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Air Quality Outcomes in pollution regulation: strengths, limitations and potential

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Steve Killeen

Steve Killeen

Head of Science

Executive Summary

With air quality still a critical issue, for the UK and globally, the Environment Agency is charged with finding practical solutions to complex regulatory issues in England and Wales. The current approach adopted for regulating pollutant emissions to atmosphere is based on limiting emissions to the lowest levels feasible and affordable, and ensuring that these levels do not add significantly to the overall atmospheric burden of pollutants. Such an approach can be straight-forward to implement, and straight-forward to assess for compliance, and has proven invaluable in helping to drive a massive improvement in air quality over recent decades. Further reductions are becoming more difficult to achieve, however, and proportionately more expensive. There is therefore an increasing need to show that any further improvements regulators may demand are proportionate to the pollution threat, and are efficiently targeted at the most significant pollution sources. In light of this need, this review looks at the potential use of Air Quality Outcomes (AQOs) as a tool for regulating emissions of airborne pollutants from individual or groups of point sources such as those for which the Environment Agency has regulatory responsibility.

The report begins with a definition of the terms ‘effects’, ‘impacts’ and ‘outcomes’ as used specifically in this review:

Effects are defined as measurable change to a biological process, or an accumulation of a pollutant, that has no overall consequence for the organism concerned.

Impacts are defined as effects that are of sufficient magnitude to have a measurable consequence for the (long-term) functioning, vitality, productivity or survival of an organism, community or ecosystem.

Outcomes are defined as significant (i.e. “costly”, in the broadest definition of the term), measurable changes in the function or structure of an organism, community or ecosystem.

There follows a discussion of the features inherent in any outcomes-based approach. These include:

the need to show a causal link between observed effects and change in pollution concentration.

the need to assign a value to the resource affected (whether in terms of money, amenity or biodiversity) in relation to the cost of the proposed emission control.

the need to have access to appropriate knowledge – for example of expected baseline data – which will allow an outcome (as opposed to an effect or an impact) to be determined.

the need to take account of any confounding factors when designing sampling strategies.

the possibility of using biomonitoring before and after the introduction of control measures as a means of assessing outcomes.

Following a series of case studies which clearly illustrate these characteristics of the outcomes-based approach, the report discusses the relative strengths and weaknesses of the current (exposure-based) approach, the potential outcomes-based approach, and an intermediate alternative approach based on bioindicators. Factors considered include that:

current exposure-based approaches rely on generalised agreed links between the exposure to pollutants (air concentrations or deposition) and consequent adverse effects

while compliance in an exposure-based approach is assumed if exposure thresholds are not exceeded, possible outcomes of the exposure are not necessarily checked (or even known)

outcomes-based approaches require that the anticipated adverse effects of emissions must have a value that is high compared to the cost of emission controls

an outcomes-based approach requires that specific criteria for demonstrating compliance are agreed in advance, in terms of quantitative changes to the surrounding biota and their linkage back to source

an outcomes-based approach provides tangible evidence of the results of emission control which can be assigned a value

The report concludes with an evaluation of the relative usefulness of these different approaches in a regulatory environment, together with a series of practical recommendations as to how an outcomes-based approach might be used in assessing and regulating air pollutants.

There are no clear examples of “top-down” *effects*-based models – models that quantitatively link emissions of air pollutants to changes in environmental health - that could be used to support *outcomes*-based pollution regulation. The RIVPACS model is widely used for monitoring river systems, and is an example, albeit from a medium other than air, of how a “bottom-up” model can be used to establish baseline conditions for the status of a given ecosystem when unaffected by exposure to pollutants.

Recommendations include:

Carrying out of a pilot study for an outcomes-based regulatory regime for air quality.

Systematic recording of new findings on the relationship between pollution concentration and environmental effect, taking into account site-specific factors.

Development of a robust quantitative framework for determining the total value of a given ecosystem, using recent developments in the field of environmental economics.

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1. Introduction

1.1 Background

This report considers the use of Outcomes-based methods (defined below) as a tool for regulating emissions of pollution to the atmosphere. Given the Environment Agency's regulatory remit within England and Wales, the report is concerned primarily with the local impacts of individual point sources, or clusters of such sources, rather than with the impacts of emissions after long-range transport and transformations in the atmosphere.

Assessment and regulation of airborne pollution is currently based on the principle of maintaining the atmospheric concentrations of individual pollutants below set levels. These levels are fixed (at least over a defined administrative area), and are defined on the basis of the concentration above which an unacceptable level of harm might be expected to occur (or, in the case of "Air Quality Objectives", on what might feasibly be achieved by a given date). Limits on emissions from individual sources can then be set on the basis of their contribution to the atmospheric level of a pollutant, for example as a fraction of the "headroom" available between present ambient levels and the set maximum levels.

This approach has worked well in driving a reduction in overall pollution burden from historically high levels. It is straightforward to apply: ambient limits for a pollutant can be defined on an auditable basis, and the current generation of atmospheric dispersion models allows us to calculate the probable contribution to ambient concentration of given rates of emission from a source.

It has, however, some important limitations. It takes little account of sources of variability in *actual* impact that might arise from, for example, local topography, microclimate or combination with other physical or chemical stressors. It therefore lacks a close coupling to real ecological risk. As further reductions in pollutant emissions become increasingly difficult, and costly, to achieve, it is incumbent upon the regulator to demonstrate that real benefits will arise from further reductions.

In light of this, this review examines the potential of the more recently developed 'Air Quality Outcomes' (AQOs)[1] in both assessing and implementing environmental regulations concerned with controlling emissions of airborne pollutants

The interest in using AQOs as a regulatory measure lies in the fact that unlike other existing methods, AQOs are both proportionate and testable, and are based on actual measured changes to the biosphere in the vicinity of a regulated source – rather than simply on physical measurements of air quality. For such changes in the biosphere to be regarded as 'outcomes' they must be both significant (i.e. expected to affect the

[1] See definition in Section 1.2.

vitality or survival of an ecosystem or one or more of its components), and evidence-based (i.e. with a demonstrated causal link between exposure to a given pollutant and the resulting effect). An 'outcome' must also be quantifiable, in that the degree of damage (or recovery) to the biosphere must be assessed in relation to the **value** of the biota affected – whether in terms of cash value, loss of amenity, or loss of biodiversity.

This review focuses specifically on using biological measurements to assess the impacts of industrial activity on the surrounding environment due to industrial emissions of airborne pollutants. In particular, it is concerned with using measurements of existing flora and fauna to identify **temporal changes** caused by changes in patterns of emission – for example, following the implementation of emission controls. The use of AQOs for regulatory purposes requires techniques that demonstrate such temporal changes, although spatial patterns around point sources may also be assessed using measurements on biota. The practical application of biological measurements (biomonitoring) techniques that can be used to estimate the temporal and spatial scales of impact of industrial emissions will be considered in a separate report under this project.

1.2 Concepts and definitions

'Outcomes' can easily be confused with 'effects' and 'impacts', and in many instances these terms could be regarded as synonymous. However, in this context (of Air Quality Outcomes), we apply rather more specific meanings to these terms as follows[2]:

Effects are defined as measurable changes to a biological process, or an accumulation of a pollutant, that has no overall consequence for the organism concerned. The *effect* of a pollutant can be measured in relation to tissue concentrations in plants or animals, subtle changes in physiology or biochemistry, or by observing visible changes to an organism. Statistically significant *effects* may be observed that have no known long-term consequence for the behaviour, vitality, productivity, reproductive ability or fate of the organism. In many cases, an organism may be able to adapt to a pollutant stress with no obvious detriment to its health or survival. Any *effect* must be causally linked to pollutant exposure. This may be obvious from *in situ* measurements (such as the accumulation of a pollutant in tissue), or may be by analogy with controlled experiments that have demonstrated a causal link between the pollutant and the *effect*.

Impacts are defined as *effects* which are of sufficient magnitude to have a measurable consequence for the (long-term) functioning, vitality, productivity or survival of an organism, community or ecosystem. Such consequences may be observed as a change in the relative populations of plants in a community, or changes in the feeding behaviour of animals. However, inherent in this definition of *impact* is the possibility of reversibility on timescales similar to the initial change. *Impacts* may be viewed as changes in the structure of an ecosystem without large-scale changes in function.

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[2] These terms are denoted throughout this review using italics, to avoid confusion with the less specialised usage.

Outcomes are defined as significant (i.e. "costly", in the broadest definition of the term), measurable changes in the function or structure of an organism, community or ecosystem. In the case of a negative *outcome*, the term implies a degree of irreversibility, or the need for costly effort or long timescales to achieve reversibility. The term also implies some estimation of the value of the organism, community or ecosystem affected, which can be weighed against the cost of implementing, for example, a regulatory control on the emissions causing the outcome. The value may be expressed in terms of biodiversity, amenity or monetary value.

All three concepts are intrinsically linked, and can be placed in a sequence from an *effect* causing an *impact*, which leads to an eventual *outcome*. The concepts may best be described using a hypothetical example:

Fluoride emissions: potential outcomes for owl populations

Fluoride emissions from a smelter accumulate in the plant tissues of a grass. This can be seen as an *effect* (directly measurable, with a causal link to the emission source) if there is no obvious *impact* on the grass itself.

Voles feeding on the grass will assimilate the available fluoride in the plant tissues, and, if it reaches high enough concentrations, will develop dental lesions and malformation of the teeth. This can be seen as an *impact* on the vole as it affects the feeding ability of the mammal and the subsequent vitality of individuals. This has a negative bearing on the vole population and numbers of voles could decrease, but reach a new stable population density.

A decrease in the vole population will also have a negative effect on the local population of owls, which feed on the voles. Should the vole population decrease to an extent where the local owl population cannot be sustained, the subsequent loss of owls in the surrounding area can be seen as the *outcome* of the emission of hydrogen fluoride. There is a clear causal link to the emission source, even though there may be no evidence of fluoride accumulation in the declining owl population. Figure 1.1 illustrates how the transition from *effect* to *outcome* may be related to trophic level. In this example, owls are seen as having a greater intrinsic value than voles – if owls were very common in the region, then the *effect* on the owl population close to the smelter could be regarded as only an *impact*.

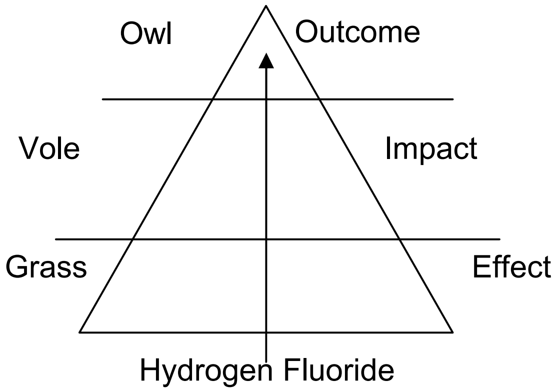

Figure 1.1: Illustration of the transition from ***effect*** to ***outcome*** as a function of trophic level in an ecosystem

From the example above we can see how the direct and indirect *effects* and *impacts* on different organisms in an ecosystem can lead to an *outcome*. If we look at the example more closely we could have reached a number of *outcomes* much lower down the pyramid structure. Hydrogen fluoride can be toxic to the plants themselves at high enough concentrations, and can lead to leaf senescence and sometimes plant death. This could be seen in itself as an *outcome*, particularly if the plant concerned were a rare species of high value, though less so if the plant were common elsewhere. Similarly, the voles could have reached a low enough population status that the integrity of the whole vole population in that location was at risk of collapse. This again could have been seen as an *outcome*. Therefore, many *effects* leading to different *impacts* within the ecosystem can lead to differing *outcomes*. These concepts are summarised in Table 1.1.

Table 1.1: Summary of terms

effect	measurable change in properties or state, causally linked to pollutant exposure
impact	measurable change in function or population structure, causally linked to effects
outcome	impact where the change has an assignable cost or benefit with respect to money, biodiversity etc.

The key issues for pollution regulators wishing to adopt AQOs lie in a) assigning some proportionate value on the *outcome* itself, b) being able to predict whether an *outcome* is likely to occur or not and c) confirming that the observed outcome can indeed be linked back to the pollution source. Once an adverse *outcome* is already occurring or has already occurred, it is often too late to implement a control regime, and an ecosystem may take many years to recover – or may not recover at all. It is therefore crucial to be able to measure the *effects* and *impacts* of changes in air quality (i.e. concentrations and deposition of airborne material emitted from an industrial site) that lead to an eventual *outcome*.

Likewise, if *outcomes* are to be related to particular pollutant sources (for example, in a complex industrialised area) then the measurable *effects* must be specific to the emissions from that source and no other. In general, the measurable *effects* will be manifest in terms of a disruption to ecosystem function which is attributable to the emissions. However, this may also include accumulation of material that is not currently known to cause harm, but which may, in time, result in exceedance of a threshold for toxicity. The progression from *effects* through *impacts* to *outcomes* could also be seen as a potential progression in time. If the accumulated *effects* over many years eventually lead to an *outcome*, then measurement of *effects* alone may act as a useful indicator of a future *outcome,* especially while there is still the possibility of arresting or reversing the progression.

One of the perceived benefits of an outcomes-based approach is that it can take account of any moderating factors that are known to exacerbate or ameliorate the responses of biota to air pollutants, including the combined effects of exposure to mixtures of pollutants rather than individual substances.

Inherent in the outcomes-based approach is the need to demonstrate the causal link between an *outcome* and the relevant emitted pollutants – something that may require considerable prior knowledge of the potential for a particular emission source to cause *effects* on surrounding biota. Although such prior knowledge in itself can give an indication of what to look for by way of an *effect* (and possibly *impact* and *outcome*), one of the additional benefits of the outcomes-based approach is that expected responses are compared against actual measured responses. Also inherent in the approach, therefore, is the idea of hypothesis testing. Any statistically significant differences (either qualitative or quantitative) from expectation lead to new insights which can be used to improve future use of the approach for regulatory purposes, and also to improve our understanding of the fundamental properties of the system being investigated.

The use of biomonitoring techniques is pivotal to outcomes-based approaches for assessing the *effects*, *impacts* and predicted *outcomes* of exposure to potential pollutants. 'Biomonitoring' is used in this review as a generic term to describe the use of measurements on biota to determine the influence (if any) of airborne emissions on them. Such measurements may range from simple measures of presence/absence to detailed biochemical or genetic changes, and they encompass both spatial measurements (geographical range) and temporal measurements (either 'real-time' or retrospective).

2. Goals and scope

The scientific literature contains many examples where biomonitoring is used to estimate the spatial extent of the influence of airborne emissions on the surrounding environment. The inferred link is generally made either by reference to the location of the source (for example ‘upwind’ vs. ‘downwind’), by reference to ‘control’ sites remote from the source, or by “before and after” comparisons which make reference to the onset of a change in source strength.

While details of biomonitoring techniques of potential value in an outcomes-based approach will be considered in a separate report under this project, **the focus of this first review is on the types of application of measurements and the philosophies that have been, or might be, employed to assess the *impacts* and *outcomes* of point sources of air pollution**. Where possible, examples have been chosen to highlight the potential for biomonitoring to assess changes in biota corresponding to changes in source characteristics – for example, following the introduction of emission controls.

This critical review is designed to use the literature – mostly published in peer-reviewed journals – to illustrate the application of biomonitoring to an outcomes-based approach to regulating emissions The review will refer to a number of case studies to outline the advantages and disadvantages of using an outcomes-based approach, and to provide a comparison with (existing) methods based on measured concentrations or deposition of pollutants. In particular, the issue of local-scale variability in confounding factors such as climate, soil type, and pollutant mixtures will be considered. This is one aspect that lends outcomes-based approaches considerable advantages over existing methods.

The case studies present a wide range of biomonitoring approaches, from straightforward *in situ* observations and measurements on vegetation around a distinct source of air pollutants, to regional studies of species presence or absence that permit quantitative estimates of ‘air quality’. Some of these case studies have been specifically designed to use *outcomes* as a measure of the success of regulatory controls on emission, and are therefore directly applicable to the development of an outcomes-based approach to air quality regulation. Others are taken from studies of aquatic pollution, where, despite the different medium, the principles involved in an outcomes-based approach can be illustrated.

In summary, this report presents a critical analysis of the potential benefits and disbenefits of an outcomes-based approach to regulation, as applied to such issues as a) the evolution of proportionate control measures, and b) the determination of compliance. The review then concludes with a series of recommendations on the feasibility of using an outcomes-based approach in regulating industrial emissions of air polluting substances.

3. Case studies

3.1 Prior knowledge and hypothesis development for outcomes-based biomonitoring

In any outcomes-based approach, the key to success lies in the confidence with which one can predict the condition of the environment surrounding an emission source in the **absence** of a pollutant stress. At the very least, existing knowledge should be sufficient to identify whether or not any observations made at the site would have been expected in the absence of the polluting source. Ideally, there should be an indication not only of the expected state, but of the direction and extent of any changes that might be expected *as a result of* pollutant exposure.

3.1.1 Establishing a baseline: the starting point for an outcomes-based assessment

Case study 1: Fluoride emissions near an aluminium works in Wales: predicting an expected *impact* on lichen communities

The first detailed study on the introduction of fluoride as an airborne pollutant into a previously unpolluted area, this work (by Perkins and Millar, 1987) provides a good example of where an *outcome,* predicted on the basis of previous knowledge, is clearly identified through subsequent investigation.

Given the established role of lichens as bioindicators, for example for sulphur dioxide, this case study focused on the lichen communities present around an aluminium works on Anglesey, North Wales. The site was surrounded by a mixture of agricultural land and small amenity woodlands (broadleaved deciduous, and evergreen and deciduous coniferous plantations), located mostly to the Northeast and within 1 km of the works. Despite the lack of earlier detailed studies on the introduction of fluoride (as a pollutant) into a previously unpolluted area, the author does cite circumstantial evidence indicating that lichens were sensitive to fluoride. On the basis of this prior knowledge, the author anticipated an impact on the lichens in response to the fluoride emissions. Amounts of airborne fluoride deposited on vegetation varied over the years of the trial, with annual averages of 4.0 g m^{-2} y^{-1} occurring at the most exposed sites downwind of the works between 1971 and 1976. Deposition reduced to 0.8 g m^{-2} y^{-1} after 1983.

Lichen growth was recorded at 48 quadrats over a large part of Anglesey by photographing changes over a 15 year period. The criteria for site selection were not described. Importantly, in 1970, quadrats were photographed around the site of the new works before emissions had started. In 1972 a further 39 quadrats were set up and photographs were taken annually until 1985 (except in 1974, 1976 and 1984).

Further to photographic methods, lichens were collected annually from hedgerows over a 2-3 week period between March and June for chemical analysis.

The lichen flora existing before the emissions commenced was typical, in both numbers and species abundance, of that found in oceanic western Britain. Corticolous (bark-dwelling) lichens occurred as epiphytes on hawthorn (*Crataegus monogyna*) and blackthorn (*Prunus spinosa*) hedgerows, and on broad-leaved deciduous trees, mainly sycamore (*Acer pseudoplatanus* L.), elm (*Ulmus spp*.) and ash (*Fraxinus excelsior L*.). The impact in this previously unpolluted area was rapid, with lichens quickly developing injury symptoms, including chlorosis, red coloration and necrosis. While levels of damage to lichens decreased with increasing distance from the works, large reductions, and even elimination, of lichens were observed immediately downwind of the site, leading to a 'lichen desert'. Moreover, damage to some lichens, such as fruticose lichens (those of "leafy" or "bearded" physical form), was generally much greater than to others. Indeed crustose lichens (those of an "encrusting" physical form) were little affected, and soon started to occupy the space formed by the elimination of others.

Analyses of fluoride content saw some species accumulate levels of more than 600 $\mu g\ g^{-1}$ after four years of exposure. Following the introduction of "improved operating conditions"[3], and a subsequent decrease in fluoride emissions, recovery of some lichen species was observed in less exposed locations and even at some of the more exposed sites. However, after initial recovery at the more exposed sites lichens redeveloped injury symptoms, leading to the conclusion that the pollution exposure could be episodic and may depend on wind direction, ambient fluoride concentrations and relative humidity.

Based, as it was, on prior knowledge of the possible effects of fluoride on lichens this case study was able to establish a predicted *outcome* on lichen populations in response to the emissions of the aluminium smelter. Surveying species composition before the emissions started provided an opportunity to reference a base-line population, which could then be monitored over time once the emissions had started. While *impacts* of the pollutant emissions were apparent, there did seem to be only rather slow recovery and recolonisation after a period of emission reduction. The author indicated that other factors, such as wind direction and exposure to the pollutant, were important in assessing impacts.

Case study 2: Biological Monitoring in the Forth Valley: establishing baselines for future studies

Case study 1 was able to demonstrate clearly the use of lichens as biological indicators of *impacts* and potential *outcomes* based on a long-term study relating air pollution *effects* to a particular source. However, such data are not always readily available for a

[3] It is not recorded whether these improved practices were introduced in response to the observed pollutant impacts.

given site and may have to be generated *de novo* before being used to investigate subsequent *outcomes*. In such cases, it is necessary to first establish the context in which any possible outcome-based change might be detected. This need to 'establish context', therefore forms an essential part of a successful outcomes-based monitoring and regulation approach, as illustrated in this next case study based on biological monitoring in the Forth Valley (Bell *et al.*, 1981).

In this case, a large-scale study was established in the UK in 1981, by the then UK Institute of Terrestrial Ecology, to develop a monitoring system using lichens and mosses to detect environmental change in the Forth Valley in east central Scotland (Bell, 1981). The area around the development of a large petrochemical complex at Moss Moran in Fife was of specific interest. The petrochemical plant was due to come on-stream in 1985.

The study area covered 7000 km^2 of east central Scotland and was designed to establish the effects of macro-climatic factors (temperature, light, rainfall) and land classification on the growth and performance of selected lichen and moss species. The primary survey in 1981 then selected the monitoring programme's sampling sites and established sampling protocols. The objectives of the primary survey were to:

Establish broad patterns of cryptogam (a term used to refer to the "lower plants" such as algae, liverworts, mosses and ferns) distribution on ash and sycamore trees and stone walls in the study area, including suspected clean and polluted sites as determined by sulphur dioxide (SO_2) measurements (Warren Spring Laboratory 1960-82);

Compile a list of potential indicator species;

Relate distribution of these indicator species to deposition of SO_2. Impacts of co-located pollutants were not considered, due to the scale of the survey and the limited understanding of multi-pollutant interactions at the time of the study.

The overall aim was to use these distributions to determine the impacts of SO_2. Permanent quadrats were established as a baseline to assess future observations, with photographic records to determine lichen growth. Two ash trees were selected in each km^2, and the 'permanent' quadrats (20 x 14 cm) located on them were photographed in the same position each time using permanent locating markers.

In this case, however, the initial baseline study concluded that rates of lichen growth were insufficient for such a method to provide unequivocal evidence of air pollutant *impacts* attributable to the proposed industrial development, and funding for further study was not continued. Although discontinued, this detailed primary study was critical, since without it any outcomes-based monitoring approach to the Moss Moran development at that time would have provided uninterpretable data. It should be noted that nowadays, improved atmospheric monitoring, identification of target species, reassessment of sampling sites selection, and use of statistical models would lead to significant improvement in the effectiveness of this type of survey.

3.1.2 Awareness of the effects of confounding factors

Case study 3: Sulphur emissions from a coal-burning power station in Pennsylvania: minimising confounding factors to assess *impacts* on vegetation

This study, by Rosenberg *et al.* (1979), investigated the effects of sulphur emissions from a coal-burning power station in Pennsylvania on a surrounding vegetation of mixed oak forest with white pine *Pinus strobus L.* and hemlock *Tsuga canadensis (L.)* Carr*.* Several parameters relating to species composition, and its variation with distance and direction from the emission source, were studied.

During the 25-year period of the power station's operation there had been various incidents where visible injury symptoms were found on the surrounding vegetation. Relevant to the distribution of such symptoms, the power plant was located on the banks of a river, some 30 to 60 m above the valley floor. The prevailing wind was to the Northeast and followed the down-river direction; early-morning low-level temperature inversions occasionally occurred, trapping pollutants near the ground.

A baseline monitoring plot was selected on the basis of the greatest severity of foliar damage, and additional plots were positioned up- and downwind of the plant for comparison. To reduce the effect of other, modifying factors each plot matched the topographical characteristics (elevation, slope and aspect) and stand characteristics (size, age and successional status) of the baseline plot. Species richness and diversity were measured in each plot using a system of subplots. The d.b.h. (diameter at breast height) of each tree was measured, including counts of total individuals and individuals in each species.

Species richness and diversity increased with increasing distance from the source and, in general, values were lower for downwind plots than for upwind plots at the same distance from the power plant. Ground vegetation and shrubs were more sensitive than overstorey trees, with ground vegetation showing the most damage. These types of plant may be more susceptible to damage than the trees, due to the microclimate, the additional stresses through competition, or, for annual vegetation, environmental stresses during initial establishment. Growth rates of understorey vegetation were not measured.

While the relationship between total basal area of overstorey trees and distance from the power station was poor, some relationships were observed within individual species. For example, in plots close to the power plant the basal areas of white oak (*Quercus alba* L) and red maple were larger, and those of sweet birch (*Betula lenta* L) and white pine were smaller than in more distant plots. It appeared that persistent tree species grew faster in plots closer to the power plant than in more distant plots because of reduced competition from the less competitive non-persistent species.

This case study illustrates an example of how sensible design can minimise confounding factors by using stratified sampling. It is often the case that ground flora are more sensitive to air pollution than trees themselves. This case study indicates that an *impact* (and even an *outcome*, in this case with respect to tree growth rate and competition) can be detected in the ground flora. Extrapolating from this, the ground flora is probably, therefore, the preferred plant community (ahead of trees) to monitor for *impacts*. (The increased sensitivity of ground flora over trees can be further shown by comparing the critical loads of nitrogen for trees (15-20 kg N ha^{-1} yr^{-1}) with those of ground flora (10-15 kg N ha^{-1} yr^{-1}) (Acherman and Bobbink, 2003)).

Case study 4: Biomonitoring PAH-induced changes in benthic (lake- or sea-bottom) community structure: the importance of prior knowledge of impacted systems

Looking beyond specifically air-related examples, this study by Oberdorster *et al.* (1999) was conducted in one of the Bayous of southeastern Louisiana, USA. Sediment and grass shrimp (*Palaeomonetes* spp*.*) were sampled at various distances from an oil refinery in the Bayou Trepagnier, which had received industrial effluents containing a wide range of thermal, PAH and heavy metal inputs over several years. Macrofauna and meiofauna (defined as benthic animals between 42 and 1000 µm in size) were sampled from the sediments. Grass shrimp – which are exposed to contaminants *via* the consumption of contaminated prey (ie. benthic invertebrates), through direct contact with sediment, and through uptake via the gills – were collected from the same sites and analysed for contaminant residues *per se* and for biochemical markers of pollutant exposure.

Analysis of the benthic community structure showed that there were eight taxa of macrofauna and seven of meiofauna in the sediments. The predominant macrofauna were nematodes, oligochaetes and midge larvae; the predominant meiofauna were nematodes, oligochaetes and rotifers, while hyrundinids, dipterans, copepods and foraminiferans were rarely found. While there was an overall low diversity and low abundance of macrofauna and meiofauna, this is apparently common for Bayous in south-eastern Louisiana. More importantly, results showed that sites with the highest contamination levels had significantly lower numbers of nematodes, oligochaetes, rotifers and total numbers of meiofaunal and macrofaunal animals. There were also fewer taxa compared to the less-contaminated sites. Grass shrimps collected from the most contaminated sites had elevated levels of enzyme activity associated with detoxification, indicating exposure to organic contaminants.

One observed *effect* was the increased levels of enzyme activity relating to detoxification in crustacean species sampled. *Impacts* were manifest in the reduced numbers of both species and individuals of benthic invertebrates. As these invertebrates form the basis of the detritus food chain, an expected *outcome* of this might be a decline in prey species available to the next trophic level – including the grass shrimp. Other possible *outcomes* might include the biomagnification of contaminants, where increased toxin accumulation in organisms at the next trophic level would in turn have implications for animals at the top of the food chain.

Since the low diversity and abundance of macrofauna and meiofauna was anticipated, the low abundance of benthic invertebrates at the less-contaminated sites was no surprise, and could not automatically be assumed to be a result of high levels of contamination. This study shows, therefore, the importance of prior knowledge of the impacted ecosystems. Indeed, in this case, it was necessary to make a comparison between relative biodiversity levels in contaminated *versus* uncontaminated sites, illustrating the vital importance of realistic control sites. Moreover, a specific biochemically-based test was used to increase the evidence for a causal link between pollutant presence and observed *impact*.

This case study specifically features a class of pollutants – PAH (polycyclic aromatic hydrocarbons) – which comprises a wide range of chemical species rather than a single compounds. Once confounders and constraints (such as inherent low abundance) were taken into account, the benefit of an Outcomes-based monitoring approach can be seen, in that organisms were integrating the combined impact of a range of pollutants.

3.1.3 Importance of field testing in developing an outcomes-based approach

Case study 5: Epinastic response of potato to atmospheric ethylene near polyethylene manufacturing plants: the relevance of laboratory results to field conditions

Tonneijck *et al*'s long-running monitoring project around a polyethylene factory in Belgium studied the direct effects of ethylene emissions in terms of levels of epinasty (asymmetric growth leading to a drooping appearance of leaves) observed in potatoes growing nearby (Tonneijck *et al.,* 1999).

While the degree of epinasty did increase when the prevailing wind brought ethylene emissions over the field, the response measured under field conditions was markedly less than had been obtained from controlled fumigations either under laboratory conditions or in greenhouses. Specifically, there was a much higher threshold concentration for *effects* to be observed. This reduction in sensitivity under field conditions was also observed in a separate study (Tonneijck *et al*., 2003), where similar effects were observed for two diagnostic bioindicator plants – petunia and marigold – that are particularly sensitive to ethylene.

These results suggest that threshold concentration levels determined in the laboratory under controlled conditions may be more restrictive than necessary to prevent damage in the field. It might be hypothesised that such differences are accounted for by, for example, the way apparent exposure in the field is estimated, or in physiological differences in the growth of laboratory and field crops. Investigations undertaken to explore such hypotheses will ultimately lead to an improved understanding of the environmental system. This will in turn improve our ability to predict *outcomes* in future cases, and hence to improve the quality and effectiveness of environmental regulations.

3.2 Outcome-based biomonitoring before and after a change in emissions

While biomonitoring studies have been used to evaluate environmental change following the introduction of emission controls, the kinds of measurements taken may not always provide a direct measure of the desired *outcome*, but only an indication of a change in *impact* which implies a favourable response to the emission control.

3.2.1 Inferred ***outcomes***

Case study 6: Control processes and environmental improvement: inferred *outcomes* following a change in pollution emissions

During the 1990s a series of studies were published by the Canadian Wildlife Service. They had monitored the *effects* of exposure to several contaminants – polychlorinated dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs) and biphenyls (PCBs) – on bird populations in the Strait of Georgia in British Columbia, Canada (Bellward *et al.,* 1990; Whitehead *et al.,* 1992; Sanderson *et al.,* 1994a; Sanderson *et al.,* 1994b). This location receives wastes containing these chemicals from its large number of kraft-process pulp and paper mills. Its bird populations include the great blue heron (*Ardea herodias*) and cormorants (*Phalacrocorax auritus*) which forage for small fish in the intertidal zone.

Several of these studies were reviewed here in an attempt to establish the effects of these contaminants upon ecosystems in relation to outcomes-based monitoring methodology. The studies documented conditions prior to the introduction of emission controls from the same area, located approximately 0.5 km from the discharge pipe of a kraft pulp mill.

The monitoring programme measured the concentration of contaminants present in the eggs of great blue herons and cormorants sampled in 1988, prior to the reduction of emissions. Concentrations of PCDDs and PCDFs in eggs were elevated compared to control sites, as were levels of detoxification enzymes present in the livers of chicks (hatched from artificially incubated eggs collected from the contaminated area). Measurements taken after the completion of the control processes showed that there was a rapid decline in PCDDs, PCDFs and PCBs present in great blue heron eggs, and that detoxification enzymes were no longer significantly elevated compared to control sites. Any change in diet, (for example, due to foraging for prey in an uncontaminated area) would be a possible confounding factor which would also account for a decline in contamination levels. However, the temporal correlation with the reduction in effluent does strongly imply a causal relationship between introduction of controls and reduction of *effects*.

Prior to effluent reductions, gross abnormalities were observed which included brain malformation (in cormorants) and oedema (great blue herons). The area's great blue heron colony had also demonstrated very poor reproductive success over several years. For example, in 1987 the colony saw no chicks fledged despite 57 active nests. Following effluent reduction there was a marked increase in reproductive success in the nests. Again the time sequence lends weight to the notion of a causal link between effluent reduction and reproductive success.

While no '*outcomes*' were determined *per se*, the evidence implies that the *outcome* in the absence of effluent reduction would be a decline in the population of great blue herons from this area in Canada. Following effluent reduction in 1988 a positive *outcome* was implied though not demonstrated unequivocally. Outcome-based regulation in this case might be based upon demonstrating an acceptable response in heron population to the levels of effluent emitted.

3.2.2 The need to consider the broader regional context during assessments with biomonitors

Case study 7: Lichen mapping in Germany : local and regional trends

In a major report on lichen mapping around the Altbach/Dizisau power station in Germany, Barholmess (2000) reported on a five-year environmental monitoring programme examining the effect of a new heat and power station on selected biological indicators. The study, which started in 1983 and continued until 1998, after the new power station was commissioned , used lichen mapping and standardised lichen exposure, augmented by air concentration measurements and soil analyses, to monitor the effects of constructing the power station.

Lichen mapping and standardised lichen exposure were considered the most suitable biomonitoring methods, given their known sensitivity to air pollution, and to sulphur dioxide in particular. Since lichen mapping is particularly important for long-term observations of air pollution (see Case Study 1) the results were compared with regional studies in Baden-Wurttemberg and in Hesse. Following the initiation of plant operation, the biomonitoring programme revealed no detectable change in the lichen population of the Altbach area, implying that the plant operated in an environmentally acceptable manner. Overall an *improvement* in air quality was observed from the start of the programme. Rather than reflecting any direct positive impact of the plant on air quality, the comparison of local with regional data suggested that the local data was in fact reflecting a countrywide trend towards improved air quality.

This study thus demonstrates the importance of setting a biomonitoring programme in the context of the broader-scale changes that are occurring in the surrounding region, which may not be specifically related to the source under investigation. Temporal changes alone (or lack thereof) may not be adequate to demonstrate a link to the presumed pollutant in the absence of the contextual study. Rather, any observed changes, or lack of changes, should be considered in the context of broader regional changes in air quality.

3.2.3 Identifying and controlling for confounding factors when designing sampling regimes

Case study 8: Remediating mining effluents: accounting for confounders when looking at effects of water quality on macroinvertebrate abundance

In their studies on the effects of mining effluents on water quality and macroinvertebrates in the Arkansas River, Colorado, Clements *et al.* (2002) and Clements (2004) demonstrated the benefit of developing well-formulated hypotheses which take into account possible confounders when exploring the effects of changes in polluting practices.

Biomonitoring of heavy metal concentrations and the benthic macroinvertebrate community in the Arkansas River was conducted over a ten-year period, from 1989 to 1999. Heavy metals were discharged into the river from mining operations. Sampling stations were located downstream from the two main sources of contaminants, known as LMDT and CG, the latter being downstream of the former. A number of the sites were upstream of CG – approximately 5 and 7 km from CG – while the downstream sites were approximately 1, 7 and 25 km from CG.

Water, sediment and periphyton[4] were sampled twice a year, in spring and autumn, for 10 years, and analysed for metal residues. Benthic macroinvertebrates were collected at the same sampling times. Since previous research had demonstrated that heptageniid mayflies were highly sensitive to heavy metals, species richness of mayflies and the abundance of heptageniid mayflies were selected as indicators of metal pollution. Remediation processes were implemented at LMDT in 1992 to improve the water quality of the river. This provided an ideal opportunity to determine the extent of recovery of the benthic community following remediation, and to compare benthic macroinvertebrate communities downstream from CG.

Predicted impacts of remediation included:

increased abundance of Heptageniidae and increased species richness of mayflies downstream from CG;

recovery of benthic communities downstream with increased distance from CG;

greater recovery of benthic communities upstream compared to downstream of CG.

The initial results demonstrated relatively high densities of metal-sensitive taxa upstream of CG compared to downstream communities which were dominated by metal-tolerant species (such as caddis flies and orthoclad chironomids). The mean abundance of heptageniid mayflies was significantly greater at the upstream sites,

[4] the matrix of algae, small crustacea and heterotrophic microbes (those relying for energy on an external source of complex, oxidisable organic compounds, rather than able to construct their own internally by, for example, photosynthesis) attached to submerged substrata

while these species were essentially eliminated from the site that was closest downstream to CG. However, abundance did recover at the most distant downstream site, approximately 25 km from CG.

Following remediation of LMDT there was a rapid increase in the number of Heptageniidae in the upstream sites (which had historically been affected by the metals released from LMDT) compared to a downstream site approximately 7 km from CG. There was also an increase in the number of mayfly taxa upstream compared to the downstream sites.

While this study aimed to assess the effects of the remediation of LMDT on benthic communities in the river, there are other, potentially confounding factors which can influence benthic community composition, including substrate composition, riparian vegetation and physicochemical characteristics. However, in this case, the design of the biomonitoring programme allowed certainty in the causal link between metal contamination and reduction in certain species. Particularly important in this respect was the repeated sampling before and after remediation, to minimise site-specific confounding factors. The same habitats were sampled throughout the duration of the study. The studies also included microcosm experiments, which, by providing control over conditions, provided support for the hypothesis that the alterations in benthic communities observed in Arkansas River resulted from heavy metal pollution.

Case study 9: Start-up of a municipal incinerator in Florida: Testing for confounding factors when using biological indicators

Rumbold and Mihalik (2002) report on the results of a biomonitoring programme initiated just prior to the start-up of a municipal solid-waste combustor in Florida, USA. Initiated in 1989, the programme collected annual measurements of chemical residues in eggs and chicks of water birds – specifically anhingas (*Anhingas anhinga*) and white ibis (*Eudocimus albus*) – occupying a rookery located approximately 0.8 km from the incinerator. These birds were chosen for two reasons: first as a surrogate species for the endangered snail kite (*Rostrhamus sociabilis*), which roosted at this colony; and second, because of their wide home range, which enlarged the area that could effectively be monitored for contaminants via their accumulation in foraging birds. The birds usually forage within 10 km of their colony.

Contaminants measured in the study were tetrachlorodibenzo-p-dioxin (TCDD), tetrachlorodibenzofuran (TCDF) and heavy metals, including arsenic, beryllium, cadmium, lead, nickel and mercury. Results from the first five years of monitoring showed that for the majority of contaminants, tissue concentrations remained at pre-operational levels. Eggs and nestlings (10-14 days of age) of both species were collected from nests. Where the contents of the eggs and carcasses of whole nestlings were analysed for contaminant residues, results showed that there was no statistically significant difference in contaminant levels from before the incinerator began to operate to after 10 years of operation. This was the case for all contaminants and for both bird species.

To investigate a possible confounding factor – that birds had changed their foraging behaviour in response to environmental change – the stomach contents of the nestlings were analysed to determine whether the birds were no longer foraging in the landfill, where the resulting ash from the incinerator was deposited. Results of these analyses showed that some of the ibises appeared to forage in the landfill and that this was consistent throughout the 10-year study period, indicating that the lack of concentration differences in tissue residues were not due to a change in food source.

In the absence of any clear confounding factor, therefore, it appears that the control processes put in place when the incinerator was built - a 76-m tall stack with two operational flues, dry flue-gas scrubbers and electrostatic precipitators - have meant that the incinerator has resulted in no subsequent effects, impacts or adverse outcomes.

3.2.4 Demonstrating recovery rates following emissions control

Case study 10: Reduced smelter emissions in Canada: non-linearity of consequent biological and chemical improvements

In addition to emitting heavy metals into the atmosphere, smelters also release large amounts of sulphur dioxide. Lake acidification can occur as a consequence, and this also impacts on heavy metal concentrations in the lake water so these two impacts are closely linked to each other in terms of *outcomes* and biomonitoring studies.

There have been a number of studies on the effects of closing smelters in the Sudbury region of Ontario, Canada during the early 1970s (Mallory *et al*., 1998; Nriagu *et al.,* 1998; Keller *et al.,* 2003). Control processes have also been implemented to reduce SO_2 emissions from smelters in this area. Several studies before and after major reductions in SO_2 emissions have measured the chemical and biological status of water bodies in the vicinity. They demonstrate that the decline in SO_2 emissions over this period resulted in improvements of water quality in the local area, which coincided with recovery in phytoplankton and zooplankton, zoobenthos and fish communities in lakes in the Sudbury basin. However, there was little improvement in toxic metal concentration in many lakes, leading to the conclusion that the unchanged metal concentrations were due to saturation of copper and nickel through mobilisation of these metals stored in soil and from surface run-off. Results did show, however, that there was a decrease in metal concentration in the top 3 cm of sediment, which was attributed to a reduction in metal inputs.

Thus, remedial processes to reduce SO_2 emissions from smelters improved not only the air quality, but also the water quality, in nearby lakes and other water bodies. However, the high metal concentrations in the soil in areas within 45 km of the smelter were still sufficient to impact the biota in these areas, despite the control measures applied.

This study illustrates the non-linearity of some environmental systems, in that *impacts* may be dependent on indirect *effects*, as well as the direct rate of contaminant input or removal. In such cases the linkage between pollutant concentration and effect on biota can be more complex than simple concentration measurements might suggest. Measurements of chemical changes to soils and waters may be necessary to identify the true extent of causal links between emissions and *outcomes*. In cases where indirect processes determine *outcomes*, then these too must be managed in order to allow effective recovery

Case study 11: Lichens as bioindicators in the Ruhr Basin: documenting changes in air quality and lichen health

Lichens have been used as bioindicators of air quality for about 40 years in the Ruhr area of Germany, after strict monitoring became necessary due to the region's concentration of heavy industry and high population density. This led to a number of studies documenting changes in air quality and lichen health since the 1960s, the most recent of which is that of Kricke and Feige (2004).

An earlier study in 1966 examined pollution in the central Ruhr area, where large sections of habitats studied contained no lichens at all (a 'lichen desert'). Other areas did contain some known pollutant-resistant species, including *Lecanora conizaeoides*, which occurred mostly in the presence of a layer of green algae. Small amounts of lichens were found at the southern edges of the study area, in an area representing the 'struggling zone'. Even outside areas of high population density, lichen growth was extremely low, while numbers of species were down to as little as four.

From 1989-1993, areas that had previously had no lichen coverage did recover to some extent, but large tracts of the Ruhr area still displayed very poor lichen growth. Affected species in many areas suffered stress on a scale described as "high to very high".

Transect research between 1998-1999 divided the Ruhr area into three sectors in a north-south direction. This established that air quality had improved in comparison to the findings of the 1989-1993 studies. Further studies from 1998-2002 confirmed that a recolonisation process was taking place, with a corresponding increase in lichen diversity up to 60 species. In addition, the study revealed that no areas were now suffering from very severe pollution.

Different study methods were used over the 40-year period considered. The studies were neither evenly-spaced in time, nor conducted as part of an integrated project. Consequently a direct comparison of the recovery with changes in emissions was not possible.

Although Kricke and Feige (2004) showed that lichen re-colonisation had occurred in the Ruhr area it was not possible, on the basis of the component studies, to establish a direct correlation between the lichen re-colonisation and individual pollutant sources. Their report did not give any data on the amount of emissions reduction across the study area, and provided no background air concentrations for comparison over the years of the study. It did imply a possible *outcome*, as a re-colonisation process has taken place over a period when a regional air pollution policy was being applied. This study has implications for assessing emissions abatements over a whole region rather than a single source, not least with respect to the requirement for long term consistency in monitoring methods.

A similar example of regional lichen recovery following regional improvements in air quality can be seen in a recent study in Sweden by Hultengren *et al* (2004). In that case, lichens were found to be slowly re-establishing, though there was a time lag between measured air quality improvements and the recovery of lichen communities. Although the study was not able to estimate how long it would take for lichen communities to recover, the authors cite Gilbert (1992) who estimated that the lag period in London was around 10 years. The possibility of such time lags should be borne in mind when attempting to interpret monitoring data and its relationship with changing air quality.

4. Outcomes *versus* concentrations as a regulatory tool

Environmental regulation exists to protect the environment from levels of loss or damage, arising from human activities, which are deemed to outweigh the benefits accruing from those activities. Controversy can arise because the costs (here used in the broadest sense rather than just necessarily financial) of damage and the benefits of the activity are dependent on point of view. A local community member whose livelihood does not depend on an activity might reasonably be expected to have a higher impression of cost, and a lower impression of benefit of, say, a factory's activities than an employee of that factory who may live some distance away. The regulator must identify a "tipping point" between cost and benefit that is acceptable from all viewpoints, and back this up with tangible evidence on benefits and disbenefits.

The idea of using direct measurements of the consequences of industrial emissions as a tool in regulating their impact on the environment has an immediate appeal for all parties involved. Such measurements can offer tangible evidence that emission controls are providing effective protection, that remedial action has produced an environmental benefit, or that environmental consequences continue to occur despite controls. By providing a quantitative link between control and impact they can also provide evidence as to whether expenditure on emission controls has yielded worthwhile environmental benefits.

An *Outcomes*–based approach to regulation offers the possibility of a regulatory policy based on locally-assessed environmental risk. Such policy would be more directly – and visibly – targeted at, and proportionate to, any specific threat than policy based on generalised pollutant reference values in the atmosphere. The regulator is thus seen to be providing an appropriate level of environmental protection without imposing undue burdens on the operators of potentially polluting activities.

In the following sections the outcomes-based approach to regulation is compared with the more conventional generalised exposure approach.

4.1 Exposure-based approach

4.1.1: The conventional exposure-based approach

Using a conventional exposure-based approach, an environmental impact assessment of an activity should identify, on the basis of current knowledge, those environmental components that are most susceptible to predicted changes in pollutant emissions arising from the activity. Assessment is based on expected emission rates and consequent air concentrations or deposition levels of the pollutant, based on local-scale dispersion modelling. Such assessments are necessarily made on the basis of a generalised understanding of pollutant effects on natural ecosystems, based on the concepts such as *critical levels* (for air concentrations) or *critical loads* (for deposition fluxes).

Critical levels and *loads* are not designed to prevent <u>all</u> damage to an ecosystem, but rather are designed to minimise risk to an agreed acceptable level. A *critical load* can be defined as "a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge" (CEH 2005). A *critical level* is defined similarly, but in terms of a concentration above which harmful effects (i.e. *impacts*) are known to occur. The *protection level* is defined at the level (e.g. *critical load* or *critical level*) at which the most sensitive (known) element of the ecosystem is expected to be protected.

Critical loads/levels are defined separately for different ecosystems, thereby introducing an element of specificity to any site evaluation. However, national maps of critical loads/levels rely on a statistical approach that combines land use-specific critical loads within a grid square, so that (typically) 95% of ecosystems within that grid square are 'protected' – i.e. have critical loads below the value set for the whole square. In practice the value for a grid square may be determined on the basis of considerably less than 95% of the grid-square's area, depending on the degree of land use heterogeneity in the grid square. Moreover it is possible for the key habitat in need of protection within a grid square to occupy less than 5% of the total area, its very scarcity being the reason for its designation as, for example, a Natura 2000 site which must be protected under European law. In such circumstances a site-specific critical level/load would offer a greater certainty of protection than a value chosen using a generic protocol.

When regulating emissions from a point source the surrounding land types and uses can generally be determined, on the basis of local knowledge and such sources as the land use classification periodically carried out by CEH. Critical load can therefore be estimated. Where available, some information on confounding factors is already incorporated into this (exposure-based) approach, but only at a fairly broad level. For example:

Critical loads for freshwater ecosystems may be calculated on the basis of measured, rather than assumed, chemistry in a water body. This gives greater site specificity than can be assumed solely on the basis of generic knowledge of the relationships between chemistry and critical load which have been derived regionally across the UK;

Critical loads for nitrogen deposition are habitat-specific. They take into account, for example, the fact that a low-nitrogen upland moor will be affected adversely by a lower input of atmospheric nitrogen compounds than a lowland pasture;

The critical level for sulphur dioxide is lowered for regions that experience severe winter conditions, since it is known that damage to vegetation by exposure to sulphur dioxide occurs at a lower concentration under such conditions.

4.1.2 Strengths and limitations of the exposure-based approach

The major advantage of the conventional regulatory approach, based as it is on measured or modelled pollution levels from which exposure is inferred, is that such levels can be assessed by physico-chemical monitoring – either of stack emissions, or the surrounding air – coupled with models of atmospheric dispersion and transport. Calculated or measured air concentrations and/or deposition fluxes can be compared with some pre-defined level that has been judged as protecting sensitive parts of the environment to an acceptable degree under most circumstances. Criteria for demonstrating regulatory compliance can be derived on the basis of a combination of actual data and modelled results, usually reported as time-averages, over the spatial domain of interest. The underlying processes that control transport of air pollutants are sufficiently well understood that predictions of concentration and deposition can generally be made to a standard acceptable to regulators. If necessary, predictions (for example from dispersion models) can be tested using additional physico-chemical measurements.

A critical issue with the exposure-based approach lies in the fact that, although regulatory compliance can be assessed on the basis of physico-chemical measurements with relative ease, there is the possibility that even full compliance may not be sufficient to ensure an acceptable level of environmental protection. For example, there are site-specific factors – including local climate, soil chemistry, presence of other pollutants and land use history – that render ecosystems more (or less) sensitive than predicted to a given pollutant exposure. There is rarely any attempt to discover whether or not the controls employed are actually protecting the surrounding environment (as is the intention of an outcomes-based approach). Compliance with the physico-chemical criteria is assumed to demonstrate that the environment has been adequately protected. The regulatory authority cannot, therefore, demonstrate *definitively* that the environment has been adequately protected. Rather, it relies upon reference to the criteria originally used in setting the prescribed levels of emission and/or critical levels/loads. Neither can it demonstrate to the industry under regulation that imposed levels of control have been necessary, sufficient, or proportionate to protect the environment to the required level. The industry, meanwhile, cannot estimate the cost-effectiveness of the control procedures other than by reference to the original criteria, which may include margins of safety that are too high or too low, or which may not be appropriate to a specific site.

In practice, identification of the most sensitive component of an ecosystem is generally hampered by a lack of information on dose-effect responses to individual pollutants by individual species or communities. Generalisations made when setting critical levels are usually based on only a limited number of experimental observations. These may have been made under very different climates or soil chemistries and may, therefore, not be appropriate for a given site or circumstance. For example, neither the influences of seasonal/phenological interactions on pollutant effects, nor the influence of episodic (as opposed to chronic) exposures are well understood. As a consequence of such deficiencies in our knowledge of pollutant interactions with climatic and other abiotic or biotic stresses, emission controls may be made either more stringent or more lax than necessary to achieve the desired level of protection. (An example of how laboratory and field levels of impact may differ was given in Case Study 5). Nor are we yet fully-informed as to the combined effect of *mixtures* of pollutants, though there is evidence that the impact of some pollutant combinations may be greater than the sum of the impacts of their individual components.

In the field of ecotoxicology, a protection level-setting process is applied that is similar to that for setting critical levels/loads. In this field a *predicted no-effect concentration* (PNEC) is calculated on the basis of dose-effect relationships derived under experimental conditions. The PNEC can be defined for multiple species systems as "the concentration below which a specified percentage of species in an ecosystem are expected to be protected". When extrapolating from laboratory to field this approach is prone to the difficulties and uncertainties described above. The process of defining PNEC includes an assessment of the level of certainty to which the dose-effect relationships are known. Such an assessment is crucial to a truly risk-based assessment of environmental impact, and uncertainty analysis is increasingly featuring in such process as critical load setting (for example Wadsworth and Hall 2005). The process of defining appropriate uncertainty limits for dose-effect relationships is, however, restricted by lack of data.

Continuous monitoring of emissions and/or the surrounding air is often (though not always) associated with the need for expensive capital equipment, together with staff with the skills both to operate the equipment and to interpret the resulting data. In principle, biological monitors of air pollutant exposure can be used as a proxy for instrumental physico-chemical measurements (see Section 3.4.3), particularly in cases where pollutants accumulate in tissue. As well as sometimes representing a cost saving, this approach offers the advantage of bringing exposure assessment a step closer to the actual experience of biota in the field, thereby incorporating some of the potential confounding factors – particularly if the biomonitors are growing *in situ*. There may be scope for using standardised plants as bioindicators, eliminating the component of measurement uncertainty arising from the different susceptibilities of the various cultivars or varieties of a given species. However, compliance criteria must still be translated into an effective biological concentration, or *vice versa*. Like instrumental monitoring, the use of bioindicators or biomonitors requires skill and knowledge. Measurements *may* be taken less frequently than for instrumental air monitoring, especially if the biological measurements record the accumulation of material over a period of time, but may be more labour-intensive. Note also that biomonitoring in this context may simply involve recording *effects*: the bioindicator shows whether a biological response can be observed, but not whether this is biologically relevant to the ecosystem of concern. Such measures of actual exposure do, however, reduce the uncertainty associated with translating air/soil/water concentrations of a pollutant into exposure experienced by the biota. They might therefore be regarded as a step closer to reality than purely instrumental physico-chemical measurements of pollutant concentrations.

4.2 Outcomes-based approach

Using an *outcomes*-based approach in environmental regulation would require some form of preliminary evaluation of sensitivities and risks, similar to that used in current environmental impact assessments. During this process the elements of the surrounding environment likely to be at risk from the emitted pollutant(s) would be identified, their value (see Section 1.2) established and acceptable levels of *impact* determined. Should there be no existing information on the response of key components of the affected ecosystem to specific pollutants, some selection of indicator species amenable to diagnostic measurements would be necessary. At this stage, any knowledge of confounding factors can also be introduced into the risk assessment.

On the face of it, this sounds little different from the process of regulation on the basis of exposure. A fundamental difference does exist, however. Though a link between *impacts* and emissions is still required (in order that emissions can be controlled in such a way as to avoid unacceptable *outcomes*), regulatory thresholds under an *outcomes* regime would be expressed in terms of an acceptable level of *impact* rather than of pollutant concentration or of flux.

The main advantage of this approach, particularly with respect to new pollution sources, is that where new emissions result in the detection of no unacceptable changes in biota (either over time or by comparison with nearby 'control' sites) this can be regarded as demonstrable evidence of compliance. If the *outcome* under a given set of operating conditions differs from that expected for the exposure level arising from those conditions, then changes must be made to the operating conditions in order to achieve compliance. If an unexpected level of harm is seen then, irrespective of ground level pollutant concentrations achieved, action must be taken to redress the harm. Conversely, if the actual level of harm falls below expected (and acceptable) levels, then the operator has scope to increase emissions provided, of course, that this does not lead to exceedance of any other limits or objectives. Similarly, where the objective is remediation, then the rate of recovery can be assessed in terms of the target species or ecosystem.

A further advantage is the likelihood that measurements need not be continuous. They might be possible as infrequently as annually, perhaps timed to correspond with time(s) of year when any effects of the pollutant would be most clearly manifested. In the long run, given sufficient information about likely susceptible species and the pollutant stress, it might even be possible to tailor an emissions strategy more closely to the life cycle of the ecosystem, for example permitting greater emission rates at times of year when susceptibility is low and *vice versa*. This would be analogous to procedures currently in place to restrict emissions from certain sources when meteorological conditions are such as would result in high pollution levels.

Set against these advantages is the requirement for biological, chemical and statistical expertise in recording, sampling and/or laboratory analysis of samples. There is also a need for an understanding of the range of natural variation in biological responses in order that this can be incorporated into the design of any sampling scheme employed. Detailed sampling and analysis of a range of sites and materials is time-consuming and can be expensive. Given the number of measurements likely to be required to provide statistically valid data, together with the fact that pollutants may affect several distinct ecosystem types in the vicinity of the source, this could make the outcomes approach costly to operate. Such costs should, however, be offset against the value of damage prevented, recovery made or better regulation implemented.

With respect to statistical validity there are two issues that must be considered:

Are measurable changes statistically significant (i.e. different from 'control')?

Are statistically significant changes biologically or ecologically significant for the species, community or ecosystem being measured (i.e. greater than the threshold of change required to produce an *outcome*)?

Both of these are potential areas of contention, and can only be addressed through rigorous statistically-driven study design. Poor study design can lead to an apparent lack of statistical significance in measured differences in time or space, even where biologically significant changes have occurred. The design of the monitoring regime should therefore be developed from the first with statistical sensitivity in mind.

The specific requirements for adopting an outcomes-based approach are laid out in greater detail below (Section 5). Ideally, measurements should be made both temporally and spatially. By focusing on designated 'target' areas (likely to be exposed to emissions) repeated measurements of the same component before and (at pre-defined intervals) after the proposed changes in emissions can thus be used to assess whether a measurable change has occurred. Because an outcomes-based approach requires the development of changes in the affected organism, community or ecosystem, the time-scales for repeated measurements are likely to be of the order of months to years. Spatial measurements, comparing similar components in areas subject to the emissions to those in areas not exposed, would be required to demonstrate that any changes observed in the target areas were not the result of more widely operating causes (eg. climatic) that were unrelated to the emissions.

Even if adverse changes were to be observed, it might be difficult to ascribe them to specific pollutant emissions in a manner that could be used legally to demonstrate non-compliance. This is an important practical issue: can changes in outcome due to specific pollutants from specific sources be distinguished from background effects, or even non-pollution effects such as climate change? In general, physico-chemical measurements are probably easier to attribute to specific sources than measured outcomes. The detailed criteria for non-compliance would, therefore, need to be agreed as binding beforehand. In addition, given that *outcomes* are defined in terms of the value ascribed to them, some measure of the economic, amenity or biodiversity value of the *outcome* must be assessed compared to the cost of the emission control measure, in order that proportionate control measures can be put in place. This necessarily involves quantitative evaluation of an *outcome*, rather than simply qualitative evaluation such as the presence or absence of a key species.

4.3 Summary

Conventional assessment methods based purely on physico-chemical measurements of pollutant concentration or flux assume agreement between industry and the regulator that such measurements represent a realistic evaluation of the risk to the environment – whether or not damage is actually caused. The test for compliance is then greatly simplified, in that it concerns the interpretation of (quality assured) physico-chemical measurements and any spatial interpolation or extrapolation using physically-based models. Attribution is also relatively simple.

Outcomes-based methods are based on whether or not the target organism or ecosystem is actually affected, regardless of the level of emissions that occurred. This offers the possibility of better regulation, that is targetted at the most harmful emitters and proportionate to actual risk or harm. However, agreement needs to be reached on the direction and magnitude of any effect that is regarded as biologically significant, on the statistical design that will deliver the measurements with the required precision and accuracy, and on factors inferring confidence of attribution to the regulated source. The responses of biota to pollution stresses can be similar or identical to those brought about by other stressors such as climate change or changing land management. Effective outcome measurements are dependent upon having the means to take such confounders into account. Outcomes-based monitoring therefore has higher burdens with respect to effective network design, statistical planning and understanding of the environmental system under scrutiny than exposure monitoring. The latter does not escape these burdens, however. Rather, they are in effect deferred to separate studies of dose-response relationship from which acceptable proxy levels are defined.

A comparison of the two approaches is given in Table 4.1 and illustrated in Figure 4.1. The latter illustrates how uncertainty in the level of protection achieved is reduced with a move towards *outcomes*-based regulation, while uncertainty in the control measures required to achieve compliance increases. The burden of uncertainty – and the need to resolve that uncertainty – thus transfers to the polluter in an *outcomes*-based regime. (Under an exposure-based regime the burden of uncertainty lies more with the body responsible for settling protective standards on the basis of interpretation of physico-chemical levels in terms of receptor impacts).

A crucial question, then, in weighing the benefits of outcome and exposure measurements in a given situation is whether residual uncertainties arising from confounders which decouple outcome from pollution pressure are more or less than those uncertainties which arise from applying dose-response responses, often gained under laboratory or idealised field conditions, to real ecosystems.

An ideal situation would be one in which the “real world” outcomes measured were ones which were free of confounders, and which could therefore be linked unequivocally to pollution pressure. It is essential, therefore, to keep a watching brief on developments in biomonitoring in the hope of finding techniques which can bring us closer to that ideal.

Figure 4.1 Schematic representation of both an exposure-based and an outcomes-based approach to environmental regulation

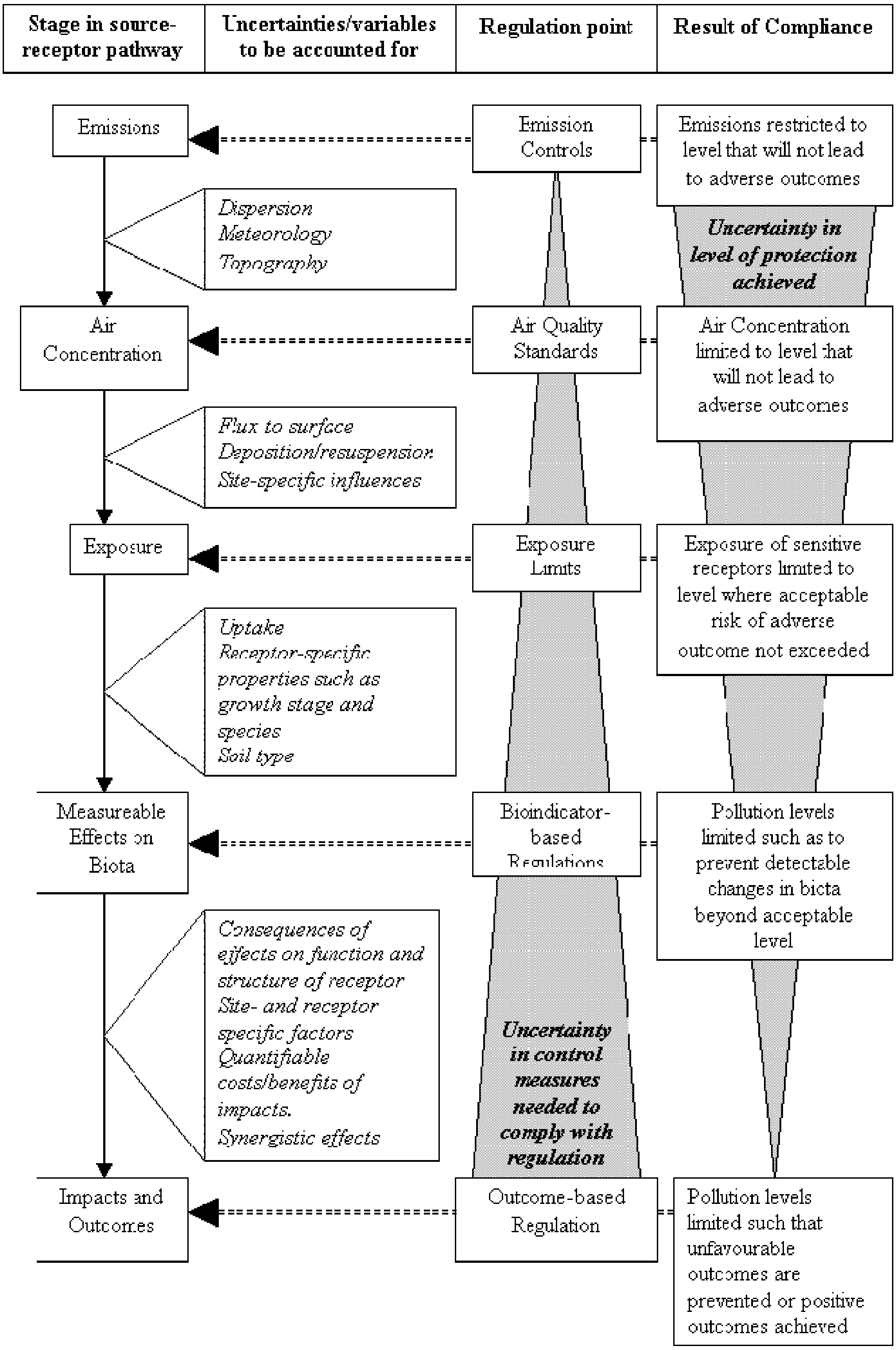

Table 4.1 Comparison of conventional and outcomes-based approaches

Conventional (exposure-based)	Outcomes-based
Objective monitoring of air concentrations or flux rates. Compliance with the regulations can be relatively easily demonstrated. Link between source and concentration/flux relatively easy to make. Criteria for compliance are generic	Measurements of response. Absence of unacceptable outcome demonstrates compliance. More difficult to make unequivocal link between outcome and presumed causes/sources, Apparent non-compliance can be contested. Criteria for compliance have to be agreed on a site-specific basis.
May be expensive to implement, Requires continuous operation to detect where thresholds are exceeded. Requires knowledge of the operation, and of the strengths and weaknesses of the instruments in order to produce representative data Requires rigorous sampling strategy design	May be expensive to implement Emphasises long-term changes, so measurements are episodic, perhaps annual. Requires biological skills in application and interpretation. Requires rigorous sampling strategy design
Criteria for limits based on the generalised risk to surrounding biota. Basis for this generally laboratory, and occasionally field, measurements of dose-response to a single pollutant by one or more ecosystem component, probably in an idealised situation (eg. well watered, without other abiotic stress, without competition, without biotic stress from pests or diseases).	Criteria for limits designed for each specific application, based on existing flora and fauna, soil conditions (including underlying geology), climate, and the chemical and physical nature of the cocktail of pollutants emitted.
Includes a margin of safety that may be very large to take account of unknown environmental interactions with the pollutant and unknown responses of ecosystems of interest	May be more or less stringent overall, or may identify conditions under which controls can be varied, based on known deposition/uptake pathways and the biota most at risk from the particular pollutant(s) emitted. Chosen level will be auditable.

Takes no account of synergistic effects with co-occurring pollutants (from same source, or from others nearby).	Takes into account the impact of the environment as a whole, including unforeseen factors and interactions, in setting ‘baseline’ conditions.
Takes no account of unforeseen modifying factors on biological response.	Has to consider the impact of the environment as a whole in setting ‘baseline’ conditions.
Takes no account of climatic variability, or any seasonal differences in susceptibility except via ‘worst case’ scenarios.	Should explicitly consider such interactions by use of one or more ‘control’ sites.
Relatively easy to attribute culpability to individual sources based on inverse pollutant pathway modelling from physico-chemical receptor to source	More difficult to attribute culpability to individual sources. Risk that outcomes due to other factors (e.g. climate change) may be wrongly ascribed to local pollutant sources
Relatively immediate indication that a change in pollution level has taken place so that action (e.g. emission reductions) can be taken quickly and harm minimised. (Note that such immediate action presumes a good level of confidence in the connection between pollutant level and consequent outcome).	Potential time-lag between pollutant emissions/inputs and appearance of *Outcome* so that harm must take place before corrective action is seen to be needed.
Does not require baseline information on specific receptor status, relying instead on generic information relating pollutant concentrations to receptors in general	Requires baseline information on receptor status prior to a release of pollutants, in order to distinguish receptor changes arising from pollutant changes
Takes no account of rate of accumulation, or pattern of exposure (e.g. allowing for recovery between intermittent exposures)	Effects of pattern of exposure inherently taken into account to outcome based, but may not identify whether episodic or chronic exposures are responsible for any observed effects.

5 Practical suggestions for an outcomes-based approach

Previous sections have considered the philosophical basis of *outcomes*-based regulation, explored its feasibility through the use of case studies and considered the factors other than pollution concentration which may have some bearing on the *outcome* resulting from a given level of pollutant emission. In this section we examine how *outcomes*-based regulation might be developed in practice.

Figure 5.1 shows a decision flow diagram outlining the basic steps a regulator could take when assessing the need for or benefit of taking an outcomes-based approach to pollution regulation:

Decision 1 – asks if the assessment area contains species or habitats that have been identified as having biodiversity/conservation, economic or amenity value.. This begins to identify possible *outcomes*. To estimate the biodiversity value of an ecosystem surrounding an industrial plant some form of biological survey is required. An Environmental Impact Assessment report could be consulted if this has been carried out in the past. Other information on protected sites could be available from the local conservation officer at the conservation agency Natural England. Information on economic value could be obtained from timber estimates if trees are present, while local councils and conservation NGOs could provide information on the amenity value of an ecosystem;

Decision 2 – asks if the ecosystem is sensitive to the emitted pollutants. Information helping in this decision could include, for example, prior knowledge of ecosystem-pollutant impacts or could be derived from online-tools such as the Air Pollution Information System (APIS)[5].

Decision 3 – takes into account any factors which might mitigate or exacerbate expected pollution-induced *effects*. Such factors could include, for example, presence of other pollutants, soil characteristics, climatic stresses, topography and land management practices such as grazing. Information on these could be acquired from soil maps, models and, if available, land management action plans.

Decision 4 – examines if there are any likely predicted *outcomes* for the ecosystem. It also asks if these *outcomes* are proportionate in respect to the estimated value of the ecosystem. Information acquired at decision stages 1 and 2 can be used in supporting this decision.

Decision 5 – asks how the *outcome* relates to the desired present or future level of emission control.

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[5] See: www.apis.ac.uk

Decision 6 - asks the question, "is the level of outcome observed acceptable or is there a requirement/scope for further changes to the existing emissions regime?" If the latter, then the decision process returns to earlier decision stages for a further iteration.

Where any potentially sensitive species (target organisms) are identified, their occurrence and condition should be recorded in the (presumed) affected area, to provide a baseline for subsequent studies. A sampling scheme with detailed protocols for measurements, statistical design and statistical testing would need to be drawn up, with appropriate quality control and quality assurance measures in place. (Note: in the case of species of high conservation value, it may be inappropriate to use destructive sampling methods, and appropriate surrogates within the environment may need to be identified (see, for example, Case Study 9 above)). For studies intended to test for a reduction in adverse *outcomes* following emission controls, comparison should ideally also be made against an area likely to be unaffected by emissions, which will therefore act as a control indicating the condition expected in the absence of the pollution emitter.

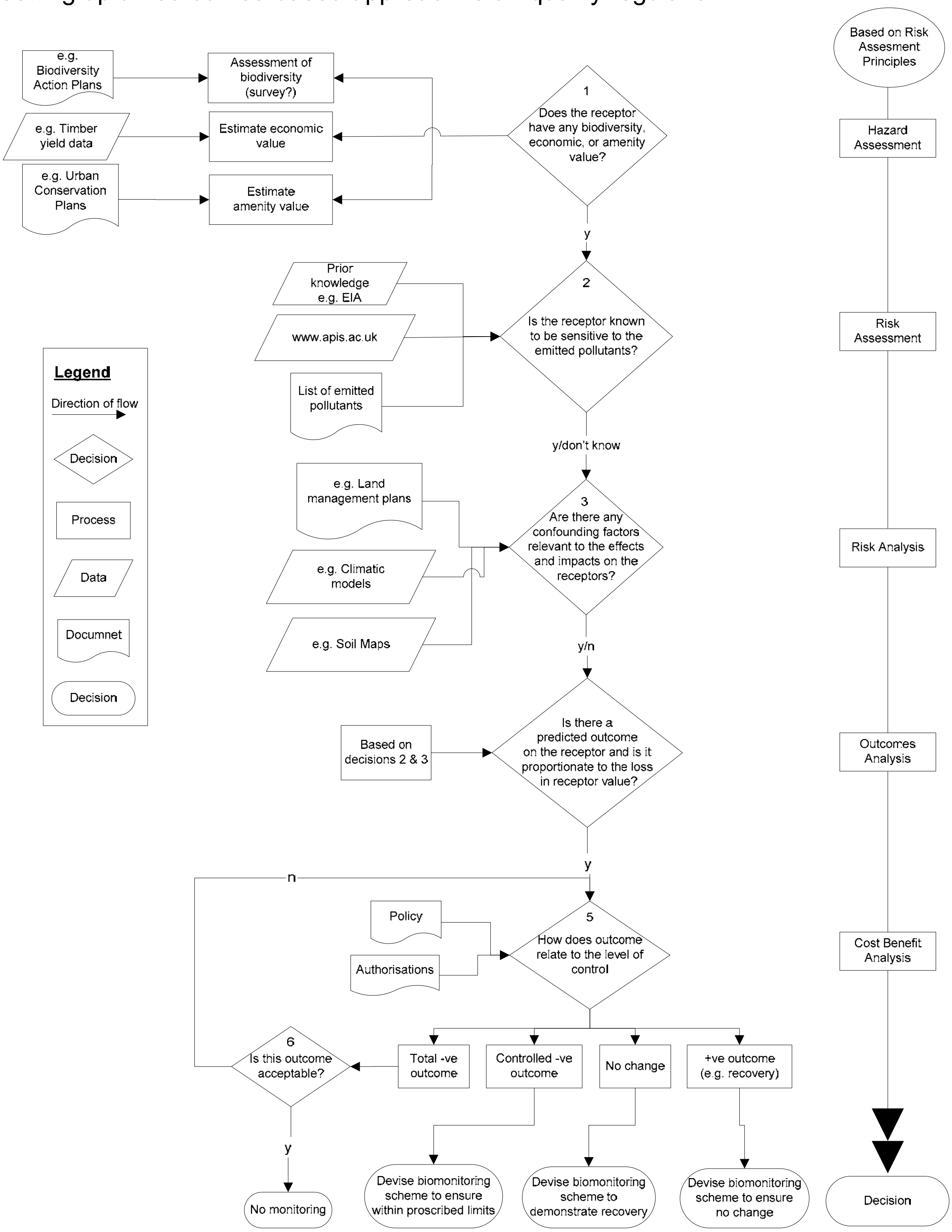
The types and levels of change which could be regarded as significant, in both biological and statistical terms, should be agreed by all stakeholders at an early stage. For example, although statistical significance is usually assessed using a Type I error (apparent effect when there is none) rate of 5%, it may be more important in this context to consider Type II errors (no apparent effect when there is one), or with a different level of confidence. (Note that the 5% above refers to level of uncertainty rather than the magnitude of any change that might be observed.) A statistically significant effect may or may not be deemed relevant for the future health of the organism or ecosystem.

The baseline study can then be used to estimate the spatial variability across the study area, including any areas designated as 'controls'. This will allow better estimates to be made of what is required for a sampling strategy to deliver the required precision and statistical power (including power to attribute source culpability). At this stage some refinement, redesign or revision of agreed 'significant effects' might still be required.

Subsequent measurements, following the introduction of the new emissions regime, are then made and evaluated. If no changes are detectable, then there is scope for renegotiating the emission guidelines to permit different amounts, or patterns, of emission. If significant changes are within acceptable limits for outcome then the regulatory regime is performing appropriately. If unacceptable *outcomes* are observed, the magnitude and types of the effects will then be used to determine what reduction in emissions, or change in the pattern of emissions, is necessary.

Depending on the chosen biomonitoring scheme, compliance may be tested weekly, monthly, annually, or over longer periods, to take account of temporal variations in ecosystem responses and the time-scale over which damage might occur. In cases where controls have been imposed in expectation of the recovery of species or habitats, the basic principles are similar, except that Type I errors are now probably more relevant in determining whether introduced changes in emissions have produced the desired change for the better.

Figure 5.1 : Flow diagram showing decision tree for assessing the requirements for setting up an outcomes-based approach to air quality regulation



6 Feasibility of effects-based models

6.1 Introduction

The *outcome*-based approach to pollution control aims to provide a closer link between pollutant and consequence than the more conventional concentration/flux based approach.
While the latter is currently aided by many well-tested numerical models that simulate the transport and chemistry of pollutants in the atmosphere, there are few models that attempt to incorporate biological or ecological factors so as to predict the impacts, effects and outcomes of pollutants on the biosphere. More usually, chemical and physical measurements or predictions are used to predict the subsequent biological status of an ecosystem in a way similar to that used in physico-chemical measurement-based regulatory regimes, as described above, using generic, often laboratory-derived data to infer pollution effects.

In principle, an "effects-based" model – defined here as a model that goes beyond prediction of concentrations and fluxes, linking emissions through to actual consequences for the biosphere - could be constructed from a knowledge of the interactions of biota, abiotic factors and air pollution. Such a model could then be applied to a particular site, taking into account its climate, aspect, soil geology, hydrology *et cetera* to build an expectation of the range, status and diversity of biota in the presence, or absence, of air pollutant stress. This expectation could then be used to generate the specific hypotheses against which measurements made at the site would be assessed. In this way, significant deviation from the expected status of the site, if causally linked to the air pollutant(s) concerned, might then be used as evidence for an air quality *outcome*. This philosophy underpins the outcomes-based approach to regulation, which would be supported by effects-based models in much the same way that current concentration/flux-based regulation is supported by current transport/chemistry models. Effects-based models have, however, not yet been formulated for practical use.

6.2 Example of an effects-based model

The closest example of an effects-based model is provided by the computer-based RIVPACS (River InVertebrate Prediction And Classification System)[6] model of freshwater quality (Wright *et al.*, 1993). Although a close approximation, however, it can not currently be regarded as a fully effects-based model because of the lack of information on cause and effects of measured deviation from expectation.

[6] RIVPACS was developed as a biological technique for the assessment and management of river systems (See: http://dorset.ceh.ac.uk/River_Ecology/River_Communities/Rivpacs_2003/rivpacs_introduction.htm).

Two distinct components of the RIVPACS system were developed; the first to generate site-specific predictions of the macroinvertebrate fauna to be expected in the absence of environmental stress, and the second a process for locating sites of high biological quality. RIVPACS was built on a classification system of unpolluted river systems (or in systems subject to only very minor anthropogenic alterations), based on multivariate assessment of macroinvertebrate fauna across the whole of the UK. The idea behind the classification was to provide the ability to predict the type of macroinvertebrate community expected, at a site-specific level. This could then be compared with the actual biological quality giving an overall indication of the status of a river system's water quality.

The RIVPACS system was developed from biological and environmental data obtained across the UK. It incorporated varying environmental characteristics such as altitude, slope, hydrochemistry and hydromorphology, as well as biotic features, and it was hoped that it would provide a very useful tool for assessing river quality at 'any' location within the UK. Where sites of a 'similar' physical type are absent, RIVPACS cannot define a reference condition for that site. However, by selecting the best available site targets RIVPACS can be used to improve other sites to this level.

The RIVPACS approach has now been adopted across other European countries including Sweden – for streams (SWEPACSRI) and lakes (SWEPACLLI), and in the Czech Republic (PERLA). World-wide this has also included RIVPAC versions in Australia (AUSRIVAS), Canada (BEAST), and New Zealand, where the RIVPACS approach has been applied to fish and macrocrustaceans. In the USA a multi-metric approach is used, measuring an array of indices or metrics to provide an overall indication of site status. However, this approach is not always optimal as the multi-metric results are strongly influenced by natural physical gradients which make it difficult to assign a target reference. The use of effects-based models in the context of an outcomes-based approach must explicitly include the role of known confounding factors.

Although RIVPACS provides a well referenced and wide-ranging model for assessing the predicted water quality of a river system, it cannot isolate water quality outcomes to specific pollutant sources (unless of course sampling were undertaken above and below a suspected source – see case study 8 above). Another downside of RIVPACS, is that it requires a large quantity of prior knowledge (data) that defines what is 'normal' for given site characteristics. Under the Water Framework Directive there is now a Europe-wide requirement to ensure that relationships between the biological state and physical and chemical properties of surface waters are sufficiently well understood to enable the management of catchments and rivers to achieve their ecological objectives. To date, however, there are few biomonitoring protocols or methods for freshwater systems for use in pollution prevention.

6.3 Development of effects-based models

Bottom-up approach

The approach taken in the design of RIVPACS could be applied to the development of similar comparative systems for terrestrial monitoring of ecosystem health, for example on the basis of lichens or other epiphytes. For many decades lichens have been used to assess the quality of air, and a number of methods have been used to indicate nutrient nitrogen pollution, sulphur dioxide and acidifying pollutants. However, given the history of industrial development in the UK, and the long times required for lichen species composition to recover once the pollution stress has been removed (see eg. Bates *et al.,* 2001), it may prove difficult to ascribe observed differences between actual and predicted lichen distributions to current, or recent air pollution. Moreover, designing and implementing a lichen index of this nature across the UK would be no trivial exercise. This type of approach has been attempted in Italy, using climatic and distribution data for lichens to produce a scale of 'Deviation from Naturality' (Loppi *et al.*, 2000) in a way similar to RIVPACS. While RIVPACS is solely used to assess river quality, and lichen distributions are relatively insensitive to the substrate on which they grow, the scope for assessing the full range of UK terrestrial habitats on the basis of overall ecosystem status is much larger. The use of 'Ellenberg indices' of species composition might also be regarded as the basis of an effects-based model. Changes and geographic patterns in plant community composition can be related to air pollution, and useful indicators have been identified, especially mean Ellenberg fertility values (Hill *et al.,* 2000; Ellenberg, 1988), as an indicator of high nitrogen deposition. The ground flora of a particular habitat is expected to contain a certain range of herb/grass/sedge species; presence or absence of key species would be regarded as an indicator of nitrogen or acidification influences (but not necessarily the source). Changes in temporal or spatial patterns might identify sources. There is certainly potential in such a scheme; vegetation classifications have already been carried out across the UK, including Rodwell's National Vegetation Classifications (NVC) (see Rodwell's *British Plant Communities* 1991-2000), and Clapham *et al's The Flora of the British Isles* which was first published in 1952. There is also an extensive literature on local lichen information gathered by dedicated experts, plus soil maps also for the whole of the UK, as well as temperature, rainfall and other climatic data. From this type of information it may be possible to assess "deviation from normality" for a given site.

Top-down approach

This approach essentially involves building on the output of existing air quality models. Predicted ground-level air concentrations are used to forecast the outcome likely to arise from exposure to those concentrations. Such a model would in essence be an embodiment of the expert judgement that currently goes into setting the generically acceptable concentration levels. By incorporating the site-specific factors discussed above, however, a site-specific concentration level, and by back-calculation, emission level, could be calculated. This would meet the "Modern Regulation" objectives of targeted and proportionate control measures. For example, a calcium–rich soil might provide ‘protection’ against harm from SO_2, so that locally the critical level of SO_2 could be increased. A wet climate might make exposure to dust less of a problem (because it is washed off) than in a dry climate, while shallow soils and a dry climate might mitigate against stomatal uptake of pollutants because of drought-induced stomatal closure.

Models have been developed on the basis of physiologically-related exposures. For example, the AOT40 metric for assessing risk of damage to vegetation from ozone[7] uses the accumulated exposure above a threshold concentration during daylight hours in the growing season as a measure of predicted yield loss for crops, and has been extrapolated to semi-natural vegetation. Similar methods have been used in the United States, where the NCLAN (National Crop Loss Assessment Network) project (see for example, Lefohn and Foley, 1992) used controlled exposures of crops in open-top chambers to assess the likely yield-loss of economically important crops. These models all use either measured or modelled atmospheric concentrations, or deposition, as drivers, and are used to predict actual economic losses, or are projected ‘backwards’ to assess the thresholds for regulatory purposes, based on an acceptable level of damage.

6.4 Discussion

A top-down effects-based modelling approach, as described above, differs from the concentration-based approach in terms of end point. While current air quality models produce a pollutant concentration which can be compared against generic air quality objectives or limit values, an effects-based model will go further, taking into account site specific factors which will impact upon the effects caused by modelled pollution concentrations. The latter’s end point is therefore an assessment of the site-specific consequences of a given level of pollutant emission.

The biggest obstacle to producing such models is a shortage of data on pollution effects under a wide range of conditions. Effects calculations are of necessity currently rather crude. For example, the critical load approaches that define different values for different habitats, and critical levels that are defined differently for winter/summer for conifer forests (see Section 4.1) are based on average air concentrations over specified exposure periods or annual deposition rates. Critical loads and levels also contain large uncertainties due to the fact they are based predominantly on a few field experiments, particularly in the case of critical loads of nutrient nitrogen to soil, and model-based critical loads for acidity. Critical loads in these cases could be ±20% of any given critical load value. Moreover, the current critical loads map for the UK is based upon a relatively coarse grid based on dominant conditions over areas in the order of a square kilometre. Critical sites (such as Natura 2000 sites, which have statutory protection against adverse pollution effects) may represent only a small fraction of such a grid square. Effects-based models for the purposes of pollution regulation require a detailed understanding of the controlling biological, chemical and physical factors, which can be

[7] See: http://icpvegetation.ceh.ac.uk

applied to specific sites. While the critical loads and levels approach could provide a useful screening tool for sites at risk, it is only the use of site-specific models or monitoring that will provide a clearer understanding of the potential outcomes at any particular site. Other work commissioned by EA Science has, however, begun to address this matter and the findings of that study ("An investigation into the best method to combine national and local data to develop site-specific critical loads" EA Science Report SCHO0905BJRZ-E-EP) are a valuable step towards making effects-based modelling feasable.

The "bottom up" approach to effects-based modelling aims to make the best possible prediction of what ecological conditions should exist in a given location in the absence of pollution pressures, providing a baseline against which actual conditions on the ground may be compared. While of less use for specifying what changes to the control measures are required in order to bring about favourable ecological status, such models are of great value in determining whether or not existing control measures are adequate. Moreover, by comparing actual and projected ideal conditions, some indication is given as to the extent of the *effect* experienced at a site. By considering the cost (financial and otherwise) of this effect, decisions can be made as to the net benefit to be gained by applying further (or fewer) controls on a pollution source. Regulation is thus seen to be proportionate to the threat of harm.

7 Summary, conclusions and recommendations

This review has investigated the potential for use of Air Quality Outcomes in regulating emissions of airborne pollutants from individual or groups of point sources. On the basis of this investigation the following conclusions can be drawn:

7.1 Summary

Based on the definitions of ‘effects’, ‘impacts’ and ‘outcomes’ introduced here, the *outcome* of a particular course of action must be **causally linked through a demonstrable pathway or set of linked cause-effect relationships to the pollutant emissions** that are subject to regulation. This is true even where there is no **direct** link that can be measured in the eventual organism or community affected.

Implicit in the concept of an *outcome* is that the change produced in the surrounding species or ecosystems is of sufficient importance (evaluated in terms of biodiversity, amenity or economic value) that the costs of mitigating such a change, by introduction of emission controls, are proportionate. This necessarily implies the economic valuation of biodiversity and amenity, which may not be straightforward.

While the current literature reveals no published examples of a full outcomes-based approach to regulation, several case studies from the published literature have been used to illustrate:

The range of knowledge required to guide measurements which can potentially determine an *outcome*;

The need to incorporate explicitly any confounding factors in sampling design;

The need to use biomonitoring before, during and after the introduction of control measures as a means of assessing *outcomes*;

Current regulatory approaches, based on environmental concentrations of pollutants to which receptor organisms may be exposed, rely on generalised agreed links between the exposure to pollutants (in terms air concentration or deposition flux) and consequent *effects*. There is little flexibility for adapting emission controls for site-specific factors. Compliance is relatively easily demonstrated, but does not guarantee that vulnerable species or ecosystems have been protected.

An outcomes-based approach requires detailed prior assessment of the likelihood of *outcomes*; that is to say, *effects* that have an estimated value that is significant compared to net benefits gained and/or the cost of emission controls.

The main benefit of an *outcomes*-based approach lies in the use of tangible, site-specific evidence relating to consequences of emissions or emission control, which can be given a quantitative value. This approach adds robustness and auditability to the justification for stipulations laid down by regulators. Besides giving an appropriate level of environmental protection these stipulations will consequently be more difficult to contest, either by those who think them too harsh or by those who would see a greater level of restriction on emissions. In either case the onus would be on the contesting party to demonstrate that a more appropriate level of protection could be achieved by changes to the regulatory regime. An *outcomes*-based regulatory regime would give greater flexibility to the operator, in that operational practice is not necessarily prescribed; rather, it is the *result* of effective control that is defined.

Practical recommendations as to how an *outcomes*-based approach might be used in assessing and regulating air pollutants have been given in a flow diagram and are summarised below. Site-specific criteria for demonstrating compliance must be agreed in advance, in terms of quantifiable changes to the surrounding biota. These criteria may need to be expressed relative to changes at unaffected sites, to allow also for non-polluting environmental effects (so-called confounding factors).

One drawback to a purely *outcomes*-based approach to regulation is the likelihood of a significant time-lag between pollution releases and observed consequence, with removing the possibility of an immediate control response to changes in emissions. Action in these circumstances would depend on some harm already having been done. The use of "real-time" physico-chemical monitoring – which would in any case often still be required to enforce human health-based air quality standards – could be used in conjunction with an outcome-based network to provide a warning of when additional sampling for early signs of ecological *effects* might be required.

One might anticipate that a significant area of uncertainty, or at least of contention, in the use of *outcomes*-based regulation, might be the assignment of value of the ecosystem impacted. This topic has been considered widely over the last decade, during which Ecological Economics has emerged as a discipline in its own right (see for example Costanza, 1997, and Princen, 2005). The task of assigning value, so pivotal to the feasibility of an *outcomes*-based regulatory system, is becoming increasingly systematised and objective.

In practice, the implementation of *outcomes*-based regulation would require the use of tools analogous to the dispersion models currently used to support concentration-based regulations. Such a tool would take into account current knowledge of the impacts of pollutants on organisms, as well as site-specific factors, to determine the likely *effects* arising from a given regime of pollution emission. The *effects* could then be used in association with current estimates of ecosystem value to calculate whether or not the predicted *outcome* would be acceptable.

A comparison is required between the observed state of an ecosystem affected by a regime of pollutant emission and the state that might be expected were those emissions not present. This could be achieved in practice by a “bottom-up” modelling approach that describes the ecosystem status normally expected for a given combination of physical parameters (such as latitude, soil-type, microclimate and underlying geology). The RIVPACS model provides an example of such a model that is widely used for assessing the status of river systems. While not originally designed for outcomes monitoring, this “bottom-up” modelling approach suggests a means by which base-line ecological status might be defined.

7.2 Conclusions

Concentration-based regulations have served us well in reducing the adverse *outcomes* which arise from releasing air pollutants into the environment. However the easier targets for control – those yielding large environmental improvements at relatively low cost – have now largely been addressed. Where adverse environmental *outcomes* persist, the control options remaining involve greater cost for a smaller increment of environmental benefit. Such options are therefore increasingly likely to be contested by polluters, as imposing a cost burden on them without a commensurate improvement in environmental health. At the same time, public concern about the state of the environment remains high, and it is necessary to demonstrate that an acceptable level of environmental protection is being achieved.

Regulation based on *outcomes* offers a way to tighten the coupling between emissions and their consequences, to a greater degree than concentration-based regulation allows. As a consequence, a reduction in, or absence of, harm is more likely to be achieved, while a clearer link made between control measures and the benefits they deliver can demonstrate the justification for imposing control measures. **The result of this would be a better-protected environment alongside a greater consensus that regulation is both effective and proportionate to the threat it aims to mitigate.**

Obstacles to the practical implementation of outcomes-based regulatory regimes may be:

Lack of information on the links between emissions and their environmental consequences, under realistic, site-specific conditions and taking into account such confounding factors as local climate, topography as well as growing conditions such as soil type and availability of water;
Lack of sampling protocols and monitoring tools which could be implemented with sufficient statistical rigour to assess ecological status at the required levels of temporal and spatial sensitivity;
Lack of a rigorous framework for assigning monetary and especially non-monetary values to a particular ecosystem.

All three of these areas have been the subject of considerable work in recent years, as demonstrated, for example, by the case studies in this report and the biological monitoring methods described in its sister report under this EA Science project.

We are therefore in a stronger position than ever before to consider the practical implementation of *outcomes*-based regulation. With the growing need to demonstrate that regulation is Targeted, Proportionate, Testable and Evidence-based, we are perhaps also more obliged than ever to consider the implementation of such regimes.

7.3 Recommendations

- This report suggests that the necessary pieces are in place to enable us to consider the practical implementation of an outcomes-based regulatory regime, and that there are compelling reasons to do so. In order to develop a practical framework for such implementation, and as a further test of the feasibility of such an approach, **we therefore recommend that a pilot study of the implementation of an outcome-based regulatory regime should now be carried out**.

- Outcomes-based regulation requires the best available information on the linkage between air quality and environmental health (as, indeed, does effective level-setting for concentration-based regulation). **We therefore recommend continued monitoring of scientific developments that will allow improved quantitative coupling of environmental pollutant concentrations with their effect on environmental health.** Such developments should be logged in a form readily accessible to those tasked with developing regulations, perhaps through the medium of APIS, the Air Pollution Information System set up under part-sponsorship by the EA.

- Costed *outcomes* will constitute a key part of the evidence-base justifying regulatory intervention. Conversion of a quantified *effect* on environmental status into such a costed *outcome* depends on being able to assign a value to the aspect of the environment being affected. **We recommend development of a robust framework, incorporating recent developments in the field of environmental economics, which may be used to assign values to changes in the status of specified ecosystems in an auditable and defensible way.**

8. Abbreviations

APIS	Air Pollution Information System
AQO	Air Quality Outcome
CL	Critical Load
Dbh	diameter at breast height
NCLAN	National Crop Loss Assessment Program
PAH	Polycyclic Aromatic Hydrocarbon
PCBs	biphenyls
PCDDs	polychlorinated dibenzo-p-dioxins
PCDFs	polychlorinated dibenzofurans
PNEC	predicted no effect concentration
RIVPACS	River InVertebrate Prediction And Classification Systems
SO_2	sulphur dioxide
TCDD	tetrachlorodibenzo-p-dioxin
TCDF	tetrachlorodibenzofuran

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