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Integrated environmental health impact assessment: lessons from the INTARESE case studies

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Abstract

A series of case studies were undertaken as part of the EU-funded INTARESE project. Case studies were designed to apply, test and demonstrate methods for integrated environmental health impact assessment of seven different policy issues: urban road transport, housing, agricultural land use change, domestic water supplies, chemicals in household articles and products, waste management and climate change. This paper outlines the studies and reviews some of the key lessons learned. While the studies confirmed the feasibility and potential benefits of undertaking integrated assessments to guide policies likely to affect human health, they also highlighted some of the conceptual and methodological challenges that need to be overcome. Clear yet inclusive framing of the issue to be assessed was seen to be an especially important in order to limit biases in the assessment. Severe challenges are, however, faced in trying to identify the boundaries of complex policy issues, which cross traditional policy boundaries and touch upon the lives of many different stakeholder groups. Systemic issues of this sort also imply the need to develop realistic and dynamic scenarios, that reflect the way in which social and human, as well as environmental, systems respond to change. Modelling of intake (or exposure) under these scenarios needs to deal with the limitations of data and knowledge that exist, and thus has to rely on relatively simple and robust estimation techniques. Likewise, gaps in the available data on exposure-response relationships, and the lack of data on which to base preference weights, create uncertainties in the estimation and aggregation of health effects.

Introduction

Making effective policy decisions about the environment and health requires reliable evidence and information about the causes and consequences of human exposures to hazards. At root, much of this evidence derives from research endeavours, especially in epidemiology and toxicology. Relatively rarely, however, does policy-making draw directly on this science; instead, the science is translated into policy-relevant information via some form of assessment. Two forms of assessment are crucial in this respect. Diagnostic assessments are needed to evaluate the prevailing situation, and the magnitude and causes of any existing problems or threats. Prognostic assessments are needed to estimate the potential implications of future developments, such as new technologies or policies.

In the area of environment and health, the traditional paradigm for assessment has been that of risk analysis (Covello and Merkhofer 1993), and the main policy instruments have been environmental standards or guidelines, aimed for example at limiting the use of hazardous substances or practices, and ensuring that environmental contamination does not exceed specified limits. In general, this approach has served well as a basis for controlling individual hazards, such as toxic chemicals or carcinogens. Many modern threats to human health, however, are the product of more complex systemic risks (Klinke and Renn 2006), emerging from subtler but more pervasive environmental, social, economic and political forces, often at broad regional or global scales. Climate change is one stark example, but other recent health concerns such as bird flu, BSE, the 'obesity pandemic' and the health impacts of urban transport are also systemic in nature, in that they are rooted in the interdependent and interacting mechanisms of society, both past and present. Such issues cannot be easily addressed using unitary, short-chain methods of risk assessment. Instead, they need a much broader perspective, and must consider longer and more divergent chains of causality.

Over the last 10 years or so, integrated assessment (IA) has emerged as a powerful tool for such analyses (Rotmans and van Asselt 1996, Toth and Hizsnyik 1998), especially in support of environmental policies. Despite its obvious potential, however, applications to issues of environmental health (i.e. the health effects of environmental stressors and exposures) has so far been slow. To redress this, the EU-funded INTARESE (Integrated Assessment of Health Effects of Environmental Stressors in Europe) and HEIMTSA (Health and Environment Integrated Modelling and Toolbox for Scenario Analysis) projects have endeavoured to extend IA into the health area, under the banner of Integrated Environmental Health Impact Assessment (Briggs 2008). Their goal has been to develop, demonstrate and establish methods and tools for assessing health impacts of complex environmental stressors, in support of policy. In the process, a number of case studies have been undertaken, aimed at testing and illustrating the use of this approach, and showing also its potential and limitations. This paper draws on results from seven of these studies, carried out within the INTARESE project, to highlight some of the major lessons that have been learned.

Integrated environmental health impact assessment: the INTARESE approach

Concepts and principles

Integrated environmental health impact assessment can be defined as a means of assessing health-related problems deriving from the environment, and health-related impacts of policies and other interventions that affect the environment, in ways that take account of the complexities, interdependencies and uncertainties of the real world (Briggs 2008). Underpinning this approach is recognition that relationships between environment and health involve many-to-many relationships, such that single causes operate via multiple pathways and spread out to have multiple effects, and individual health outcomes derive from a multitude of interacting sources and agents. These relationships are represented by the concept of the causal chain – or more strictly 'web'. Assessment involves modelling the ways in which hazards propagate through this web, from distal sources, via different environmental media to exposure, and then the ways in which these translate into impacts on health. As such, assessment is concerned not with the direct, individual risks to health as a consequence of the inherent properties of a specific hazard (as in traditional risk assessment), but with the totality of health impacts across the population arising under a specified set of conditions (e.g. a policy scenario). Typically, these include many secondary impacts, arising both from the indirect (and often delayed) effects of the sources of interest and from the behavioural responses (e.g. risk avoidance and management strategies) of those involved. The models used in assessment thus need to take account of the environmental and social interactions and feedback mechanisms implied by the scenario being examined. Because the impacts usually comprise a range of different health outcomes, some of which may be beneficial and others adverse, assessment also implies the ability to aggregate the results into some form of synoptic measure of impact. While this might, in simple situations, be done by a metric such as the overall mortality rate or life expectancy, it is often more appropriate to use some form of value-weighted measure, such as disability adjusted life years (DALYs) or monetary value (Dickie and List 2006, Grosse *et al.* 2007). A further key feature of integrated environmental health impact assessment, which distinguishes it from traditional risks assessment, is that it is explicitly scenario-driven. Thus assessments involve the comparison of impacts under two or more predefined scenarios, typically representing some form of business-as-usual situation or the status-quo, and an 'alternative scenario' which depicts the world as it might be under a set of predefined assumptions.

How integrated environmental health impact assessment is done is likely to vary depending on the specific purpose and context of the analysis, and the resources available to support it. Four general steps in the process can, however, be defined: issue-framing, design, execution and appraisal (Figure 1). The first involves defining and agreeing the question to be assessed, and delimiting the scope of the assessment – e.g. which stakeholders need to be involved, which factors should be included, and the bounds and structure of the system under analysis. In the design stage, this broad conceptual model of the issue is translated into a detailed protocol for assessment, including a clear specification of the scenarios involved, the variables to me measured or modelled, the data and models to be used, and the impact indicators to be assessed and reported. Execution comprises the formal (and generally quantitative) analysis: modelling of exposures under the different scenarios, selection or generation of the appropriate exposure-response functions, computation of health impacts, and weighting and aggregation of the results. In the last stage, appraisal, these results are reviewed, checked against the initial aims of the assessment, and interpreted to provide an agreed statement of the outcomes (e.g. a ranking of the policy options).

The INTARESE case studies

While the INTARESE approach described above builds on existing techniques of health impact and integrated (environmental) assessment, it also substantially extends and formalises the concepts and methods involved. The practicability of integrated environmental health impact assessment, and the challenges involved in trying to apply it, are therefore largely unknown. Nor are ready-made illustrations of its application yet available, although one recent and innovative example is the assessment of the potential health effects of oil and gas development in Alaska (Wernham 2007).

For these reasons, case studies were conducted in the INTARESE project, aimed at testing the approach (and learning from the experience), demonstrating its utility, and providing worked examples to guide future assessments. Seven issues were selected (Table 1), representing different types of assessment from different policy perspectives. Those on transport, agriculture and wastes all focused on economic sectors, and considered how policy-induced changes fed through to affect health. Those on domestic water and chemicals in household articles and products were defined in terms of the exposure medium; that on housing in terms of the physical setting for exposures (the home). The case study on climate took as its starting point the environmental phenomena (temperature and ozone) and followed these through to their health impacts. Likewise, case studies were targeted at different scales of analysis, from local (e.g. individual cities) to national and European.

In order to provide a clear and consistent framework for the case studies, guidelines were established at the start, which all studies were required to follow. These set out the steps in the analysis (as in Figure 1), and the way in which results should be reported. All case studies thus started with issue-framing, during which a conceptual model was built on the basis of a ‘full chain framework’ specified in the guidelines. This was then expanded into a formal protocol, which was reviewed and approved by a small oversight group. Execution, however, was conducted separately by the different study teams, using the data, methods and models that they regarded as most appropriate, and which were feasible under the prevailing circumstances. This latter consideration necessitated a number of pragmatic choices. For the most part, for example, data were drawn from routinely collected and publicly available sources; opportunities for new data collection or purchase of proprietary data sets, were limited. The models used were also generally those that were available within the institutions of the study teams concerned, or could be obtained at little or no cost. In some instances, also, adjustments to the protocols of individual case studies had to be made as they progressed, to match ambition to practicability. Thus, while a number of different hazards (agents), pathways and health outcomes were identified and assessed in each of the case studies, it was not always possible to follow up all the factors (e.g. sources, agents, pathways and health outcomes) identified at the issue-framing stage. Some case studies ended up focusing on a smaller study area than originally intended, in order to optimise data availability and resource use. For all these reasons, the findings of the case studies can be seen as indicative of the challenges and problems of doing integrated environmental health impact assessment, but the studies do not wholly reflect the decisions that would need to be made – and the issues likely to be encountered - in doing an assessment 'for real'.

Table 1. Summary of INTARESE case studies

Case study	Scenario (and study area)	Stressor	Health outcome	Findings	Limitations and uncertainties	Other comments
Transport	*Prognostic:* Traffic zoning/congestion charging (London); Traffic zoning/emission controls (Rome); Traffic management/circulation planning (The Hague) Congestion charging/modal shift (Helsinki)	PM_{10}, NO_2; noise	Respiratory disease; cardio-vascular disease	Large reductions in emissions (10-30%); moderate reductions in concentrations (1-2%); small gains in health (ca 450 years per 100,000 people) from air pollution; ca. 2% reduction in noise annoyance.	Excludes: non-exhaust emissions (e.g. tyre wear); effects of physical exercise; non-residents (outside city); specific effects on susceptible sub-groups	Upstream effects greater than downstream (health) effects - vital importance of analysing full chain; effects vary geographically; time duration of policy crucial.

Housing	*Prognostic:* Energy efficiency/improved thermal insulation (UK)	Cold and radon	Cardio-vascular disease; cancer	Small reductions in risk due to improvements in indoor temperature, almost offset by increased risks of cancer due to radon exposures, giving overall impact of < 0.4 days of life gained/person.	Excludes other pathways of effect - e.g. via outdoor air pollution; lack of detailed data on building characteristics and ventilation means that modelling of exposures subject to substantial uncertainties	Results depend on 'hazard multiplier' used to weight effects of cardio-vascular disease versus cancer.
Agriculture	*Prognostic:* Policy-driven land use change/PRELUDE (UK & Macedonia/Thessaly)	Pesticides (herbicides, insecticides, fungicides)	Breast, kidney. Pancreatic, prostate cancer; leukaemia; non-Hodgkin's lymphoma; congenital anomalies; stillbirths	Moderate reduction in pesticide usage (<10 kg/ha) leads to small reduction in cancer risk (ca. 230 cases/year) and congenital anomalies (130/year); stillbirths reduced by ca. 30/year.	Data on pesticide usage are highly aggregated and modelling of usage rates has large uncertainties; pesticide usage provides poor proxy for exposure; data on exposure response functions sparse; land use scenario is too generalised	Upstream effects (on usage) greater than downstream effects on health; effects vary geographically, depending on land use. Need to take account of impending changes in permitted pesticides, and effects due to climate change.

		Pesticides (active ingredients)	Cancer	Moderate reduction in pesticide usage (<4kg/km2) leads to small reduction in risk of cancers (1/300,000)	to provide sound basis for modelling changes in farming practice; no account taken of impending (independent) policy-induced changes in pesticide usage	
Water	*Diagnostic:* Current versus counterfactual of no contamination (England & Wales)	THMs (UK)	Small for gestational age	Moderate excess risk of low birth weight (ca. 2-2.5%); equivalent to 1600-1700 cases/year	Lifelong effects of low birthweight (e.g. increased risks of cardiovascular illness) not considered.	Large geographic variations in risk, reflecting variations in THM concentrations
			Bladder cancer	Moderate excess risk of cancer (ca. 6-7%) in males, but low in females (ca. 0.3%); equivalent to ca. 580 cases/year (1400+ DALYs)	Data on exposure-response functions derived from pooled analysis across several countries; relevance to study population is unclear; differences in oxidation state (and thus toxicity) or arsenic not allowed for	Risks in males ca. 20 times those in females; marked geographic variability in THM concentration and risk

		Arsenic		Low levels of arsenic in drinking water imply small excess cancer risks (<0.1% - equivalent to ca. 9 cases/year, or 24 DALYs)	Large uncertainties around risk estimates, due to sparseness of epidemiological data	Risks 2-3 times higher in males than females and mainly in 65 years+
		Nitrates	Methaemo-gloninaemia	Extremely low excess risk (ca. 2%), equivalent to 1 case/10 years)	Large uncertainties in data on water consumption by children	
Chemicals in household articles and products	Controls on use in toys and cosmetics (EU)	DBP	Reproductive	No exceedances of no observed adverse effect level, so no detectable health risk	Use of effect level extrapolated from animals; uncertainties in data on concentrations in products; exclusion of bystander exposures	Need to apply Monte Carlo methods to model distribution of (and uncertainties in) exposures and risk estimates; need for improved (population-specific) severity rates for DALY calculation
	Controls on use in adhesives and spray paints (Serbia)	Toluene	Neurological	Small risks, equivalent to 73 DALYs across Serbia	Exclusion of products with very low concentrations; effect level extrapolated form human volunteeers; uncertainties in usage data and concentrations in indoor environment	

	Prognostic: Classification as category 1 carcinogen under EU Directive 67/548/EEC	Formalde-hyde	Eye irritation; masopharyngal cancer	Moderate excess risk, equivalent to 3470 DALYs across EU	Exclusion of some health endpoints (e.g. sensitisation); uncertainties in reliability of no effect level (based on human volunteers); gaps in data on usage and concentrations; uncertainties in severity weighting for ocular effects	
Waste	*Summative:* Impacts of current waste management strategies (Italy/UK)	PM, NO_2	Respiratory disease	Ca 1-1.25 days of life lost per person in each country, representing a total of 3000-4000 YLL across the 3 countries	Limited number of studies on which to base exposure-response function fro dioxin; uncertainties in effects of changes in technology on emissions from incinerators; exclusion of exposures during transport and storage	Important differences in risk over time due to changes in technology and emission control; long-term risks may be associated with closed landfill sites; need to include other links in the waste management system (collection, transport, storage) for overall impact assessment.
		Dioxin	Cancer	Ca. 3800 additional cases in three countries, mainly in Italy and UK, and mainly (90%) due to past (pre-2001) exposures from old incinerators		

		Unknown	Congenital malformations	Ca 100 additional cases of low birth weight, and 5-6 cases of congenital anomalies across the 3 countries, each year	Use of distance as proxy for exposure; very limited range of epidemiological studies on which to base exposure-response functions; lack of detailed information on landfill characteristics, waste composition, or emissions; exclusion of exposures during transport; exclusion of closed landfill sites.	
Climate	*Diagnostic:* Current burden of heat and cold *Prognostic:* IPCC climate change/SRES A2 & B1 (Rome, London, Helsinki)	UV radiation	Skin cancer	Number of skin cancer cases in the 2 cities predicted to rise from 10,400 in 2001 to 14-15,000 by 2030, and 15-16,000 by 2050 (largely - 80% - due to BCC; deaths to rise from 270 to 380 and then to 450 (mainly - 90% - due to melanoma); equivalent to ca. 9000 DALYs/year, compared to 7000 in 2001.	Baseline disease rates derived from other regions; incidence-mortality ratios derived from Australian data and assumed to apply in Rome and London; uncertainties in projections of UV radiation	Risks somewhat greater in males than females, and vary over time (peaking in 2030) as effects of past exposures work through the population.

		Heat, cold	Heat and cold attributable mortality	Estimated mortality due to heat in the baseline year (2001) was : London 286 (95% CI 23, 574), Rome 316 (6, 575); and Helsinki 8 (1,16)	Estimates are highly sensitive to dose-response functions and thresholds assumed in the model. Dose-response functions are heterogeneous between populations and over time.	Lack of information on the effectiveness of policy measures to reduce current heat and cold burdens at the population level.

The lessons learned

Details of the individual case studies are being published elsewhere. Table 1 summarises the main findings. As this implies, the disparate nature of the case studies, and differences in the way in which they were ultimately conducted, inevitably means that many of the experiences encountered were specific to individual studies. Indeed, one of the major conclusions from the analysis was that many elements of doing an integrated environmental health impact assessment are circumstantial, and thus cannot readily be systematised through the provision of generic rules or tools. Nevertheless, a number of more general, cross-cutting lessons did emerge, which – even if they require different solutions in different cases – help to highlight some of the key conceptual and practical problems involved in assessment.

Issue-framing

Issue-framing is clearly crucial in any assessment, for it determines not only the question that is being addressed, but also to a large extent the answers that may be obtained and the stakeholders who might thus be affected. Framing was done in these case studies by constructing a 'system model', setting out the key factors that were considered relevant and the causal pathways and connections between them. Inevitably this poses problems, for real-world systems are invariably open and have diffuse and porous boundaries. The limits to any issue, and questions of what is important and what is not, are therefore ambiguous. In the case of more traditional forms of risk or impact analysis, the difficulties are often reduced to some extent because the organisations which commission the study do so within the priorities and limitations of their own authority or interest. In the case of integrated assessments, however, the problem is amplified, both because the issues themselves are more open and complex, and because a wider (and less closely defined) range of stakeholders can claim to have interests (Briggs in press). As a consequence, neither the bounds of the issue, nor the criteria by which to decide what is or is not important, are predefined, but have to be deduced as part of the assessment.

Doing this without any underpinning and pre-existing framework or set of rules would be a chaotic and fraught process. It might also lead to major biases due to the differing strength of different voices in the discourse. To avoid this, here, the conceptual models were built on the basis of the full-chain framework, as shown in Figure 2. The purpose of this was to help structure the issues in a more balanced and consistent way, and also to act as a check-list which could ensure that crucial causal variables and links had not been omitted.

To a large extent, the full-chain framework was found to serve these roles well, for it provided a point of convergence as well as a stimulus for thought. It was nevertheless apparent that it did not fit all issues equally. In both the climate and housing case studies, for example, the focus was on the latter parts of the chain (from the hazard onward) so that the notion of sources, releases, transport and transformation were of limited relevance. In every case, issue-framing ultimately broke away from the strict format of the full-chain framework to some extent, as the conceptual models were progressively extended and revised, in order to incorporate more contextual factors, feedback mechanisms and secondary or contingent effects.

Deeper conceptual questions also emerged during issue-framing. One of these concerns the definition of 'environment'. This intrinsically sets the boundaries for what should be covered in integrated *environmental* health impact assessment, but little consensus exists. Different study teams therefore, made different choices, some of which might be seen to have created biases or lacunae in the assessments. Road accidents, for example, were excluded from the transport study, and poisonings from the agriculture case study, despite the fact that these account for a substantial part of the health burden from these two sectors and are included in environmental burden of disease assessments by the World Health Organisation (Valent *et al.* 2004). The lesson is that any framework represents a specific (though not always explicit) world view, which is liable to skew the way in which the issue is conceived. The ideal solution to this is undoubtedly to involve stakeholders in the process of issue-framing (the discourse of design as shown in Figure 1) – and in doing so, to ensure that they have the chance to challenge, adapt and formulate the underpinning framework. It also needs emphasising that this discourse should not be restricted just to policy-makers or other professionals, but needs to include all stakeholders with legitimate interests in the issue. How to do this is problematic, because of the large number, and varied experience and authority, of the stakeholders concerned. It is certainly difficult for one-off and ad hoc assessments, like those done here, for considerable time and effort are needed to identify stakeholders, build their trust and engage them effectively in the assessment process (Briggs in press).

Scenario development

As already noted, integrated environmental health impact assessments are scenario-driven, in that they are concerned with the health impacts which arise under one set of circumstances, compared to those that would arise under one or more others. Scenarios thus define the parameters of the assessment, and thereby have a fundamental influence on its outcomes and meaning.

In real-world assessments, some form of scenario will often be inherent in, or implied by, the issue that motivates the study. A new policy proposal, for example, will imply comparison of health status under the policy compared to that without it. Likewise, concerns about the health impacts of an existing policy or technology would suggest assessing the health consequences compared to a counterfactual situation in which the policy of technology did not exist. In the case studies done here, scenarios had to be devised for the purpose. The scenarios selected for analysis are summarised in Table 1.

Application of these scenarios revealed a number of important issues for integrated assessments. Notably, several of them failed to produce significant health effects; modelled impacts were well within the margins of error of the assessment, and in some cases close to zero. One reason was that, in many cases, there was marked and progressive dilution of impacts along the causal. In the case of road transport, for example, a moderate change in road traffic volumes led to a more-or-less proportional change in pollutant emissions, but a much reduced and localised change in ambient concentrations, yet smaller changes in exposures and thence to very small reductions in morbidity or mortality. Likewise, in the agriculture case study, moderate changes in cropping patterns and farming practices, implied by the scenarios, resulted in smaller and localised changes in pesticide usage, and ultimately very limited changes in exposures and health effect. To some extent, these results are not surprising, for real-world systems are subject to various thresholds and internal checks that limit their responses to change, while pollutants are diluted as they pass through the environment. As a consequence, many interventions are likely to see some degree of effect dilution between source and health impact. Given that, not all scenarios are likely to merit full-scale integrated assessments, and some form of screening, to judge the approximate magnitude of effects, would seem essential. In some cases, this might be achieved qualitatively, by some form of expert elicitation. In many instances, however, it requires a form of rapid assessment, using simple estimation models and tools.

At the same time, other, artefactual reasons for the limited impacts arising from these scenarios need to be recognised. In both the agriculture and climate studies, for example, the scenarios were broad in scope and coarse in spatial resolution, yet the study areas were relatively small. This dilutes estimated impacts because environmental changes are averaged across large areas, thereby masking hotspots. Because these often coincide with areas of higher population density - e.g. due to urban heat island effects in the case of climate, or the location of intensive farming near to urban areas in the case of agriculture – the consequence is to under-estimate risks.

The temporal scale of the scenarios was also an unexpected yet profound issue in several of the studies. For most prognostic assessments, impacts were estimated for specific target years (e.g. the current year for household chemicals, 2035 for agriculture, 2030 and 2050 for climate). Selection of these snapshot years can be crucial, for they can greatly distort the magnitude and ranking of the health effects, depending on the latency and duration of the impacts. In particular, both early acute effects (including death) and delayed effects (e.g. many cancers) might be missed. Inter-generational impacts are also likely to be ignored. A more realistic approach, therefore, is to analyse impacts as a continuous process, extending many years into the future. This was done, here, in the case of the wastes case study, which explored the health consequences of policy and technological changes in incineration during the 1980s up to 2050, under different scenarios. The results showed that both the timing and magnitude of peak impacts, and their overall duration, varied substantially depending on assumptions about how effective the policy measures were in reducing emissions, and the latency functions following initial exposure and after exposure ceased. Defining the true time-scale over which the scenario needs to be run, and modelling the temporal variations in impact across this time, are therefore important yet challenging issues. To date they seem to have received little attention; they need further research.

These time-varying patterns of health response hint at a further, and equally neglected, issue. Real-world systems are not conditionally stable, switching instantaneously from one steady state to another, but are to a large extent incrementally adaptive. These adaptations operate not only within the environment, but also through social (e.g. population distribution and structure) and behavioural changes. Together they may cause substantial changes in exposure and susceptibility. These responses, in turn, may trigger other, secondary effects, some of which may feed back to alter the original stimuli for change. In the case of integrated assessments, these dynamic and secondary changes are of considerable importance for they may fundamentally affect the overall system response, either amplifying or damping down the ultimate impacts. Nor are these changes tightly confined in space or time, but instead may extend over large areas and take many years (or even centuries) to play out.

These adaptive and evolutionary changes pose important challenges for scenario development. It has often been emphasized that scenarios are not intended to be explicit predictions or forecasts of what will happen, but instead merely represent a coherent and plausible description of what might take place under a specified set of assumptions. Traditionally in policy analysis, therefore, scenarios have been conceived as no more than a given set of conditions or outcomes, as would be achieved if policy targets for emissions, or environmental quality had been met. These are what Carter *et al.* (2001) refer to as exogenous scenarios. Greater realism demands endogenous scenarios – i.e. ones that emerge from within the system under study in response to specified changes in the system inputs. While scenarios of this sort are used to some degree in integrated environmental assessments, especially in the climate field, their use in assessment of health impacts is almost unknown.

Without doubt, this weakens the validity of such assessments, for it fails to account for many of the most important and long-lasting responses that occur, most notably those that operate via social and human behaviours. Developing and using adaptive scenarios nevertheless poses enormous challenges. It means, for example, that scenarios can rarely be predefined, separate from the rest of the assessment process. Instead, scenario modelling, exposure assessment and health impact assessment have to be an intricately interwoven whole, each dependent on the other. The distinction between scenarios and models thus begins to become blurred. It also means that it is often inappropriate simply to use pre-existing scenarios as a basis for integrated environmental health impact assessment, for these may differ considerably in scope, spatial resolution and timescale from those needed to evaluate health impacts. The implication is that much more effort needs to be given to scenario development if integrated environmental health impact assessments are to be truly informative.

Intake assessment

The full-chain framework shown in Figure 1 implies that health impacts can be assessed according to the following, general algorithm:

$$HI = \sum_{j=1}^{n} \left[S.R.D.C.A.Z.P.V \right]_j \qquad \text{[Equation 1]}$$

where:

S is source intensity (mass.length^{-2})

R is release efficiency (dimensionless)

D is environmental dilution rate – e.g. due to dispersion, transformation and deposition for the specific medium and micro-environment (length^{-1})

C is duration of human contact (time)

A is absorption efficiency (dimensionless)

Z is hazardousness (e.g. toxicity) of the agent - i.e. the risk of a given effect per unit of intake ([time.mass/length^{-3}]$^{-1}$)

P is exposed population (individuals)

V is preference weight attached to the health effect (cost.individual^{-1})

HI is value of the overall health impact (value units, e.g. DALYs, Euros)

Summation is across all relevant (j) combinations of agents, media, micro-environments, health outcomes and population subgroups.

The first five terms in this equation represent the links from source to intake, while the latter three cover the further link to health effect. Progressive multiplication of the earlier terms in the equation (S,R,D,C) also produce, respectively, emissions, concentration and exposure.

For many epidemiological studies, and for the purposes of health surveillance, the aim is to measure intake (or a close proxy such as exposure) directly – e.g. through biomonitoring or personal exposure monitoring. For health impact assessments, however, direct measurement has limited value for several reasons. First, the data themselves tend to be scarce and unrepresentative of the wider population (or specific population subgroups) under consideration. Second, they have often been generated for other purposes, and thus do not represent the specific attributes of interest. In particular, for health impact assessment, the measures of intake (or exposure) used must match those used to characterise the hazardousness of the agent being assessed. Third, for many assessments, estimates are needed of intake either in the future or under theoretical (e.g. counterfactual) conditions. This clearly cannot be achieved with monitoring. In almost all cases, therefore, modelling is essential to estimate intake, either by interpolation from whatever measured data exist, or through some form of process model or simulation. In this context, major constraints tend to be the availability of the knowledge to parameterise the models and the input data needed to run them.

Such was the situation in the seven case studies undertaken here. Adequate measurement data for the agents, time periods and populations of interest were not available. Instead, they had to be estimated indirectly. For exposures to THMs, arsenic and nitrates in domestic water, for example, measured data on concentrations in tap water at small sets of sample homes were first extrapolated to the whole water supply areas, and then adjusted using crude 'exposure factors' to reflect the effects of different consumption and bathing patterns. For particulates and NO_2 emissions from both road traffic and incinerators, exposures were estimated using dispersion models. The reliability of the modelling in both cases was constrained by weaknesses and inconsistencies in the input data: road traffic data for Rome were only available at the city-level, though the policy was being implemented more locally; emissions data for incinerators were based on national averages yet applied to individual sites. In the agriculture case study, the limitations were even greater, for little is known about how pesticide residues diffuse through the environment and the resulting levels of exposure. Data on pesticide usage are also sparse and generally highly aggregated. For the UK study, therefore, proxies for exposure were derived by disaggregating county-level statistics on usage rates to the local (ward) level using GIS techniques. This had implications for later stages in the analysis, for it meant that exposure-response functions had to be based on epidemiological studies that had employed similar, usage-based exposure metrics. These were both few in number and broad in terms of the pesticide groups that they used. Similar problems arose in relation to health effects of landfills and cancers associated with incinerators in the waste study. For these, the only reliable epidemiological studies had employed simple measures of residence within a specified distance of the source as a proxy for exposure. The same distance measures therefore had to be used, likewise, in the assessment, even though this meant that the effects of changes in emission rates and distribution around the sites were essentially ignored.

Inevitably, therefore, estimation of intake or exposure in these case studies suffer from major uncertainties. The exposure metrics themselves were often somewhat crude and unspecific, and were rather insensitive to the policy and environmental changes being examined. Little or no account could be taken of the local and circumstantial factors that affect exposures and intake: for example, micro-environmental variations in pollutant concentrations, time activity patterns or absorption rates. Considerable generalization and attenuation of the true exposure distributions within the study populations therefore inevitably occurred, and there was little ability to detect variations between different populations sub-groups. The lack of independent data meant that errors in the exposure estimates made in these case studies could not be accurately quantified, but sensitivity analysis in the water case study, for example, suggested that errors due to variability in the monitoring data were substantial, and probably second only to the exposure-response functions as a source of error in the impact assessment.

The implications of these uncertainties in the exposure data are undoubtedly important. At worst, they prevent any serious attempt to model or detect changes in exposure under the scenarios of interest. More generally, they lead to significant uncertainties in the estimates of health impact. Overcoming these deficiencies is one of the most fundamental and time-consuming tasks in assessment, and often involves considerable ingenuity. Solutions do not always (and may rarely) lie in the development or use of more complex exposure models, for the data needed to run these remains sparse. Instead, the main need is to devise and validate simple estimation methods that can be applied despite the paucity of data. Likewise there is a need to compare and cross-validate existing approximation methods and exposure proxies, under different situations (e.g. in different areas and at different spatial and timescales), and to develop and test methods for converting between different exposure metrics and markers in order more easily to pool data from different studies. To a large extent such needs run against the grain of normal research, which is geared towards greater sophistication of technique rather than increased robustness.

Health effect estimation

As indicated in Equation 1, above, health impacts represent the product of intake (or exposure), the hazardnousness of the agents concerned, the size of the exposed population, and the severity weights associated with those health effects. In principle, information on the first of these elements (hazardousness) can be obtained from toxicological studies. For various reasons, however, such data tend to have limited utility. In many cases, for example, studies continue to be based on the concepts of thresholds, so that results are reported in terms of a ‘safe’ or tolerable level, such as no-observable adverse effects level (NOAEL) – the concentration below which detectable harm should not occur. Without further assumptions and manipulation, this does not give information on the shape of the relationship between exposure and outcome, that can be used to estimate population-level impacts. In many cases, also, toxicological studies rely on animal models, or studies of very small numbers of human individuals in controlled experimental settings, making it difficult to extrapolate the findings to real-world human populations. As a consequence, epidemiological studies typically provide the main basis for assessment. These have the advantage of usually being based on much larger (and more representative) study groups, and on exposures which better reflect the range of conditions seen in the real-world. They also usually provide information on the slope of the association between hazard and health effect, in terms of a dose- or exposure-response function. Nevertheless, epidemiological studies suffer from problems of their own. One of these is the common use of indirect measures of intake or exposure, which do not translate readily into health impact assessments. Another has been lack of consistency in design between different studies (including methods and metrics of exposure analysis, choice and definition of health outcomes, and characteristics of the study population), which make often it difficult to pool or compare data from different studies. Both epidemiological and toxicological studies also tend to provide relatively little information on how hazardousness varies as a result of differences in susceptibility across the population. Data are often reported for different genders and age groups. Dose-response functions for different socio-economic groups, however, are rarely estimated, despite evidence that susceptibility may vary with socio-economic status (Forastiere *et al.* 2007, Levy *et al.* 2002).

The assessment methodology developed in INTARESE provides two approaches for estimating exposure-response functions, and thus for deriving measures of hazardousness. Where possible, systematic reviews of the available studies should be done using formal procedures to weight and combine estimates from different studies; where the evidence for this is insufficient, expert elicitation methods should be employed. Here, attempts to undertake systematic reviews met with varying degrees of success. For air pollution impacts (e.g. related to transport and wastes) and UV radiation, the situation was relatively straightforward, since several recent and comprehensive reviews have been undertaken, covering the health outcomes of concern. The evidence for other exposures, however, was much more limited. In the case of disinfection byproducts (THMs) in domestic water, for example, the exposure-response function for bladder cancer was derived from a published pooled analysis that was based on only six studies; that for birth weight was obtained from a purposely-designed meta-analysis of just five studies. Both were considered to involve substantial uncertainties, not least because total THMs provides rather poor characterisation of the individual contaminants that might be responsible for health effects. For nitrates and bladder cancer, the available evidence was so equivocal that no assessment of health impact was considered feasible. As a basis for assessing health impacts from pesticides in Great Britain, a published systematic review was used. This provided estimates from a large number of studies, but differences in the classes of pesticide, associated health outcomes and study populations meant that these could not be directly compared. The exposure-response function thus had to be based on only a small number of relevant studies, implying high levels of uncertainty. In other cases (e.g. pesticides in Greece and household chemicals), the lack of epidemiological studies necessitated the use of toxicological information, although in many cases these were based on animal studies, and could not easily be converted into suitable metrics for impact assessment.

The uncertainties inherent in almost all these data on dose- or exposure-response functions (or toxicity) merits emphasis. In several of the case studies they were considered to represent the main source of potential error in the assessments. As already noted, they required the use of relatively poor exposure metrics. Lack of suitable (or reliable) measures also meant that some potential effects were ultimately excluded from consideration in several cases. While this may seem justifiable, it needs to be recognised that, to do so, implies that the excluded effects have exactly zero toxicity, for they are not only being given no weight in the assessment, but are also not being considered within the confidence intervals attached to the estimated health impacts.

As Equation 1 indicates, data on the size of the exposed population are also crucial in assessing health impact. Small-area population counts are readily available for most countries from national censuses, and these form a basis for estimating the population-weighted distribution of exposures in many cases. The geographic resolution of published census data is nevertheless variable, and often relatively coarse, limiting the ability to detect local variations in exposures (Briggs *et al.* 2007a). In recent years, finely gridded (~100 metre) estimates of population density have been mapped across Europe (Briggs *et al.* 2007b; Gallego 2009), which can thus be intersected with data on pollutant concentrations to give estimates of exposure. For prognostic studies, however, population projections are essential; methods to produce these are not wholly consistent or reliable, and the assumptions involved for longer-term projections (as implied by the need to consider impacts far into the future) involve substantial uncertainties, especially regarding migration.

Estimation of the preference-weights (*V* in Equation 1) needed to compare and aggregate impacts on different health outcomes posed even greater problems. The weights concerned are intended to adjust health outcomes according to the importance of their impacts – e.g. depending on their degree of disability, duration (age of those affected), life years lost, or financial cost. Various techniques for deriving weights have been developed, often through some form of expert elicitation method or public survey; various different measures of impact have thus been derived (Hofstetter and Hammitt. 2002). Systematic surveys aimed at obtaining weights for different health effects and population groups, however, have rarely been done. For integrated assessments like those done here, this creates inevitable difficulties because of the need to weight and combine effects across a large number of often diverse outcomes. Only three of the case studies therefore attempted to compute weighted measures of health outcome, and these all used disability adjusted life years (DALYs). One of these (climate) drew on weights from a Dutch study of 53 diseases (Stouthard *et al.* 1997); the other two (water and household chemicals) used a (somewhat derivative) Australian study by the Victorian Government Department of Human Services (2005). How appropriate these weights are to the study populations considered here is open to debate. Considerable difficulties were encountered, especially, in applying the weights to secondary health effects (e.g. arising from low birth weight), and to select appropriate rates of discounting over time. The uncertainties which thereby arise may be substantial, though it seems likely that compensation occurs to some extent, so that the errors partially cancel out when a large number of outcomes are involved. Nevertheless, methods and data for preference weighting require considerable further development and testing if they are to be applied systematically in integrated assessments of this sort.

Conclusions

The case studies undertaken in the INTARESE project have helped to demonstrate the possibilities and the problems in attempting to carry out integrated environmental health impact assessments. Many of the problems encountered derive largely from limitations of data and knowledge, uncertainties in modelling, and the difficulties in defining bounded systems, and realistic scenarios, within which to conduct the assessments. Overlaid on these, however, are multiple layers of ambiguity that arise because of the (often unseen) differences in concept and language between the different, contributory fields involved in integrated assessments.

In the presence of these difficulties, a clear, conceptual template for assessment, and rigorous structure for analysis, is shown to be an imperative. The full-chain framework proved valuable in this respect, while the four-stage analytical process, with its emphasis on issue-framing, was likewise found to be effective. Access both to guidance on how to plan and carry out an integrated assessment, and tools for analysis, is also crucial if assessments are to be done efficiently and with some degree of consistency. Perhaps the strongest lesson to come from the case studies, however, is the need for adaptability. There is certainly much that is common in integrated assessments, and these elements are amenable to some form of standardisation in practice. They can thus make use of ready-made assessment protocols and tools. Much, however, is inevitably circumstantial: each assessment brings its own, unique requirements for information and methodology, and faces its own specific problems of data, understanding and technique. Dealing with these demands flexibility and inventiveness. It also demands a pragmatic and sensible approach to uncertainty. All assessments suffer from uncertainty. In integrated assessments, these uncertainties are almost invariably inflated because of the added breadth and complexity of the issues being considered. On the one hand, therefore, it is important to attempt to quantify and track these uncertainties, and report them as part of the results. On the other hand, uncertainty cannot be used as a reason to exclude factors from consideration, or *in extremis* not doing an assessment at all. Avoiding things that we do not fully understand does not reduce uncertainty, but merely hides it in an intellectual vacuum. Success thus depends on the ability to assess in the presence of uncertainty. In this context, the need is not for more sophisticated techniques, which inevitably bring higher data demands. Instead it is often for simpler robust tools – including robust estimation methods and screening models, as well as expert elicitation methods - that can be adapted to the situation. Integrated environmental health impact assessment, in other words, is both an art and a science: the art of design and of doing the best one can with whatever is available, but the science of estimation on the basis of the best evidence that can be obtained.

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Captions

Figure 1. The process of integrated environmental health impact assessment

Figure 2. The full-chain framework.

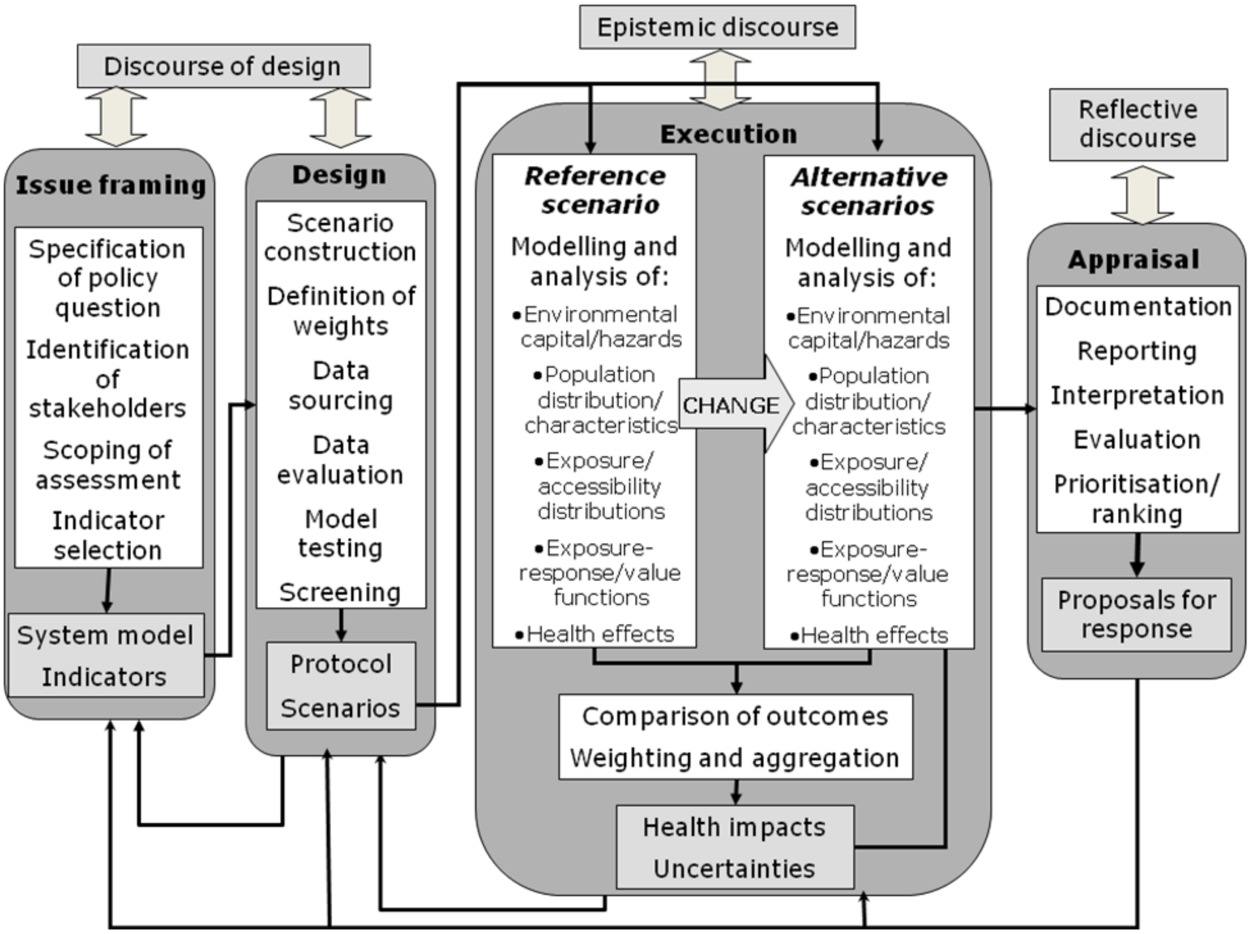
Discourse of design
Epistemic discourse
Reflective discourse
Issue framing
Specification of policy question
Identification of stakeholders
Scoping of assessment
Indicator selection
System model
Indicators
Design
Scenario construction
Definition of weights
Data sourcing
Data evaluation
Model testing
Screening
Protocol
Scenarios
Execution
Reference scenario
Modelling and analysis of:
•Environmental capital/hazards
•Population distribution/ characteristics
•Exposure/ accessibility distributions
•Exposure-response/value functions
•Health effects
CHANGE
Alternative scenarios
Modelling and analysis of:
•Environmental capital/hazards
•Population distribution/ characteristics
•Exposure/ accessibility distributions
•Exposure-response/value functions
•Health effects
Comparison of outcomes
Weighting and aggregation
Health impacts
Uncertainties
Appraisal
Documentation
Reporting
Interpretation
Evaluation
Prioritisation/ ranking
Proposals for response

Figure 1

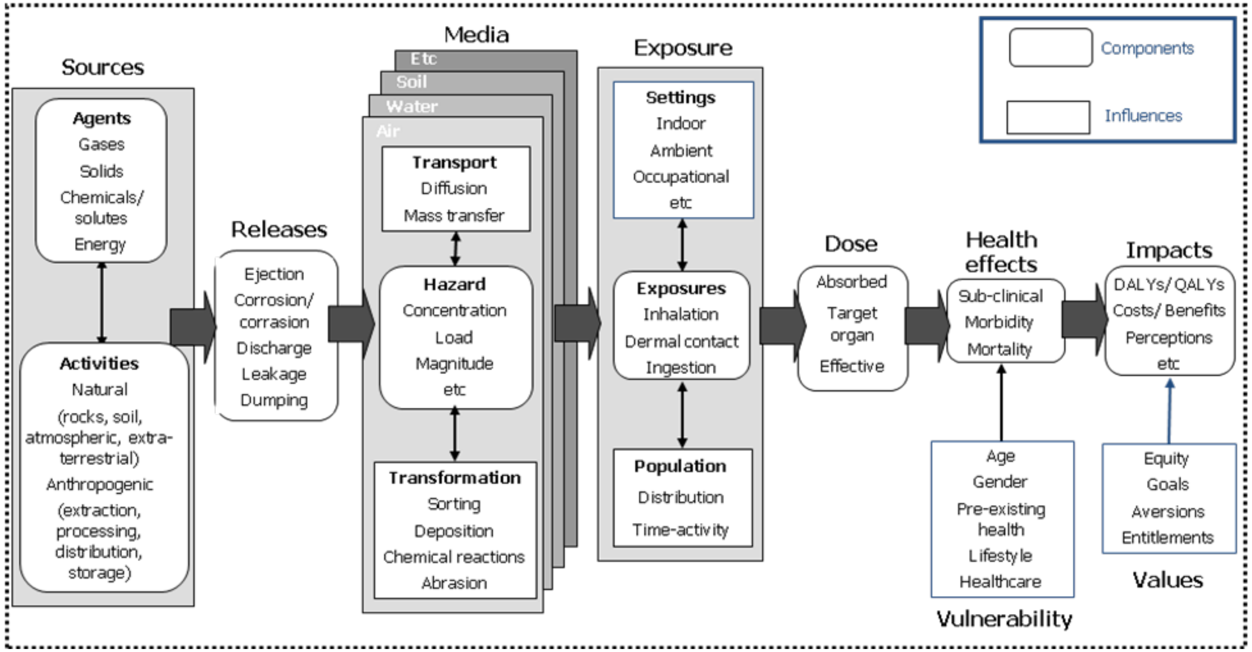
Sources
Agents
Gases
Solids
Chemicals/ solutes
Energy
Activities
Natural
(rocks, soil, atmospheric, extra-terrestrial)
Anthropogenic
(extraction, processing, distribution, storage)
Releases
Ejection
Corrosion/ corrasion
Discharge
Leakage
Dumping
Media
Etc
Soil
Water
Air
Transport
Diffusion
Mass transfer
Hazard
Concentration
Load
Magnitude
etc
Transformation
Sorting
Deposition
Chemical reactions
Abrasion
Exposure
Settings
Indoor
Ambient
Occupational
etc
Exposures
Inhalation
Dermal contact
Ingestion
Population
Distribution
Time-activity
Dose
Absorbed
Target organ
Effective
Health effects
Sub-clinical
Morbidity
Mortality
Age
Gender
Pre-existing health
Lifestyle
Healthcare
Vulnerability
Impacts
DALYs/ QALYs
Costs/ Benefits
Perceptions
etc
Equity
Goals
Aversions
Entitlements
Values
Components
Influences

Figure 2