Eutrophication series

Overview of the state-of-the-art of models and their use in OSPAR predictive eutrophication assessments

OSPAR Commission

2008

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. It has been ratified by Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, Netherlands, Norway, Portugal, Sweden, Switzerland and the United Kingdom and approved by the European Community and Spain.

La Convention pour la protection du milieu marin de l'Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d'Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998. La Convention a été ratifiée par l'Allemagne, la Belgique, le Danemark, la Finlande, la France, l'Irlande, l'Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume-Uni de Grande Bretagne et d'Irlande du Nord, la Suède et la Suisse et approuvée par la Communauté européenne et l'Espagne.

ISBN 978-1-906840-14-3
Publication Number: 373/2008

contents

Executive summary

This overview report brings together information and experience established by OSPAR and other fora, and from scientific literature on the state-of-the-art of models and their use in predictive eutrophication assessments. It builds on a series of model applications tested by experts in nutrient reduction scenarios through the Intersessional Correspondence Group on Eutrophication Modelling under the remit of the OSPAR Eutrophication Committee. Such scenarios and future work on transboundary nutrient fluxes are intended as tools by the OSPAR Joint Assessment and Monitoring Programme to assist OSPAR in prospective evaluations of the progress made towards the overall objective of the OSPAR Eutrophication Strategy to achieve and maintain, by 2010, a healthy marine environment where eutrophication does not occur, and in the direction of future actions to achieve this objective.

The report gives an overview of the state-of-the-art of available predictive models, examples for use of models in nutrient reduction scenarios and addresses the needs for further development of modelling work to meet policy requirements.

Récapitulatif

Ce rapport d'ensemble rassemble les informations et expériences établies par OSPAR et par d'autres forums, ainsi que provenant de la littérature scientifique sur l'état de l'art dans le domaine des modèles et de leur usage pour les évaluations prédictives de l'eutrophisation. Il est construit sur des séries de modèles testés par des experts en scenarios de réduction des nutriments, à travers le groupe de travail intersessionnel par correspondance sur la modélisation de l'eutrophisation, sous l'égide du comité OSPAR sur l'eutrophisation. Ces scenarios et travaux futurs sur les flux transfrontaliers de nutriments sont destinés à être les outils du Programme conjoint d'évaluation et de surveillance continue, pour assister OSPAR dans les évaluations prospectives sur les progrès réalisés pour atteindre l'objectif global de la stratégie OSPAR sur l'eutrophisation – atteindre et maintenir d'ici à 2010, un environnement marin sain où l'eutrophisation ne se produit pas – et les directions à prendre pour les futures actions pour atteindre cet objectif.

Le rapport donne une vue d'ensemble sur l'état de l'art des modèles de prédiction disponibles, des exemples d'utilisations des modèles dans les scenarios de réduction de nutriments et présente les besoins pour les développements futurs des travaux de modélisation pour atteindre les exigences des politiques.

1. Introduction

The OSPAR Joint Assessment and Monitoring Programme (JAMP) (OSPAR agreement 2003-22) envisages further work in the period 2003 – 2010 on predictive models as one of several tools supporting OSPAR eutrophication assessment work. This is to assist prospective evaluations by OSPAR of the progress made towards the overall objective of the OSPAR Eutrophication Strategy to achieve and maintain, by 2010, a healthy marine environment where eutrophication does not occur, and to help directing future actions. The JAMP requires an overview of predictive models for eutrophication assessment and nutrient reduction scenarios, including tansboundary fluxes within the OSPAR maritime area, and of the possibilities of adopting relevant models for use by OSPAR Contracting Parties. To this end, existing national and international modelling activities relevant for OSPAR eutrophication work has been reviewed with a view to exploring possibilities for cooperation and coordination of these activities.

The present report builds on the continuous work of the Intersessional Correspondence Group on Eutrophication Modelling, under the remit of the OSPAR Eutrophication Committee, which has tested relevant models in nutrient reduction scenarios in a series of workshops. The Intersessional Correspondence Group has prepared this overview by combining the experience from their and previous OSPAR work with information from scientific projects in other fora and from scientific literature to give an account of the current state-of-the-art of available models and examples of their use in exploring nutrient reduction scenarios, and point to development needs to meet policy requirements. The degree of model complexity required for eutrophication modelling is identified as a critical issue to be considered.

2. State of the art

Models can play an important role in helping to predict and diagnose the anthropogenic process of eutrophication. This section identifies some of the processes that need to be incorporated in such models, the range of possible approaches and the degree of complexity that may be implemented illustrated with examples.

The process of (pelagic) eutrophication can be seen as falling into 3 stages:

(1) nutrient addition at a rate sufficient to overcome dilution or losses such as denitrification and leading to potential enhancement of local concentrations;

(2) stimulation of phytoplankton growth by these extra nutrients (given sufficient light), followed by the accumulation of extra biomass resulting from this growth (which will not occur if losses of phytoplankton increase to match increased growth), all of which may be summed up as increased primary production;

(3) the possible consequences, especially the harmful ones, of the increased production or of change in the balance of organisms resulting from relatively greater stimulation of some species or types of phytoplankton).

In each case, models must simulate key features of the physical environment as well as relevant chemical and biological processes. There are two matters to be considered here. First, the environment may be described as a simple box (a point, or 0-D, model), or in terms of variation along 1, 2 or 3 dimensions. However, the simulation of complex physical environments requires detailed seabed topography and more information about initial and boundary conditions, as well as creating more difficulties for numerical integration of model equations. Second is the matter of the degree of complexity in the chemical and biological models. Here the main problem is that of finding good values for parameters that increase in number, generally more than proportionately, to the number of state variables. Models that are simple both physically and biologically may, therefore, have advantages which may outweigh their lack of detail.

Except in cases where denitrification is an important process, requiring chemically complex models for its description (Middelburg et al, 1996; Di Toro, 2001), stage 1 of eutrophication is comparatively easy to simulate. The simplest approach balances nutrient inputs against dispersion losses from a box. Such a model, used to assess 'Equilibrium Concentration Enhancement' (ECE) of nutrients (Gillebrand & Turell, 1997), has proven useful at identifying Scottish sea-lochs most at risk from fish-farm nutrients. Gillebrand (2001) used a 2-D physical model of a sea-loch to examine some of the approximations involved in a box model.

Stage 2 concerns the conversion of nutrients into biomass and hence requires biogeochemical models. These were defined by Tett & Wilson (2000) as conserving the totals of the elements simulated, with the advantage of constraining the outcomes of simulations and thus rendering prediction more reliable. One of the simplest of these models was developed by the UK's Comprehensive Studies Task team (CSTT, 1994, 1997; Tett, 2000) and uses a single parameter for the yield of phytoplankton chlorophyll from nutrients to convert ECE nutrients into worst-case biomass. This use of yield was proposed by Gowen et al. (1992) and the value of the yield parameter was investigated by Edwards et al. (2003). The CSTT model has recently been applied to 6 semi-enclosed coastal waters studied by the OAERRE project which concerned "Oceanographic Applications to Eutrophication in Regions of Restricted Exchange" (Tett, Gilpin, et al., 2003).

At the next level of complexity are models with between 3 and 10 state variables yet which remain tightly constrained by the biogeochemistry of nitrogen cycling through a single productive compartment. These include the "strategic fjord simulation model" of Ross et al. (1993a, 1993b, 1994) and the microplankton model used by Tett & Walne (1995) in a simulated 2 layer water column, by Smith & Tett (2000) in a 1-D, depth resolving, model, and in the 3-D model COHERENS by Luyten et al. (1999). Also in this class is ECOHAM1, which simulates phosphorus-limited phytoplankton within a 3-D representation of the North Sea (Moll, 1998; Skogen & Moll, 2000).

Some of the undesirable consequences of stage 3 of eutrophication can be dealt with by comparatively simple models. An example, dealing with transparency and oxygen concentration of fjordic waters, is the FjordEnv model of Stigebrandt (2001). These remain within the scope of biogeochemical modelling. However, some aspects of undesirable disturbance require simulations of food webs with many components and hence fall into the domain of 'ecological' modelling. This was defined by Tett &Wilson (2000) as involving at least one state variable that was not constrained by a conservation rule. Such models can give rise to simulations involving Lotka-Volterra oscillations or even chaotic behaviour, and hence lead to more uncertain predictions. However, such behaviour can be avoided by a suitable choice of parameter values, or by the inclusion of terms that dampen oscillation, for example through simulated switching between prey types. The best-known example of a complex ecosystem model is the European Regional Seas Ecosystem Model (ERSEM), which in its version II (Baretta-Bekker et al., 1997) emulates a system containing diatoms, dinoflagellates, autotrophic flagellates, "picoalgae", heterotrophic nanoflagellates, bacteria, microzooplankton and mesozooplankton. It has been successfully used by Pätsch & Radach (1997), with dinoflagellates replaced by Phaeocystis, to simulate changes in the diatom-flagellate balance induced by anthropogenic nutrient enrichment of the southern North Sea. Initial applications of ERSEM used a multicompartment representation of space, with stratified waters divided into an upper 30 m layer and a lower water column. Proctor et al. (2004) have embedded ERSEM in a highly-resolved 3-D simulation of North Sea circulation in order to calculate nutrient fluxes between different parts of the north-western European continental shelf. More recently, Polimene et al. (2006) performed a similar coupling for the Adriatic, while Vichi et al (2007) have re-coded ERSEM into BFM (Bio-geochemical Flux Model), and embedded it in a global ocean circulation model. ERSEM is also used in operational forecasting (Siddorn, 2007).

ERSEM uses "characteristic organisms", each such organism being a bulk parameterisation of a single population of a typical flagellate, diatom, etc typifying a normally heterogenous mixture. The lower part of the pelagic food web is assembled explicitly out of links between the characteristic organisms. In contrast, the microplankton models used by Tett & Walne (1995) and Smith & Tett (2000), deal with microbial processes in a different way. A microplankton compartment contains both autotrophs (phytoplankton) and microheterotrophs (bacteria and protozoa), with the parameter, η, giving the ratio of microheterotroph to total microplankton carbon biomass. The simulated compartment can be thought of as a mixture of chloroplasts carrying out photosynthesis and driving nutrient assimilation, and mitochondria, responsible for respiration.

Diatoms sink, and flagellates can swim small distances vertically under calm conditions, and the simulation of these motions may be important for predicting the balance of organisms. The microplankton model of Tett & Walne (1995) was implemented in a physical structure of 2 layers (of variable thickness summing to a constant total. ERSEM was originally implemented in 2 thick layers – the upper water column down to 30 metres, and a deeper water layer, with vertical exchange (as well as lateral exchange between boxes) taken from a separate physical model. In both cases, such a structure produces a large numerical enhancement of any advective term and does not allow the formation of midwater features such as a deep chlorophyll maximum. Ruardij et al. (1997) implemented ERSEM within a depth-resolving 1-D physical model and found that properly simulating stratification had "a major impact on the biota". ERSEM version III has also been implemented in a 1-D framework and applied to the Baltic and Adriatic (Vichi, 2002; Vichi, 2003; Carniel, 2007) and also recently in unpublished work in the North Sea.

A depth-resolving 1-D model has advantages over a 3-D model when it comes to numerical experiments: it is computationally much simpler, and there is no need to acquire and apply data for extensive, and possibly poorly known, lateral boundary conditions. However, a 1-D model is inaccurate in waters where lateral transport fluxes are substantial, as is the case in the anthropogenically enriched coastal waters of the southern North Sea, for example at the PROVESS southern site (Wild-Allen et al, 2002).

The case for complexity and the use of ecosystem models within a 3-D hydrodynamic framework is made by Moll and Radach (2003). They identified 11 three-dimensional coupled physical-biological models for the North Sea. They reviewed 7 in more detail in terms of the complexity, spatial and temporal resolution, the degree of trophic complexity and the processes relating state variables to each other.

They concluded that 7 (NORWECOM, GHER, ECOHAM, ERSEM, ELISE, COHERENS and POL3dERSEM) of these three-dimensional ecological models of the greater North Sea have provided consistent distributions and dynamics of the lower trophic levels on their regional, annual and decadal scales. The results from these model simulations have either confirmed existing knowledge derived from field work or given new insight into the ecosystem structure and function in the North Sea. Model simulations have contributed to improved knowledge of the temporal and spatial development and magnitude of primary production, its mechanisms of limitation, the mechanisms of nutrient regeneration, the effects of riverine and atmospheric nutrient inputs causing eutrophication of coastal waters and the budget for nutrients.

Three of the models, reviewed in more detail by Moll and Radach (2003) have been applied by groups within the UK. These include; POL3dERSEM, ERSEM and COHERENS. Recent applications of POL3dERSEM (now termed POLCOMS-ERSEM) have been carried out within the EU funded MERSEA Strand 1 project (Allen et al., 2004). An important conclusion from this work was the need to improve representation of the underwater light climate in shallow (< 20 m) coastal waters in order to better simulate the light-dependent growth of phytoplankton and thereby their response to changes in nutrient input. As the underwater light climate in these waters is largely determined by suspended matter concentration, the representation of the deposition and resuspension processes needs to be improved.

Moll and Radach (2003) concluded that lack of complexity was a weakness in many ecosystem model formulation and they cited ERSEM as an example of a model with the necessary degree of complexity for realistically simulating the North Sea system.

A physical model is a general requirement to provide a framework for application of the biological model as noted earlier. Skogen et al (2005) showed that the choice of the physical model is decisive in determining the response of a coupled ecosystem model. The state of the art of physical modelling will not be dealt with here but a review has been carried out by (Lenhart & Pohlman, 2004). Nevertheless, it is important to be aware of current limitations particularly in the application of 3-D hydrodynamic models. These authors note the need for reliable data on boundary conditions. Obtaining such information, for example, on riverine inputs of freshwater and the associated flux of dissolved and particulate nutrients is a non-trivial task and should be recognised as a potential limitation on the reliability on coupled physical-biological models used in relation to eutrophication. As a general rule the simpler the physical model the more important the accurate specification of boundary conditions becomes.

3. Scenario testing

Multiple effects are difficult to distinguish in field data and it is the power of models to distinguish between multiple effects, which makes them very good tools to carry out (numerical) experiments on the effects of nutrient reduction. In particular, models may be used to distinguish between riverine or atmospheric inputs of nutrients. A further example would be to distinguish between the effects of bed-load transport of particle-bound nutrients and the transport of dissolved nutrients.

An earlier initiative related to scenario testing included the workshop on “Modelling of Eutrophication Issues”. This workshop was the second in a series organised by the Environmental Assessment and Monitoring Committee (ASMO) of the Oslo and Paris Commission (OSPAR). The first part of the workshop examined the conceptual basis of the models and evaluated the agreement between model results and in-situ data. The responsiveness of models to anthropogenic input reduction was the subject of the second part of the workshop. The model responsiveness to the following scenarios was tested and compared;

- the effects of actual trends in riverine inputs, e.g. a 50% reduction of phosphate loadings between 1985 and 1995, as indicated by the available data sets;
- the potential effects of 50% reduction of anthropogenic inputs of both phosphate and nitrogen.

It was shown that there is a wide range of predicted response to nutrient load reductions. The responsiveness exercise has shown that a nutrient load reduction of 50% does not linearly translate into a 50% reduction in any of the chosen measures of eutrophication, such as average annual primary production, maximum primary production, summer mean chlorophyll concentration, peak chlorophyll concentration, winter mean chlorophyll concentration, winter mean nutrient concentrations or oxygen depletion. The models in general predict a greater responsiveness in the coastal regions than in the northern North Sea.

Further published work (Lenhart et al., 1997) using ERSEM II, investigated a 50% reduction in river nutrient loading to the North Sea. Net primary production decreased by up to 15% in the coastal region but barely 2% offshore. They concluded that the discharges of the major rivers hardly affect the central North Sea but lead to significant changes in nutrient limitations and mass flows in the coastal area. Although the model was run for only 1 year the initialisation of the standard run and the reduction scenario is obtained by running the model for 30 years with repeated forcing to generate repeating annual cycles. As a result the (slow) effects of depleting benthic nutrient pools under long-term reduced nutrient inputs will have adapted to changes in the environment.

EU funded work carried out between 2000 and 2004 (Eurocat; www.iia-cnr.unical.it/EUROCAT/) aims to study the impact on coastal water quality of future socio-economic changes in European river catchments. This is a large European project researching 8 major European catchments and their coastal zones. A suite of models has been coupled to provide an integrated approach to scenario testing that is linked to a socio-economic assessment. The model suite that includes ERSEM II is used to investigate 3 specific nutrient reduction scenarios; DG (Deep Green), PT (Policy Target) and BAU (Business as usual) defined within the Eurocat project. The standard ERSEM II model was driven by a hydrodynamic model (HAMSOM), which provided the required transport information (http://www.ifm.uni-hamburg.de/~wwwsh/res/HAMSOM/hamsom.html). Work describing results for the Elbe and German Bight can be found at http://www.springerlink.com/index/10.1007/s10113-004-0082-y. While simulations near shore, with the emphasis on the catchments, may result in realistic simulation for the coastal zone there is greater uncertainty in the robustness of the results for offshore regions.

An important outcome from the earlier work on scenario testing is the apparent lack of responsiveness of the North Sea system to reductions in nutrient loading, on the basis of model output. There does not appear to be a linear relationship between nutrient reduction and changes in the level of assessment variables.

An OSPAR workshop on Eutrophication Modelling (University of Hamburg, September 2005) set out to determine the current state of the art and to carry out nutrient reduction scenario tests. The report on workshop results, conclusions and recommendations for future work is available at http//www.cefas.co.uk/eutmod. The workshop concluded that models could be usefully applied to support the application of the OSPAR Common Procedure. Models could be used in a variety of ways to directly support the assessment process, for example, through extrapolation to provide model results on the levels of assessment variables. Also models could indirectly be used to explore the behaviour of indicators under a variety of condition or be used to identify potentially new indicators of eutrophication. In order to facilitate the application of models for the workshop the convenors collated key datasets for the forcing and boundary conditions and provided access to these via an FTP site. An important outcome from the workshop was to bring together a comprehensive data set on continental and UK riverine nutrient inputs never before accomplished. A novel aspect of the nutrient reduction scenario tests reported at the workshop was the reconstruction of the Comprehensive Procedure step 2 of the OSPAR Common Procedure (agreement 2005-3) using model results for assessment variable levels. A comparative exercise formulated specifically for the workshop required models to be run for pre-defined target areas previously classified as Problem Areas following the first application of the Common Procedure. The target areas were in Dutch and German coastal and offshore waters. The responsiveness of the modelled assessment parameters varied between different models but in general showed a larger response in coastal rather than offshore water bodies. Further work was identified and proposed as a result of the workshop to confirm and bolster confidence in the results. Models used to prepare results for the workshop and presented at the workshop are listed in the table below.

Country	Model type	Application area	Biological model	State variables	Benthic included
Belgium	3D	Channel area	MIRO&CO-3D	32	yes
France	3D	South channel area	ECO_MARS3D	19	yes
	box	North-France lagoons	Elise		
Germany		North Sea	ECOHAM3	26	yes
Netherlands	3D	North Sea	Delft-3D_GEM	21	yes
	box	Wadden sea	EcoWasp		
Norway	3D	North Sea	NORWECOM	8	yes
Portugal	3D	Tagus estuary	WASP		yes
Sweden	box	Skagerrak/Kattegat	under development		
UK	3D	North Sea	GETM_IOW	9	no

Nutrient reduction scenario tests for the target areas were carried out by 4 groups (Germany, the Netherlands, Norway, UK) for 5 appointed target areas in the Dutch and German North Sea territories, plus at least one additional area for each participating country. The scenarios included 20, 30, 50, 70 and 90 % reductions in riverine input into the North Sea. A limited, non-linear response of the ecosystem was observed in coastal areas. The offshore target areas showed no influence of riverine input reduction, except for the UK Southern Bight offshore area (modelled only by the UK).

Results presented by Belgium and France showed the capability of models with respect to transboundary transport of nutrients. In particular, the French delegation offered a simple means of tracking nutrient input from different rivers within a hydrodynamical model, thus allowing quantification of transboundary nutrient transport. Results from the French and Belgium contingents showed the influence that the French rivers can have on nutrient input in the Southern Bight of the North Sea.

A follow-up OSPAR workshop was organized again by the ICG-EMO (Intersessional Correspondence Group on Eutrophication Modelling) at Cefas in Lowesoft in September 2007. The OSPAR objectives for this workshop were: “to carry out nutrient reduction scenarios for 2002 showing the predicted consequences for selected eutrophication assessment parameters in problem areas with riverine nutrient reductions of 50% and 70% since 1985 (the OSPAR reference year)”. The reductions in phosphate, nitrate and ammomium per country between 1985 and 2002 had to be taken into account. Therefore, first the river load data set was extended considerably, and the data were analysed in terms of the achieved reductions. Based on this assessment, individual reduction scenarios were prescribed for each country to represent the overall 50% and 70% reductions. In addition to the OSPAR objectives, also the requests from the Hamburg 2005 workshop for improved spin-up and the use of a common set of boundary conditions had to be taken into account. The spin-up procedure required the models to reach equilibrium by repeating at least 3 annual runs. Since the boundary data could not be provided from measurements, especially not for the reduction runs, the boundary data were extracted from model runs with the AMM POLCOM/ERSEM model, which was selected because it has the widest regional extent and therefore offered the opportunity to cover the model domains of most of the models which took part on the workshop. The boundary data were extracted for the hindcast run for 2002 and the two reduction scenarios, and supplied for the MIRO&CO-3D model (Belgium), ECO_MARS3D (France), ECOHAM4 (Germany), Delft3D-GEM (Netherlands), NORWECOM (Norway), MOHID System (Portugal) and the Cefas GETM_BFM model (United Kingdom). In addition to this extensive set of boundary conditions, data on the atmospheric nitrogen deposition were made available from EMEP on a monthly basis for the years 2001 and 2002.

After the national presentation for each individual model, the results for a selected number of parameters from all models were presented in one graph for each target area and discussed with the participants. This exercise led to the statement from the participants that the major goal was reached by achieving comparable results from the different models mainly by the provision of the common set of boundary conditions and that all participants run their model into equilibrium. Some of the participants used longer spin-up times of up to ten years, mainly depending on the complexity of the benthic module used.

Generally the largest reductions in the winter nutrient concentrations occurred in the target areas which represent coastal waters of less then 34.5 salinity. In the coastal areas the results exhibit a strong response in the winter DIN concentrations to the reduced loads. Offshore the decreases are less pronounced in DIP but still apparent in DIN. These findings are consistent with the setup of the river load reduction for the individual countries, where the DIP loads are reduced to a much lower degree for a number of countries where phosphorus reduction has already been achieved between the years 1985 and 2002. The overall response to the chlorophyll concentration vary between 11% – 36 % in comparison to the hindcast run for 2002. For the parameter oxygen concentration, which should indicate the potential problem of oxygen deficiency, the mean oxygen concentration over the target areas appeared to be problematic. The discussion between the participants led to the conclusion that the minimum oxygen concentration and the time during which the concentration is below the threshold set by the Common Procedure would be a better assessment criterion.

In terms of model performance the need for an improved suspended matter representation was pointed out in order to get a better representation for the light attenuation especially within the coastal areas. Furthermore the problem of the organic river load was discussed, which can not be quantified in the reduction scenarios since the organic load consistent with the reduction of DIN and DIP in the rivers is not know. However, in comparison to the SPM problem the organic load problem was categorized to be of minor importance. Finally the use of the cost function for the validation of the model results was discussed. Since the available data for several target areas were too sparce in space and time to calculate representative means and standard deviations comparable to those calculated from the model results, the results of the cost function could be biased, suggesting that model results are worse than they actually are. This bias can only be reduced by including more observations to reduce the sparcity, or by making a point-to-point comparison rather than comparing averages. Lessons from previous exercises documented in the literature (e.g. Allen et al., 2007) should be taken into account in improving the cost function formulation. The full workshop report is available at http://www.cefas.co.uk/eutmod; based on this OSPAR adopted a report on nutrient reduction scenarios (OSPAR, 2008).

4. Future needs

There is a need for wider debate amongst the key players in order to share experiences and identify, when appropriate, opportunities for collaboration. A number of issues identified below would benefit from joint discussions and in some cases shared effort.

As a general conclusion, Moll and Radach (2003) note the lack of complexity as a drawback in most models they reviewed apart from ERSEM. They also note the lack of a systematic way of determining the necessary complexity required in a model in relation to the particular application. This is an important conclusion with regard to future scenario testing. It is therefore important to consider the necessary degree of complexity to realistically simulate the consequences of nutrient reduction scenarios in relation to eutrophication. Examples of the key biogeochemical and ecological processes that need to be realistically described (and not over-simplified) and robustly parameterised include:

- nutrient and carbon dynamics
- oxygen dynamics – to include air-sea exchange
- microbial dynamics and balance of organisms
- benthic-pelagic coupling
- controls on sub-surface irradiance (suspended matter deposition and resuspension)

Such a list needs to be established and potentially extended through further dialog with relevant parties. As part of such a dialog, a rationale for determining model complexity should be developed.

With regard to the nutrient reduction scenario testing it is likely that the benthos plays a crucial role in buffering the system, through internal nutrient pools, against rapid changes in external nutrient forcing. It is critical that models used in scenario testing include the necessary level of complexity to ensure that this ‘buffering’ role is properly described. Moll and Radach (2003) specifically identify shortcomings in the representation of sediment chemistry. They note the absence of burial and remineralisation of organic matter in most of the models cited, with the exception of ERSEM. Evidence of the role of the benthic system in nutrient regeneration is provided by Vichi, 2002 .

There is also a requirement to ensure that model runs are sufficiently long to enable any new equilibrium to be achieved. Evidence from unpublished work with a 1-D hydro-dynamically coupled version of ERSEM III suggests that the effect of a reduction in nutrient loading is not apparent until approximately 5 years later. In order to allow realistic multi-year runs the appropriate meteorological forcing data must be available. The results of the OSPAR 2005 workshop should also be viewed in this light: the prescribed run up period of 1 year may not be sufficient to fully capture effects of the reduction scenarios. The 2007 workshop imposed a spin up period for this reason, requesting every model to run repeats of the standard year until equilibrium was reached, and a similar relaxation period for the reduction scenarios. Results show that this repetition may condition the model towards the chosen year/scenario because of the absence of interannual variability. For the standard run, such a bias could be overcome by applying a multi-year spinup based on realistic years, but it is not trivial to apply that same method to reduction scenarios.

As ecosystem models achieve the necessary complexity and are coupled to appropriate 3-D hydrodynamic frameworks, checking models for realistic behaviour prior to operational use is difficult due to the high computational costs of long model runs and difficulties associated with interpreting complex data sets. One approach to address this problem efficiently is to carry out sensitivity analysis of the ecosystem model in a 1-D framework. The rapid run-times achievable on a typical workstation mean that models can be easily run for decades, allowing even long-term effects of any scenario to manifest themselves, prior to coupling the model code to the associated 3-D hydrodynamic model.

A debate regarding the most appropriate method for validation continues. This issue is addressed in the discussions at the Workshop on Future Directions on Physical-Biological Interactions held under the auspices of ICES in 2004. Of particular relevance to eutrophication modelling is the choice of state variables to compare with observations. Although most ecosystem models represent phytoplankton biomass as carbon they output chlorophyll concentration as a proxy of phytoplankton biomass and validate the model against Chl *a* observations. These values are typically derived from fixed carbon:chlorophyll ratios. Clearly, this ratio is variable in nature and as a result introduces uncertainty into the validation procedure. A better approach may be to choose system level quantities that are not subject to such uncertainties such as SPM, O_2 and nutrient concentrations. These state variables have the advantage of being directly measurable quantities, unlike phytoplankton carbon. With depth resolving physics, modelled and observed concentration profiles can be compared, providing key information on vertical dynamics. Oxygen is a key variable in relation to conveying information about system response to nutrient inputs as well as being regarded as an indirect indicator of eutrophication. For example trends in oxygen concentration will inform of the relative importance of auto- and heterotrophy and hence the likely response to nutrient inputs.

There is also a strong case for comparing modelled rates, such as primary productivity, with field data. The implication of such approaches to model validation is that the choice of variables to be monitored may need to be adjusted to ensure that future model applications are well-tested, robust and reliable.

The availability of sufficient river nutrient input data is of critical importance for realistic hindcasts of ambient nutrient concentration and the consequent response of the ecosystem. Lenhart and Pätsch (2001) have made 25 years of daily nutrient loads (ftp://ftp.ifm.uni-hamburg.de/pbu/data/riverload/) available. A similar data set has now been put together by Cefas for the UK rivers. A further effort is required for the other rivers entering the North Sea. Such an effort will need to be organised nationally via the appropriate agency tasked with monitoring river flow and riverine nutrient concentrations. A minimum requirement is for monthly resolved nutrient input data with daily nutrient loads the ideal frequency.

5. Supporting policy need through the use of models

Where models are used to support policy implementation (e.g. prospective assessment of eutrophication and nutrient reduction scenarios) or development there is a fundamental requirement for robustness and reliability. As a result the attributes of the model(s) to be applied need to be clearly understood and there needs to be clarity about the way in which the model will be used. A useful distinction is between models used as ‘engineering tools’ or as tools for testing hypotheses. In the first instance the answers must be reliable and stand up to scrutiny. This use is distinct from the latter scientific use where numerical models are designed to improve understanding by testing hypotheses. The desirable attributes of models as engineering tools is that they are not only robust and reliable but also transparent and easily understood with the implication that they are likely to be simple rather than complex in design. However, the debate regarding the need for increased complexity identifies a difference in philosophies within the scientific community that needs to be explored further. The way forward is likely to embrace both sides of the argument by identifying where in the system to be modelled complexity is required and elsewhere that simplicity will suffice.

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