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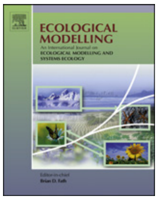

Description and sensitivity analysis for the LESV model: Water quality variables and the balance of organisms in a fjordic region of restricted exchange

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ABSTRACT

In this paper we describe a new ecological model for Regions of Restricted Exchange (RRE), such as fjords, estuaries, rias and lagoons. The model is intended to simulate the impact of external nutrient input on microplankton (phytoplankton plus pelagic microheterotrophs) in RREs. We have implemented the model with the practical purpose of finding a safe limit to the capacities of RRE to assimilate fish-farm waste. Sea-cage farming of fish is increasing in fjords in northern and southern hemispheres, and its external nutrient input can lead to environmental problems such as eutrophication and deoxygenation. The model includes a physical system of three layers with exchanges driven by tidal movement, freshwater input, wind stirring. The biological part includes two microplankton compartments, each parameterizing a microbial loop and each containing chlorophyll. The first compartment represents diatoms and associated heterotrophs, and the second compartment represents flagellates and associated heterotrophs. As well as the balance of these organisms, the model simulates concentrations of nutrient N, P, and Si, dissolved oxygen, and water transparency. Chlorophyll and nutrient change are linked by yields (q). Losses of microplankton to grazing by mesozooplankton or benthos are simulated by a temperature-dependent grazing pressure acting on a mean loss (L_0). The model also includes the ability to simulate point source inputs of nutrients or organic matter and a generic tracer with first order decay. Sea-cage fish-farms exemplify such point sources. In order to explore model behaviour, we included inputs from a 1500 tonnes salmon farm multiplied by a factor (γ). We carried out sensitivity analysis to identify the most influential model parameters and forcing variables in the case of the shallow Scottish fjord, Loch Creran, in 1975 before the introduction of salmon farming. We tested the model fit to this pristine state ($\gamma = 0$), using Major Axis Regression of simulated variables on observed variables. The model successfully follows the seasonal cycles of chlorophyll (summer over both microplanktons) and the limiting nutrients (P, N). The sensitivity analysis identified three sets of key parameters: (γ) and other fish-farm coefficients, which control farm waste effects on an RRE; (L_0) parameters for each microplankton, which link these to the rest of the ecosystem and which have implications for future inclusion of shellfish farming in the model and, chlorophyll yields from nutrients (q), which are crucial for the predication of eutrophication and the ecological understanding of the model.

1. Introduction

1.1. Eutrophication, aquaculture and ecological modelling

Many human activities introduce waste into the environment, thus harming biota and ecosystems and threatening their sustainable human use. Anthropogenic nutrients are one component of these wastes, and can lead to *eutrophication*, defined as "the enrichment of water by nutrients, especially compounds of nitrogen and phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and the quality of the water concerned" (OSPAR, 2003).

An illustrative case of a potentially harmful human activity, which will be focus of our work, is the marine aquaculture in coastal waters. Nutrient waste is a by-product of most farming activity and of animal growth. Its massive production is almost unavoidable in intensive fish-farming, which involves the less-than-perfect conversion of many tonnes of feed, rich in nitrogen and phosphorus compounds, into marketable fish. Thus, eutrophication is a potential risk to water bodies in which fish are farmed. Moreover, the input of particulate organic matter (POM) by farmed fish, and its associated biological (or biochemical) oxygen demand

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(BOD) can have effects that are similar to those of eutrophication.

Despite such negative effects, human needs for protein have driven the growth of aquaculture in recent decades (FAO, 2007). In order to maintain sustainable human use of water bodies used for fish-farming, managers of aquatic environments need to be able to diagnose eutrophication and to manage waste loads. Efficient load management requires knowledge of the relationship between indicators of 'pressure' and indicators of 'state' or 'impact', where the terms 'pressure', 'state' and 'impact' are taken from the DPSIR framework of the European Environment Agency (Luiten, 1999). 'Pressure' indicators include the loadings of water bodies with nutrients or POM. As reviewed by Tett et al. (2007), 'state' indicators (such as chlorophyll concentration) can point to '. . . accelerated growth of algae . . .' and those such as AMBI (Borja et al., 2000); or PCI (Tett et al., 2008) can point to '. . . undesirable disturbance to the balance of organisms . . .' in the plankton or benthos. 'Impact' indicators include the frequency and magnitude of fish kills due to algal blooms or deoxygenation, and the frequency and extent of closures of shell-fisheries due to the bivalves' intoxication with algal toxins.

Models for eutrophication can be used to explore the connections between pressure, state and impact in an ecosystem. They can provide cost-effective estimates of rate processes and fluxes. For instance, only "new production" (Eppley, 1992), supported by external inputs of nutrients, has the potential to cause "undesirable disturbance", but is easier to calculate from a model than to measure. Where eutrophic conditions are recognized by breach of legally defined Ecological Quality Objectives (EcoQOs) relating to 'state' or 'impact' indicators, models can help diagnose past or present eutrophication in cases where monitoring data are scarce. Finally, models can be used to restrict or manage aquaculture in relation to the capacity of ecosystems to assimilate farm waste. *Assimilative capacity* can be defined as the amount of waste that a water body can accept without breaching EcoQOs or other relevant water quality standards. In the context of eutrophication, there is, therefore, a need for models that can make cost-efficient predictions of likely changes in 'state' or 'impact' variables as a function of waste loading in a water body of defined size, shape and physical conditions.

1.2. *Progress in modelling eutrophication in RREs*

Many models have been made to describe eutrophication. Here we focus on matters concerned with (i) simplicity versus complexity in models, and (ii) models applicable to the Comprehensive Studies Task Team (CSTT)'s zone B scale in Regions of Restricted Exchange especially, fjords. This scale B corresponds to the scale of a body of water with a residence time of a few days, which is sufficiently long for waste nutrients to be converted into phytoplankton (CSTT, 1994, 1997).

Some very complex, coupled physical–biological, models of marine ecosystems have been developed during recent decades, but it is expensive and time-consuming to apply these to a small water body such as a typical fjordic RRE. Therefore there is a need for relevant models that are simple, but no simpler than is necessary. Stabilising the simplicity principle for modelling eutrophication, we agglutinate previous works in three modules: a pelagic, physical and an external waste input module. The novelty of the model presented here lies not only on the imbrication of previous model developments but on its eventual modification or improvement to modify a general porpoise model for simulating macro-nutrients and microplankton dynamics into a model to assess the capacity of ecosystems to assimilate external input waste, that we have exemplified by fish-farm activity.

For the external waste input module we have used the model developments started with the Equilibrium Concentration Enhancement (ECE) model applied in Gillibrand and Turrell (1997). The ECE model concept is that of a single box that receives input from the surrounding land and from local human activities, and which exchanges with the adjacent sea that provides the main boundary conditions.

The ECE model used in Gillibrand and Turrell (1997) does not include any biological module and only identify Scottish RREs at greatest risk from added nutrients. Therefore, the pelagic module considered here starts off with the modelling strategy employed in the CSTT (CSTT, 1994, 1997) and that has been proof a good screening method for eutrophication (Tett et al., 2003a). The CSTT mode is built on top of the ECE outcome, modelling the worst case, in which all available 'Dissolved Available Inorganic Nitrogen' (DAIN) is converted to phytoplankton, at a yield q of chlorophyll from nutrient (Gowen et al., 1992). These models might be considered too simple in their steady-state solutions and their lack of detail in the pelagic ecosystem or nutrient recycling. As a result, we used and improved the dynamic version of the CSTT model made by Laurent et al. (2006) where they were able to simulate seasonal changes in concentrations of DAIN, Dissolved Inorganic Phosphorus (DIP) and phytoplankton chlorophyll. The model represented a fjord by three boxes, one of which relaxed to a reference condition. Nutrient inputs to simulations were provided by time-varying river discharge multiplied by a fixed freshwater nutrient content and by salmon-farm waste values calculated from monthly totals of feed. However, in Laurent et al. (2006) there was no simulation of the "balance of organisms" within the phytoplankton, and so the inclusion of a new limiting nutrient (dissolved silicate) and the division of the phytoplankton chlorophyll was made guided by the PROWQM model (Lee et al., 2002; Tett and Lee, 2005). Moreover, Laurent et al. (2006) it used a constant value approach for the exchange rate with the boundary conditions as in Ross et al. (1993b, a, 1994), that we will improve.

The PROWQM (Lee et al., 2002; Tett and Lee, 2005) was developed to describe seasonal changes in several life-forms within the phytoplankton and associated pelagic bacteria and protozoa. The PROWQM included a "two microplankton" biological model. The microplankton includes pelagic micro-algae and photosynthetic bacteria (i.e., phytoplankton) and microheterotrophs (pelagic protozoa plus heterotrophic and chemosynthetic bacteria) (Tett, 1987). Using a microplankton box in a model assumes a short-term equilibrium between the growth rates of pelagic micro-algae and cyanobacteria, on the one hand, and, on the other hand, the pelagic heterotrophic bacteria and protozoa that immediately consume the products of photosynthesis. As an example, the value of the microplankton compensation irradiance exceeds that of the phytoplankton compensation irradiance, because additional light is needed for photosythetic production to balance consumption by microheterotrophs as well as phytoplanktoners themselves (Tett et al., 2002). However, the microplankton algorithms used in these models, drew on "Cell-quota" theory (Droop, 1983; Tett et al., 2003b) and allowed for growth-controlling variations in algal content of N and Si relative to biomass and chlorophyll. Instead we used the parameterization based on that of PROWQM (Lee et al., 2002), but simplified to 'Monod' growth, with constant yields of chlorophyll from each nutrient linking the equations for microplankton dynamics to those for nutrient dynamics. To the two dissolved inorganic, potentially limiting, nutrients (N and P) used in the dCSTT model, the LESV model adds dissolve silica as a state variable, to allow changes in N:Si or P:Si ratio to shift the balance of organisms.

Finally the physical module was also built on previous modelling developments. The exchange rates for single-box models of RREs, in rainy climates or with significant astronomical tides, can be calculated from tidal prisms or from freshwater budgets (Landless and Edwards, 1976; Tett, 1986). However, such simple methods do not take full account of the many physical processes that drive water movements within RREs and between them and