = 3%)

	Total cost of 'chemical' induced DALYs, 2006–2020 PV, s10^9	Benefit of REACH assuming 10% reduction in total exposure PV, s10^9	Cost of complying with REACH PV, s10^9	Benefit minus cost[a]
UK	36.0–150.0	3.60–15.00	>1.0	2.60–14.00
EU	223.9–932.7	22.39–93.27	5.2	17.19–88.07
UK	19.8–82.5	1.98–8.25	>1.0	1.0–7.20
EU	123.1–513.0	12.31–51.30	5.2	7.11–46.10

[a] See footnote to Table 2.

to exceed costs and by a significant amount. At the lower value of a DALY (s50,000) 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 Model I.

One check on the reasonableness of the estimates in Table 4 is to consider the ratios of benefits to costs. These are approximately 2–10 for the lower value of the DALYand 3–18 for the higher value. These are high but not unreasonably high. The US EPA's 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 s1.6 million. The VOSL estimates are the major factor 'driving' the results of the US EPA study and they are similarly the major factor producing the s90,000 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.

Another check is possible. The health expenditure cost estimates are, in our view, fairly robust. Yet we know WTP exceeds health expenditure costs. If we knew the ratio of WTP to health expenditure costs, we could adjust the health expenditure costs by some factor. 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 health expenditures. This procedure is clearly not satisfactory, as there are few studies that estimate health expenditure and WTP. Moreover, the Rowe et al. 'cost of illness' value dates from the mid-1970s. Clearly, if their ratio was applicable, Model II would yield only a 50% increase in the benefits assessed in Model I.

Finally, some comparison can be made with other studies. A partial Regulatory Impact Assessment for the UK (ERM, 2003) produces an extremely optimistic appraisal. Acknowledging, as we do, that we do not know the likely reduction in exposure arising from REACH, ERM (2003) proceeds by taking likely economic values of health end points and then asking what change in exposure would have to occur for benefits to equal costs. In other words, it seeks the 'switchover' point where costs greater than benefits switches into benefits greater than costs. They find a remarkably small switchover value of a fraction of 1% (0.3%), i.e. exposure would need to change by considerably less than 1% for REACH to have benefits greater than costs. RPA (2003) comes closer to the 'ideal' approach set out earlier, but, again without detailed dose–response functions, makes assumptions about the effects of REACH on occupational exposure alone. Superficially, the results are comparable to those given in Table 4 above: the present value of benefits is put at s18–27 billion, suggesting that costs exceed benefits at the lower end of the range and vice versa for the upper end of the range. Once again, however, the validity of accounting for occupational effects alone is open to question, whilst the procedure will under-estimate benefits if public exposure is included, and if positive but unknown environmental benefits are included.

6. Caveats on DALYs

Our 'rapid appraisal' approach involves the use of DALYs. A number of caveats can be raised about this procedure. First, estimates of DALYs lost are currently only available on a broad regional basis. It would be better to have them broken down country-by-country. Such data should be available in the near future. Second, we have assumed DALYs lost due to chemicals exposure are the same as DALYs lost due to agro-industrial pollution as identified by the World Bank. Third, we have applied money values to lost DALYs. The procedure for doing this relies on some compatibility between DALY values and estimates of VOSL. These are currently the subject of continuing debate, although there appears to be a growing consensus that numbers like £1–1.2 million per statistical life are appropriate for the UK.

Finally, it is known that, while VOSL estimates are fairly firmly grounded in economic theory, being based on willingness-to-pay to avoid small changes in mortality risk, DALYs (and QALYs) may not be consistent with the notion of willingness-to-pay—for example, see Gyrd-Hansen (2003) and Mauskopf, 1991. Advocates of DALYs argue that they are founded in preferences because both the age–weighting function (which places the highest weight on a young adult, with weights declining for older and young people relative to this benchmark) and the severity scales used to classify health states (interval {0,1}) come from surveys of some kind. Others question the limited nature of the expert surveys used to derive these 'consensus' views. Hammitt (2002) shows that the utility function (the underlying representation of people's preferences) implied by QALYs (and hence by implication, DALYs) is restrictive, i.e. has a lot of constraints. Hammitt tests the differences between WTP and QALYs by looking at determining factors, and shows the following:

Determining factor	QALYs	WTP
Competing risk	Decrease	Decrease
Life expectancy	Increase	Ambiguous
Co-morbidities	Decrease	Ambiguous
Baseline risk	No effect	Increase
Wealth (income)	No effect (generally)	Increase

Competing risk refers to risks faced if the individual survives the risk in question: e.g. risk of dying anyway—the 'dead anyway' assumption. Life expectancy is

self-explanatory. Co-morbidity refers to the presence of other diseases. Baseline risk refers to the probability of death or morbidity. Ultimately, Hammitt favours the WTP approach but DALYs have some attractions. They are more 'egalitarian' than WTP because WTP is determined by wealth and income whereas wealth does not (directly anyway) affect DALYs. On the other hand, DALYs limit the nature of preferences for risk and we know that, in fact, preferences may be very varied.

Clearly, using DALYs in the way we have in this study can be questioned. Equally, DALYs (and QALYs) are finding widespread acceptance in policy circles because, once the substantial initial effort of computing them is carried out, they can be employed in a variety of ways. One of those ways is to ally them with some money valuation in the manner shown here.

7. Conclusions

REACH involves an extensive system of registration, risk assessment and authorisation for both new chemicals 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 study, we have therefore resorted to what we term a 'second best' approach. We make what we regard as reasonable assumptions about some of the key variables and parameters, and we then adopt two 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 true benefits of REACH exceed the estimates shown here. However, we also caution that other features of assessing REACH are not satisfactory. In Section 3, we argue that the cost estimates for complying with REACH may be under- or over-estimates. 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% rule' in the Commission's estimates of 8–12% 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 two models are based on the notion of a disability-adjusted life year, 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% 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 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).

We note that both the notion of a DALY and the idea of placing a money value on a DALYare controversial. DALYs may not be consistent with the underlying notion of utility maximisation, which in turn underlies cost–benefit analysis in its conventional form. We also note that there is some debate about the money value to be attached to a 'life year', but that recent studies are tending to confirm the kinds of values we have employed in this study.

A summary of the results from the two models is shown in Table 5. Results are shown for the EU only. We would expect Model II to produce higher results than Model I. On the basis of Model I costs could be greater than benefits, but it seems very unlikely. Model II suggests that benefits are unambiguously greater than costs. Once again, however, we stress that a number of assumptions have been made to secure this

Table 5
Summary of costs and benefits of REACH (EU only)

$10^9: present value, *s* = 3%			
Benefits	Costs	Net benefits	*B/C* ratio
Model I			
4.8–20.1	5.2	—0.4 to 14.9	0.92–3.87
Model II			
At $90,000 per DALY			
22.4–93.3	5.2	17.2 to 88.1	4.3–17.94
At $50,000 per DALY			
12.3–51.3	5.2	7.1 to 46.1	2.17–9.87

Table A.1
Chemically induced healthcare costs and avoided costs of REACH for the UK

Year	NHS expenditure[a]	CPI (base year 2000)[b]	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	40,264,710	194	46,814,372	42,841,782	257,051	1,071,045	6,016
2007	40,319,240	199	48,112,034	42,746,919	256,482	1,068,673	6,005
2008	40,373,770	204	49,413,035	42,624,118	255,745	1,065,603	5,990
2009	40,428,300	209	50,717,375	42,475,003	254,850	1,061,875	5,971
2010	40,482,830	214	52,025,053	42,301,129	253,807	1,057,528	5,949
2011	40,537,360	219	53,336,070	42,103,986	252,624	1,052,600	5,923
2012	40,591,890	224	54,650,426	41,885,001	251,310	1,047,125	5,894
2013	40,646,420	229	55,968,120	41,645,537	249,873	1,041,138	5,863
2014	40,700,950	235	57,289,152	41,386,903	248,321	1,034,673	5,829
2015	40,755,480	240	58,613,523	41,110,346	246,662	1,027,759	5,792
2016	40,810,010	245	59,941,233	40,817,063	244,902	1,020,427	5,752
2017	40,864,540	250	61,272,281	40,508,196	243,049	1,012,705	5,711
2018	40,919,070	255	62,606,668	40,184,838	241,109	1,004,621	5,668
2019	40,973,600	260	63,944,394	39,848,032	239,088	996,201	5,622
2020	41,028,130	265	65,285,458	39,498,776	236,993	987,469	5,575
Total for years 2006–2020					3,731,866	15,549,441	

Figures in thousands of Euros. Benefits from the implementation of REACH start in 2006.

[a] Extrapolations of NHS expenditure are based on average annual percentage (%) change, using 70% of 1996–1998 figures on gross NHS expenditure (*Source*: Department of Health; www.doh.gov.uk).

[b] UK Consumer Price Index (*Source*: www.statistics.gov.uk).

conclusion and its 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.

Of more importance, however, is the methodology. It was noted that full cost–benefit appraisals may be very expensive

Table A.2
Chemically induced healthcare costs and avoided costs of REACH for the EU

Year	NHS expenditure	CPI (base year 2000)[a]	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	603,970,650	113	653,675,412	598,205,601	3,589,234	14,955,140	13,424
2007	604,788,600	115	663,241,852	589,281,795	3,535,691	14,732,045	13,243
2008	605,606,550	116	672,831,775	580,390,599	3,482,344	14,509,765	13,061
2009	606,424,500	118	682,445,179	571,537,093	3,429,223	14,288,427	12,880
2010	607,242,450	119	692,082,065	562,726,052	3,376,356	14,068,151	12,699
2011	608,060,400	121	701,742,433	553,961,957	3,323,772	13,849,049	12,519
2012	608,878,350	122	711,426,283	545,249,007	3,271,494	13,631,225	12,339
2013	609,696,300	124	721,133,614	536,591,134	3,219,547	13,414,778	12,160
2014	610,514,250	125	730,864,428	527,992,013	3,167,952	13,199,800	11,982
2015	611,332,200	127	740,618,723	519,455,071	3,116,730	12,986,377	11,805
2016	612,150,150	128	750,396,500	510,983,502	3,065,901	12,774,588	11,629
2017	612,968,100	130	760,197,759	502,580,274	3,015,482	12,564,507	11,454
2018	613,786,050	131	770,022,499	494,248,141	2,965,489	12,356,204	11,280
2019	614,604,000	133	779,870,722	485,989,651	2,915,938	12,149,741	11,107
2020	615,421,950	134	789,742,426	477,807,156	2,866,843	11,945,179	10,936
Total for years 2006–2020					48,341,994	201,424,976	

Figures in thousands of Euros. The benefits from the implementation of REACH start in 2006.

[a] Harmonised index of consumer prices (HICP). *Source*: www.statistics.gov.uk. The harmonised index of consumer prices 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.

Table B.1
Willingness-to-pay estimates for the UK, where one DALY is equal to s90,000

Years	P_DALY	P_DALY discounted at 3% (from 2003)	Health benefits of REACH	
			0.6% Scenario	2.5% Scenario
2006	92,727	84,858	331,827	1,382,613
2007	93,654	83,211	315,791	1,315,796
2008	94,591	81,595	300,530	1,252,209
2009	95,537	80,011	286,007	1,191,694
2010	96,492	78,457	272,185	1,134,103
2011	97,457	76,934	259,031	1,079,296
2012	98,432	75,440	246,513	1,027,137
2013	99,416	73,975	234,600	977,498
2014	100,410	72,538	223,262	930,259
2015	101,414	71,130	212,472	885,302
2016	102,428	69,749	202,204	842,518
2017	103,453	68,394	192,432	801,801
2018	104,487	67,066	183,132	763,052
2019	105,532	65,764	174,282	726,175
2020	106,587	64,487	165,859	691,080
Total for years 2006–2020			3,600,128	15,000,532

Figures in thousands of Euros. Benefits from the implementation of REACH start in 2006.

even when full information is available. When information is as seriously deficient as in the case of exposure to chemicals, some 'rapid appraisal' technique is required which allows best judgement to be used in place of detailed scientific information. Our 'DALY' approach offers one such rapid appraisal technique. It is also more general than chemicals exposure.

Acknowledgements

We are very grateful to WWF-UK for financial support and Justin Woolford of WWF-UK and Michael Warhurst of WWF European Policy Office for detailed comments on an earlier draft. We are also indebted to two anonymous referees for very detailed and helpful comments. We alone remain responsible for any remaining errors of fact or interpretation.

Appendix A. Data and estimates for Model I

See Tables A.1 and A.2.

Appendix B. Data and estimates for Model II

See Tables B.1 and B.2.

Table B.2
Willingness-to-pay estimates for the EU, where one DALY is equal to s90,000

Years	P_DALY	P_DALY discounted at 3% (from 2003)	Health benefits of REACH	
			0.6% Scenario	2.5% scenario
2006	92,727	84,858	2,076,308	8,651,284
2007	93,654	83,211	1,973,940	8,224,749
2008	94,591	81,595	1,876,615	7,819,229
2009	95,537	80,011	1,784,085	7,433,688
2010	96,492	78,457	1,696,114	7,067,143
2011	97,457	76,934	1,612,478	6,718,659
2012	98,432	75,440	1,532,963	6,387,347
2013	99,416	73,975	1,457,366	6,072,360
2014	100,410	72,538	1,385,495	5,772,895
2015	101,414	71,130	1,317,165	5,488,187
2016	102,428	69,749	1,252,203	5,217,511
2017	103,453	68,394	1,190,442	4,960,174
2018	104,487	67,066	1,131,725	4,715,520
2019	105,532	65,764	1,075,902	4,482,925
2020	106,587	64,487	1,022,830	4,261,793
Total for years 2006–2020			22,385,632	93,273,465

Figures in thousands of Euros. Benefits from the implementation of REACH start in 2006.

References

Canton, J., Allen, C., 2003. A Microeconomic Model to Assess the Economic Impacts of the EU's New Chemicals Policy. DG Enterprise, Commission of the European Communities, Brussels.

Chilton, S., Covey, J., Jones-Lee, M., Loomes, G., Metcalf, H., 2004. Valuation of Health Benefits Associated with Reductions in Air Pollution. DEFRA, London.

Commission of the European Communities, 1999. ExternE: Externalities of Energy. 10 Volumes. CEC, Brussels.

Commission of the European Communities, 2001. White Paper: Strategy for a Future Chemicals Policy. COM (2001) 88 Final. CEC, Brussels.

Commission of the European Communities, 2003a. Consultation Document Concerning the Registration, Evaluation, Authorisation and Restrictions of Chemicals. 7 Volumes. CEC, Brussels. At http://europa.eu.int/comm/enterprise/chemicals/chempol/whitepaper/reach.htm.

Commission of the European Communities, 2003b. Questions and Answers on the New Chemicals Policy REACH. At http://europa.eu.int/comm/enterprise/chemicals/chempol/whitepaper/reach.htm.

Commission of the European Communities, 2003c. Draft Impact Assessment Form of the Chemicals Review. CEC, Brussels.

Commission of the European Communities, 2003d. Extended Impact Assessment. SEC(2003) 1171/3. Commission Staff Working Paper, Brussels.

Day, B., 1998. A meta-analysis of wage–risk estimates of the value of statistical life. In: CSERGE, Benefits Transfer and the Economic valuation of Environmental Damage in the European Union: With Special Reference to Health. Report to DGXII, European Commission, Brussels.

DEFRA, 2002. New EU Chemicals Strategy: Position Statement by the UK Government and the Devolved Administrations. DEFRA, London. www.defra.gov.uk.in.

Dixit, A., Stiglitz, J., 1977. Monopolistic competition and optimal product diversity. Am. Econ. Rev. 63 (3), 297–308.

ERM (Environmental Resources Management), November 2003. New European Chemicals Strategy: UK Partial Regulatory Impact Assessment. DEFRA, London.

Freeman III, A.M., 2003. The Measurement of Environmental and Resource Values: Theory and Methods. Resources for the Future, Washington, DC.

Gyrd-Hansen, D., 2003. Willingness to pay for a QALY. Health Econ. 12, 1049–1060.

Hammitt, J.K., 2002. QALYs versus WTP. Risk Anal. 22 (5), 985–1001.

Krewitt, W., Holland, M., Trukenmüller, A., Heck, T., Friedrich, R., 1999. Comparing costs and benefits of strategies to combat acidification and ozone in Europe. Environ. Econ. Policy Stud. 2, 249–266.

Künzli, N., et al., 2000. Public-health impact of outdoor and traffic-related air pollution: a European assessment. Lancet 356, 795–801.

Lvovsky, K., 2001. Environment and Health. World Bank, Washington, DC.

Lvovsky, K., Hughes, G., Maddison, D., Ostro, B., Pearce, D.W., 2000. Environmental Costs of Fossil Fuels: A Rapid Assessment Methodology with Application to Six Cities. Environment Department Paper No. 78. World Bank, Washington, DC.

Markandya, A., 2000. Employment and environmental protection. Environ. Resour. Econ. 15 (4), 297–322.

Markandya, A., Hunt, A., Alberini, A., Ortiz, R., 2004. The Willingness to Pay for Mortality Risk Reductions: An EU 3-Country Survey. Mimeo. University of Bath, Bath.

Mauskopf, J.A., French., M.T., 1991. Estimating the value of avoiding morbidity and mortality from food borne illness. Risk Anal. 11, 619–631.

Morgenstern, R., Pizer, W., Shih, J.-S., 2002. Jobs versus the environment: an industry level perspective. J. Environ. Econ. Manage. 43, 412–436.

Murray, C., Lopez, R. (Eds.), 1996. The Global Burden of Disease. Harvard University Press, Cambridge, MA.

Olsthoorn, X., Amann, M., Bartonova, A., Clench-Ass, J., Corfala, J., Dorland, K., Guerrrreiro, C., Henriksen, J., Jansen, H., Larssen, S., 1999. Cost–benefit analysis of European air quality targets for sulphur dioxide, nitrogen dioxide and fine and suspended particulate matter in cities. Environ. Resour. Econ. 14, 333–351.

Pearce, D.W., 2000. Valuing Risks to Life and Health: Towards Consistent Transfer Estimates in the European Union and Accession States, Paper Prepared for the European Commission (DGXI) Workshop on Valuing Mortality and Valuing Morbidity, November 13, 2000.

Pearce, D.W., Koundouri, P., 2003. The Social Cost of Chemicals: The Costs and Benefits of Future Chemicals Policy in the European Union. Report to the Worldwide Fund for Nature, Godalming, UK.

Porter, M., 1991. America's green strategy. Scientific Am. 264, 186.

Roediger-Schluga, T., 2004. The Porter Hypothesis and the Economic Consequences of Environmental Regulation. Edward Elgar, Cheltenham.

Rowe, R., Lang, C., Chestnut, L., Latimer, D., Rae, D., Bernow, S., et al., 1995. The New York Electricity Externality Study. Oceana Publications, New York.

RPA, 2001. Regulatory Impact Assessment of the EU White Paper: Strategy for a Future Chemicals Policy. Department of Transport, Environment and the Regions, London.

RPA, 2003. Assessment of the Impact of the New Chemicals Policy on Occupational Health. Final Report. CEC (DG Environment), Brussels.

RPA and Statistics Sweden, 2002. Assessment of the Business Impact of New Regulations in the Chemicals Sector. Report to the European Commission. CEC, Brussels.

Sprenger, R.-U., 1997. Environmental Policies and Employment. Organisation for Economic cooperation and Development, Paris.

US Environmental Protection Agency, 1997. The Benefits and Cost of the Clean Air Act 1970–1990. US EPA, Washington, DC. www.epa.gov.

Warner, R., Thompson, J., 2002. The New European Chemicals Strategy. Paper HSC/02/143. Health and Safety Commission, London.

World Bank, 1993. World Development Report: Investing in Health. Oxford University Press, Oxford.