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Propagation of uncertainty in ecological models of reservoirs: From physical to population dynamic predictions

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ABSTRACT

Ecological models are widely accepted in the scientific community as tools to describe, interpret and predict ecosystem functioning. However, to be used in environmental management, model uncertainties, their magnitude and sources need to be carefully assessed. A one-dimensional coupled physical–ecological model is applied to a deep Mediterranean reservoir (Lake Béznar) to determine whether or not the uncertainty existing in physical predictions affects ecological predictions, and, then to quantify this uncertainty. The sources of uncertainty include light penetration in the water column, inflow mixing and geometry, and boundary conditions at free surface. Uncertainty in the model results was evaluated following the procedures outlined in Beven (2001), based on Montecarlo simulations. At least during summer time, the largest sources of uncertainty in the physical predictions are associated to the input variables used to construct the surface (heat and momentum) boundary conditions. Uncertainties in the physical model propagate to the ecological results. Average chlorophyll-a concentration predicted by the ecological module in the water column, their standard deviations, and the timings of the successional changes in the algal community all vary depending on the magnitude of the error accepted in the physical predictions. Our results illustrate that the analysis and quantification of model uncertainty are fundamental to properly express model results and, consequently, to optimize monitoring programmes and guide management decisions.

1. Introduction

During the last four decades engineers and ecologists have devoted considerable efforts to develop mathematical models capable of simulating changes in abundance (Di Toro et al., 1975; Kuo and Thomann, 1983; Cole and Buchak, 1995; Gurkan et al., 2006; Ahlgren et al., 1988; among others) and composition (Hamilton and Schladow, 1997; Elliott et al., 1999; Omlin et al., 2001; Markensten and Pierson, 2007; among others) experienced by phytoplankton communities in lakes and reservoirs. This interest lies on the fact that these changes in the algal communities (succession) can severely affect the quality of the water and even compromise the effectiveness of treatment processes undertaken in downstream water treatment plants. For example, the occurrence of blue-green algal blooms in water supply reservoirs may lead to severe clogging problems during the filtering operations; or it may lead to taste, odour and even health problems as a consequence of several species and stocks of blue green-algae producing toxic substances (Guven and Howard, 2006; Margalef, 1983). Successional changes can be observed at species level, functional structure or size/biomass distribution (e.g. Lindenschmidt and Chorus, 1998; Reynolds et al., 2002; Padisák et al., 2009), and are largely associated to changes in the physical environment determining light climate and nutrient availability for algal growth (Margalef, 1997; Reynolds, 1997). The light environment experienced by phytoplankton cells, for example, are related to turbulent mixing, which determines the residence time of microalgae within the euphotic layer, with optimal light levels (MacIntyre and Romero, 2000). The distribution and bioavailability of nutrients in the euphotic layer are also the result of transport processes interacting with biological phenomena. Consequently, the knowledge and predictability of the composition of phytoplankton communities and its evolution need to be grounded on the knowledge of the physical environment in the water column. Consistent with this widely accepted perception of succession in aquatic ecosystems, most mathematical models predicting the evolution of phytoplankton communities are based on the appropriate description of the relationship between the physical environment (in particular, thermal stratification and mixing energy) and algal growth. Some of the proposed ecological models proposed require that the physical environment is provided as an external input (e.g. PROTECH: Reynolds, 1997; Reynolds et al., 2001; e.g. CE-QUAL-ICM:

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Cerco and Cole, 1995). Others, in turn, are grounded on predictions provided by a physical multi-layered model of transport and mixing in the water column, which is, algorithmically, coupled to the ecological model (e.g. DYRESM-CAEDYM: Hamilton and Schladow, 1997; MIKE3: Integrated Center for Computational Hydrodynamics, 2002; see also Rigosi et al., 2010). In either case, and in general, the values of the physical or ecological variables predicted by ecological models are uncertain. Uncertainty in model predictions is partly associated to the physical models, their structure (structural uncertainty), the unknown value of their parameters (knowledge uncertainty) and the intrinsic spatial and temporal variability of the hydrological and/or meteorological phenomena forcing transport and mixing in aquatic systems (stochastic uncertainty), which is difficult, or nearly impossible, to characterize in detail (Hession et al., 1996; Oberkampf et al., 2002). On the other hand, the uncertainty is associated to the structure of the ecological module, to the unknown value of the large number of kinetic parameters that are typically used to parameterized biogeochemical processes, and to the variability, itself, of the different biochemical processes and their interactions. Frequently, though, in the application of ecological models in aquatic systems, the physical description is assumed to be correct and the uncertainty in the predictions of the ecological routines to be largely determined by the ecological modelling procedures (e.g. Arhonditsis and Brett, 2005; Gal et al., 2009; Elliott et al., 2006). Beck and Halfon (1991), for example, examined model uncertainties but only included parameters related with nutrient uptake and primary production. Few ecological modelling studies have included physical parameters in their uncertainty analysis. The work of Van Straten and Keesman (1991) is one of those studies, in which the parameters of a sediment sub-model and parameters governing algal growth were included in the uncertainty analysis to obtain speculative predictions and uncertainty bounds of chlorophyll-a and phosphorus concentrations. In their calibration exercise, Arhonditsis et al. (2007) also included mostly ecological parameters, but only few physical parameters (e.g. cross-thermocline exchange rate). To the extent of our knowledge, few studies have been published in the literature in which the propagation of uncertainty from physical to ecological predictions is explicitly taken into account. Nor there have been studies comparing the uncertainty associated to physical modelling to that arising from the ecological routines.

The general objective of this work is to analyse the uncertainty, its sources and magnitude, in ecological modelling, with emphasis on models where a physical module simulates the physical structure of the water column which is then used to simulate succession in the ecological modules. In particular, our goal is to assess the level of uncertainty existing in the predictions of abundance and composition of phytoplankton communities arising as a result of uncertainties in the physical model. The uncertainty analysis is conducted in this study following the procedure described in Beven (2001), and is based on Montecarlo simulations. Out of the wide range of mathematical models available for simulating transport in lakes and reservoirs our focus is on one-dimensional models, providing descriptions of the vertical structure of the ecosystem (i.e. ignoring spatial variations in the horizontal plane). The selection of one-dimensional models is justified because this type of models (1D) are routinely used for the management of reservoirs and other engineered impoundments such as cooling ponds. In this work, a well-established and process based coupled one-dimensional physical–ecological model (DYRESM-CAEDYM, Imberger and Patterson, 1990; Schladow and Hamilton, 1997; Herzfeld and Hamilton, 2003), widely accepted as a valid water quality management tool, has been adopted and applied to a deep warm monomictic reservoir (Lake Béznar) located in Southern Spain. The choice of Lake Béznar as the case study in this work is based on the fact that, from its general hydro-morphological characteristics, it can be considered a prototypical example of Mediterranean reservoirs, occupying deep, narrow and steep-slope valleys.

2. Methods

2.1. Study site

Lake Béznar is a mesotrophic reservoir located in Southern Spain (37° latitude), draining a watershed that occupies the southwestern portion of Sierra Nevada. The watershed has a surface area of approximately 352 km^2 and shows marked changes in elevation. The average inflow Q that enters the reservoir is 1.79 $\mathrm{m}^3\,\mathrm{s}^{-1}$ (56.5 hm^3 annual volume) with large oscillations on seasonal scales. Maximum inflow rates occur during winter and spring. Minimum inflow rates occur during late summer and early fall. The maximum volume of water held in the reservoir V is 54.60 hm^3, hence, the nominal residence time of Lake Béznar, estimated as V/Q is aproximately one year (352 days) which is close to average residence time of all the others reservoirs existing in the Andalusian Mediterranean Basin (411 days). Average annual inflow, precipitation and maximum volume are also close to the Andalusian average values, thus, it can be considered a prototype reservoir (Guerrero Luque, 2006). When full, the surface area of the reservoir is ca. 170.26 ha, the elevation of the free surface is 485 m.a.s.l. and its maximum depth 102.96 m at the dam. The dam is equiped with two outlets located at 410 and 450 m.a.s.l. The bathymetry of the reservoir is shown in Fig. 1. The reservoir has an elongated shape oriented along the NW–SE direction. The largest inflows occur at the NW-end through Izbor river. The valley is wide open in the tail of the reservoir, but with lateral slopes of up to 50% in the dam area. Béznar is a warm (never reaches temperatures below 4 °C) and monomictic reservoir that typically stratifies during the summer months and then mixes during the winter.

2.2. Model setup

A coupled physical–ecological model of Lake Béznar was developed using the Dynamic Reservoir Simulation Model (DYRESM) (Imberger and Patterson, 1981; Schladow and Hamilton, 1997) and the Computational Aquatic Ecosystem Dynamic Model (CAEDYM) (Herzfeld and Hamilton, 2003). DYRESM is a process-based one-dimensional (1D) hydrodynamic model which includes descriptions of mixing and transport processes associated with river inflow, natural or manmade outflows, vertical diffusion in the hypolimnion and mixed-layer dynamics, and it is used to predict the variation of water temperature and salinity with depth and time. The lake (Fig. 2) is represented as a set of Lagrangian layers which are free to move vertically (advection), and to contract and expand in response to inflows, outflows and surface mass fluxes (precipitation and evaporation). The model has been extensively described and applied with success in the literature to simulate the vertical thermal and salinity structure for a wide range of lakes such as, Lake Burragorang – Australia (Romero et al., 2004), Lake Constance – Europe's Alps (Hornung, 2002), San Roque reservoir – Argentina (Antenucci et al., 2003) or Lake Kinneret – Israel (Antenucci et al., 2000), among many others.

DYRESM provides predictions of the physical environment which are, then, used to drive the biogeochemical simulations in CAEDYM. The coupled model DYRESM-CAEDYM results as a powerful tool to investigate interaction between physics, chemistry and biology in aquatic ecosystems (Herzfeld and Hamilton, 2003). The water body exchanges energy and materials with the environment through the open boundaries, either the free surface or the inflow/outflow sections (Fig. 2). Nutrients, for example, enter to the