Available online at www.sciencedirect.com
SCIENCE DIRECT®

Estuarine, Coastal and Shelf Science xx (2006) 1–12

ESTUARINE COASTAL AND SHELF SCIENCE
www.elsevier.com/locate/ecss

An ecohydrology model of the Guadiana Estuary (South Portugal)

Eric Wolanski [a,*], Luís Chicharo [b], M. Alexandra Chicharo [b], Pedro Morais [b]

[a] Australian Institute of Marine Science, PMB No. 3, Townsville MC, Queensland 4810, Australia
[b] Centro de Ciências do Mar, Faculdade de Ciências do Mar e do Ambiente, Universidade do Algarve, Campus de Gambelas, 8005-139 Faro, Portugal

Received 30 January 2005; accepted 20 March 2006

Abstract

A 1-D ecohydrology model is proposed that integrates physical, chemical and biological processes in the Guadiana Estuary during low flow conditions and that predicts the ecosystem health as determined by the following variables: river discharge, nutrients, suspended particulate matter, phytoplankton, zooplankton, bivalves, zooplanktivorous fish and carnivorous/omnivorous fish. Low flow conditions prevail now that the Alqueva dam has been constructed. The ecological sub-model is based on the non-linear Lotka–Volterra equation. The model is successful in capturing the observations of along-river changes in these variables. It suggests that both bottom-up and top-down ecological processes control the Guadiana Estuary ecosystem health. A number of sensitivity tests show that the model is robust and can be used to predict – within likely error bounds provided by the sensitivity tests – the consequences on the estuary ecosystem health of human activities throughout the river catchment, such as the irrigation farming downstream of the Alqueva dam, reclamation of the salt marshes by urban developments, and flow regulation by the Alqueva dam. The model suggests that the estuarine ecosystem health requires transient river floods and is compromised by flow regulation by the Alqueva dam. Remedial measures are thus necessary.

Keywords: ecohydrology; marine ecology; flushing; modelling; dam; flow regulation; Portugal; Alqueva dam; Guadiana Estuary

1. Introduction

1.1. The need for an ecohydrology estuarine model

Throughout human history, the coastal plains and Lowland River valleys have usually been the most populated areas over the world (Wolanski et al., 2004). At present, about 60% of the world's population lives along the estuaries and the coast (Lindeboom, 2002). This is degrading estuarine and coastal waters through pollution, eutrophication, increased turbidity, overfishing, and habitat destruction. The pollutant supply does not just include nutrients; it also includes mud from eroded soil, heavy metals, radionuclides, hydrocarbons, and a number of chemicals including new synthetic products.

The impact on estuaries is commonly still ignored when dams and irrigation farming are proposed on rivers. In addition, estuaries are often regarded as sites for future development and expansion, and have been increasingly canalized and dyked for flood protection, and their wetlands infilled for residential areas.

All these factors impact on the biodiversity and productivity and, hence, the overall health of estuaries and the ecosystem services they provide to humans (Nixon, 2003; Erzini, 2005). They increasingly lead humans away from the possibility of ecologically sustainable development of the coastal zone. Integrated coastal zone management plans are drawn up worldwide (e.g., Haward, 1996; Billé and Mermet, 2002; Tagliani et al., 2003; Pickaver et al., 2004; Lau, 2005). However, in the presence of significant river input, most are bound to fail because they commonly deal only with local, coastal issues, and do not consider the whole river catchment as the fundamental planning unit. It is as if the land, the river, the estuary, and the sea were not part of the same system. When

* Corresponding author.
E-mail address: e.wolanski@aims.gov.au (E. Wolanski).

0272-7714/$ - see front matter © 2006 Elsevier Ltd. All rights reserved.
doi:10.1016/j.ecss.2006.05.029

dealing with estuaries and coastal waters, in most countries land-use managers, water-resources managers, and coastal and fisheries managers do not cooperate effectively due to administrative, economic and political constraints, and the absence of a forum where their ideas and approaches are shared and discussed (Wolanski et al., 2004). To help alleviate this problem, UNESCO – IHP has launched the ecohydrology program. In this program, the concept of ecohydrology is introduced as a holistic approach to the management of rivers, estuaries and coastal zones within entire river catchments, by adopting science-based solutions to management issues that restore or enhance natural processes as well as the use of technological solutions (Zalewski, 2002).

This science-based management requires the use of a holistic model to quantify the human impact on the ecosystem health of estuaries and to enable the exchange of information between oceanographers, biologists, ecologists, engineers, sociologists, economists and water-resources managers at local and national governmental levels, and the community.

1.2. The science behind the model

The model is process-based. The dominant physical, chemical, biological and human-related processes in an estuary are assumed to follow those described by Wolanski et al. (2004) and are sketched in Fig. 1. These processes are briefly summarised below.

The ecological health of estuaries is determined by the interaction between organisms and variations in salinity, currents, waves, suspended particulate matter (SPM), bed sediments, temperature, air exposure, hypoxia, wetland contaminants and biodiversity. Like the health of a living organism, the health of an estuary or a coastal water body, cannot be measured by one single variable, indeed a number of variables are important (Balls, 1994). Well-flushed estuaries are intrinsically more robust than poorly flushed systems. As a result, environmental degradation is most often apparent during periods of reduced freshwater inflows, e.g. during drought or when human activities reduce the freshwater flow. Therefore, this ecohydrology model focuses on low flow conditions when vertically well-mixed conditions often prevail.

Once riverine-derived suspended particulate matter enters the estuary, it can be trapped within an estuarine turbidity maximum (ETM) zone (Fig. 1). The ETM is commonly located in the very low salinity reaches of an estuary. The maximum, depth-averaged, suspended solid concentration (SSC) at high water within an estuary can be predicted semi-empirically as a function of the tidal intrusion and the tidal range (Uncles et al., 2002).

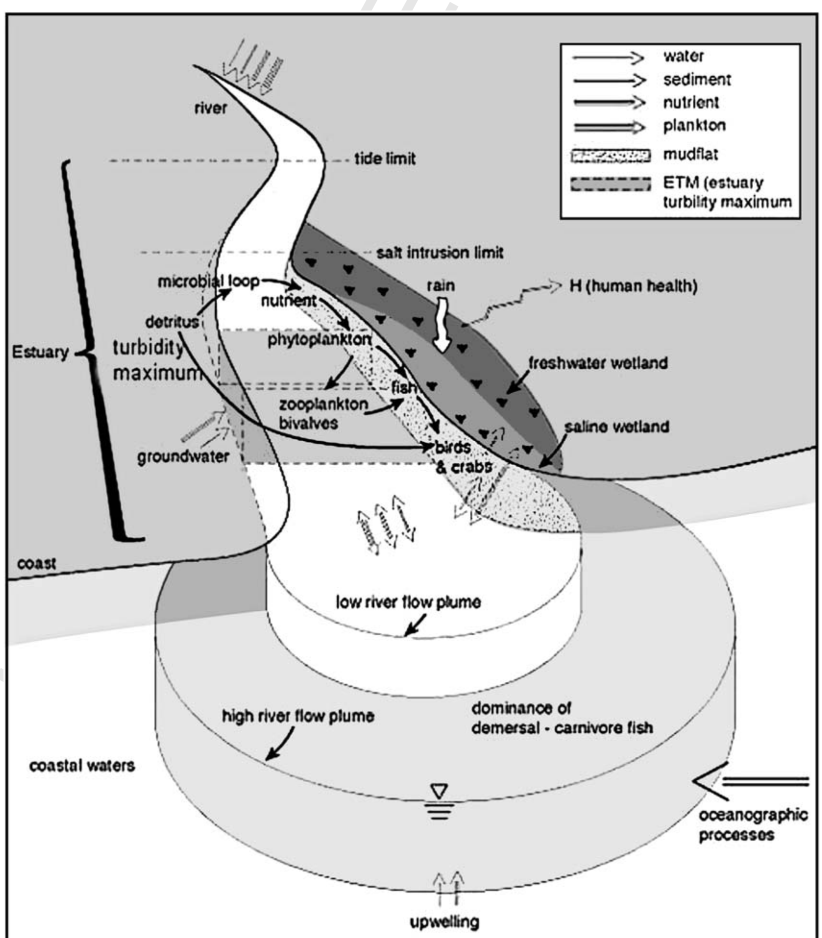

Fig. 1. Sketch of the dominant processes operating in an estuary. Adapted from Wolanski et al. (2004).

Sediment particles and aggregates within the ETM can give rise to marked changes in water quality. Fine particles can adsorb metal ions and organic macro-molecules from solution to such an extent that some metals can be completely removed from solution within a strong ETM (Salomons and Forstner, 1984; Ackroyd et al., 1986). Once nutrients enter an estuary, non-conservative behaviour can be pronounced. Key processes responsible for this non-conservative behaviour include burial in sediment reservoirs and desorption processes particularly if the sediment is nutrient-rich. Nutrients are generally mainly in particulate form (i.e. absorbed to the mud particles in suspension) in freshwater and can be released in solution in saline water.

The salt marshes of Western Europe generally produce more than 1 kg m^{-2} yr^{-1} of above-ground dry matter (Boorman et al., 1994a,b; Lefeuvre, 1996). Salt marshes export some of this organic matter. Salt marshes and their tidal creeks are also an important nursery ground, and a refuge, for larvae and post-larvae of bivalve, carnivorous/omnivorous fish and zooplankton.

The estuary is modelled as a converter of living phytoplankton to detrital particles; it is also a conveyor of detrital matter to the sea. Fishes help transfer energy and matter from estuarine plants to upper trophic levels. The great bulk of the organic matter produced (sometimes 90%) is processed through the detrital system. Zooplankton, planktivorous fish, interstitial micro and meiofauna, surface deposit-feeding molluscs, fishes and polychaeta, and filter-feeding invertebrates consume a much greater proportion of the primary production of the phytoplankton and benthic microalgae. Annual plant growth and decay provide continuing large quantities of organic detritus. In addition, there is often a considerable input of detritus from river inflow. Detrital particles and their associated microorganisms provide the basic food source for primary consumers such as zooplankton, most benthic invertebrates and some fishes. The first trophic level in the estuarine ecosystem is therefore best described as a mixed trophic level of detritus consumers, which in varying degrees are herbivores, omnivores or primary carnivores (Knox, 1986).

1.3. Study area

The Guadiana River is one of the largest in the south of the Iberian Peninsula, crossing extensive rural areas and includes the Iberian Pyritic Belt (Gonzalez, 1995).

The fluvial regime is characterised by low flows during summer and episodic runoff periods in winter with the resulting discharge of sediments into the estuary and coastal zone. The estuary is 60 km long, it has a maximum width of 550 m and the maximum depth varies between 5 and 17 m. The tidal regime of the estuary is meso-tidal, with an average amplitude of 2 m (Michel, 1980).

The estuary has an important nursery function for several fish species, such as the anchovy *Engraulis encrasicolus* sensu lato and several Sparidae, and crustacean species such as the brown shrimp *Crangon crangon*. Moreover, the outwelling from the estuary to the coastal area promotes the development of the food web and influences the fisheries (Chícharo et al., 2002; Erzini, 2005).

Several pollution sources exist in the Guadiana Estuary area, mainly resulting from urbanisation, agriculture (fertilizers, pesticides, and herbicides), cattle breeding and olive oil production. The freshwater flow reaching the estuary is at present regulated by more than 100 dams, including the Alqueva dam whose construction was completed in 2002 and that forms the largest reservoir in Europe (Alveirinho et al., 2004).

1.4. Aims

This study aimed to develop an ecohydrology model to be applied to the low flow conditions in the Guadiana Estuary. It describes such a model designed specifically for vertically well-mixed estuaries. The ecological sub-model is also simple, though still realistic. It incorporates the seven state variables: nutrients, suspended particulate matter, phytoplankton, zooplankton, bivalves, zooplanktivorous fish and carnivorous/omnivorous fish in the estuary and it predicts the ecosystem health.

2. Material and methods

2.1. Field data

Estuarine physical, chemical and biological data were obtained from the papers of M. Chicharo et al. and P. Morais et al. in this issue and from Pinto (2000), and Esteves et al. (2000). Data from river inflow were obtained online from Water National Institute (INAG), National System of Hydrological Resources (http://snirh.inag.pt/) from the hydrometric station Pulo do Lobo (lat. 37°48′ N, long. 7°38′ W), located a few kilometres above the last point of tidal influence (Mértola).

2.2. The estuarine ecohydrology model

The prototype is the Guadiana Estuary at low flow conditions − because such low flow conditions prevail now that the Alqueva dam exists. For a freshwater flow $Q_f < 50\ m^3\ s^{-1}$, the Guadiana Estuary is vertically fairly well-mixed in salinity (Fortunato et al., 2002). In a vertically well-mixed estuary, the distribution of salinity S is determined from the solution of the 1-D advection−diffusion equation (Fischer et al., 1979):

$$\partial(SA)/\partial t + \partial(QS)/\partial x = \partial(EA\,\partial S/\partial x)/\partial x \qquad (1)$$

where t is the time, Q is the flow rate (driven by tides and river flows), E is the longitudinal eddy diffusion coefficient, and A is the cross-sectional area. Eq. (1) is solved for a series of cells of volume V distributed along the length of the estuary from the tidal limit to the mouth. The time step dt is set to 1 day, thereby averaging over the tides. The open boundaries are located at the tidal limit and at the mouth. At the

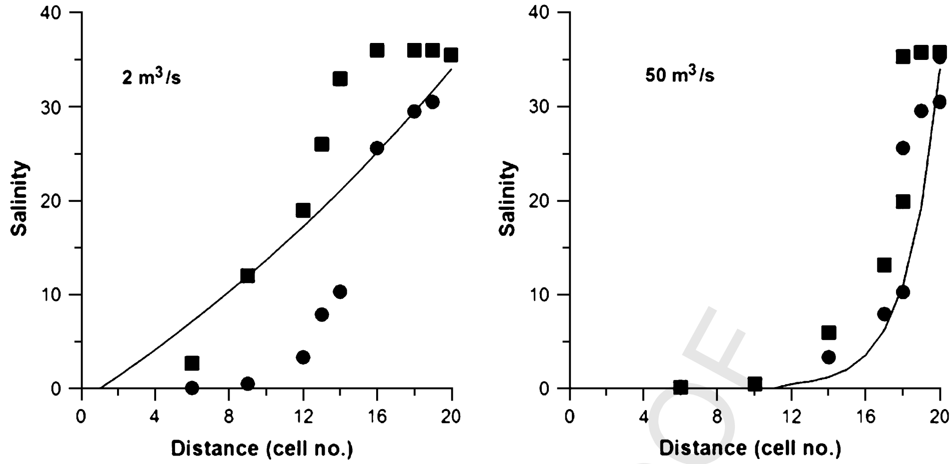

Fig. 2. Observed and predicted along-channel distributions of salinity in the Guadiana Estuary around the times of (square) high tide and (circle) low tide for salinity of (a) 2 $m^3 s^{-1}$ and (b) 50 $m^3 s^{-1}$. Cell # 1 is located at the tidal limit, 60 km upstream of cell # 20 that is located at the mouth.

tidal limit, the model assumes that the salinity $S = 0$ and it also assumes that Q_f is known. At the mouth, the salinity is assumed to be 35. Turbulent diffusion is due to tides, wind, and freshwater runoffs and is parameterised by the parameter *E*. In the model, this is determined by mixing coefficients that quantify the fraction of water in a cell that is exchanged with adjoining cells during the time step (1 day). This parameter is varied until the solution fits well with the observations. This is shown in Fig. 2 for the case of the Guadiana Estuary for two values of the freshwater discharge Q_f (2 and 5 $m^3 s^{-1}$).

The model enables one to readily calculate the flushing time of the estuary. To do that, in the model the freshwater discharge is set to be a constant and the estuary is initially filled with uniform seawater salinity at $t = 0$. The system is then allowed to evolve, and in the model salt is progressively expelled from the upper reaches of the estuary until a steady state solution is reached. This is shown in Fig. 3 for a freshwater discharge (Q_f) of, respectively, 2 and 50 $m^3 s^{-1}$. It is apparent that for $Q_f = 50\ m^3 s^{-1}$ the residence time is about 5 days, and that for $Q_f = 2\ m^3 s^{-1}$ the residence time varies between 14 days in the lower reaches and 37 days in the upper reaches of the estuary.

For a non-conservative constituent such as nutrients, plankton, detritus, fish, and bivalve, Eq. (1) is modified by including a sink–source term ΔC (Thomann, 1980), where C is the concentration:

$$\partial(CA)/\partial t + \partial(QC)/\partial x = \partial(EA\, \partial C/\partial x)/\partial x + \Delta C \quad (2)$$

where ΔC is derived from the ecological sub-model described below.

The ecological sub-model is based on the non-linear Lotka–Volterra equation. It is based on a finite-element model with the same cells as those used in the salinity model. A number of modeling equations are possible. In the absence of excretion and death not due to predation, the predator–prey relationship is often calculated by the non-linear equations (Brauer and Castillo-Chavez, 2001; Kot, 2001).

$$\partial X/\partial t = \beta X(1 - X/X_o)H(Y, Y_{o1}) \quad (3)$$

and

$$\partial Y/\partial t = -\beta X(1 - X/X_o)H(Y, Y_{o1}) \quad (4)$$

where X is the predator biomass ($X = CV$) where C is the predator concentration, Y is the prey biomass, β is the predator growth rate, X_o is the predator saturation biomass, Y_{o1} is the prey starvation biomass, i.e. the biomass at which the predator is unable or unwilling to spend energy to find this prey. H is the Heavyside function, i.e. $H = 0$ if $Y < Y_{o1}$, and $H = 1$ if $Y > Y_{o1}$. Eq. (2) also applies if Y is a nutrient. Provided starvation does not occur, the solution is an S-shaped curve whereby X initially increases exponentially in time. The growth rate is

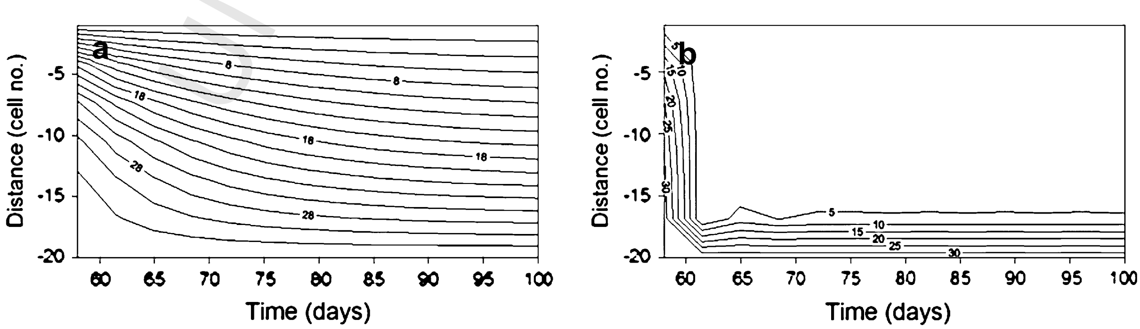

Fig. 3. Estimation of the residence time from the time series of salinity distribution in the estuary following intrusion of freshwater for (a) 2 $m^3 s^{-1}$ and (b) 50 $m^3 s^{-1}$.

zero at $X = X_o$. Because X and Y are related by Eqs. (3) and (4), Y decreases toward a minimum value.

In the model, freshwater phytoplankton and bacterioplankton in the river are subject to salt stress when freshwater mixes with saltwater; and the freshwater microbial populations die in this zone (Flameling and Kromkamp, 1994; Goosen et al., 1995). In the model, the salinity also limits the seaward distribution of saline water plankton, invertebrates (e.g. bivalves) and fishes.

In an estuary, changes in salinity constitute a major stress that can lead to death. There are other stressors, for instance, small values of the dissolved oxygen concentration. A death-excretion rate δ must then be added to Eq. (3) that becomes:

$$\partial X/\partial t = \beta X(1 - X/X_o)H(Y, Y_{o1}) - \delta X \quad (5)$$

The solution of this equation is also an S-shaped curve, the maximum value, however, is smaller than in the absence of this death-excretion rate, that is $X = X_o(1 - \delta/\beta)$. To remain realistic the solution requires $\beta > \delta$, i.e. that the growth rate is larger than the death-excretion rate.

In an estuary, fringing wetlands (mainly salt marshes and riparian ecotones, together with the tidal creeks that drain them) can be an important source of detritus and nutrients, as well as a nursery for juveniles and sub-adults as well as a refuge. This is particularly the case for bivalves. Mathematically, this is expressed by adding a source of X to the right-hand side of Eq. (5). The final equation becomes:

$$\partial X/\partial t = \beta X(1 - X/X_o)H(Y, Y_{o1}) - \delta X + \alpha \quad (6)$$

where α is the import rate from wetlands.

The ecosystem model represents mathematically through Eq. (5) the interactions summarised in Fig. 4 between nutrients concentration (N), suspended sediment concentration (SSC), phytoplankton concentration (P), zooplankton concentration (Z), bivalve concentration (B), detritus concentration (D), zooplanktivorous fish concentration (ZF), and carnivorous/omnivorous fish concentration (CF). All dying matter becomes detritus. Settling is not included in the model, because the animals (e.g. zooplankton) are mobile and can swim in the water. The model is equally complex at the lowest and highest trophic levels, which increases the model robustness (Jorgensen and Bendoricchio, 2001). Thus the ecosystem model equations are:

Nutrients (N; nitrate)

$$\partial N/\partial t = -\beta_{NP}P(1-P/P_o)H(N,N_{o1}) + \alpha_N + \gamma_{SSCN}\mathrm{SSC} \quad (7)$$

Phytoplankton (P)

$$\begin{aligned}\partial P/\partial t =\ & \beta_{NP}P(1-P/P_o)H(N,N_{o1}) - \beta_{PZ}Z(1-Z/Z_o)H(P,P_{o1}) \\ & - \beta_{PB}B(1-B/B_o)H(P,P_{o1}) \\ & - \beta_{PCF}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF}_o)H(P,P_{o1}) + \alpha_P - \delta_P P \quad (8)\end{aligned}$$

Zooplankton (Z)

$$\begin{aligned}\partial Z/\partial t =\ & \beta_{PZ}Z(1-Z/Z_o)H(P,P_{o1}) + \beta_{DZ}Z(1-Z/Z_o)H(D,D_{o1}) \\ & - \beta_{ZZF}\mathrm{ZF}(1-\mathrm{ZF}/\mathrm{ZF}_o)H(Z,Z_{o1}) \\ & - \beta_{ZCF}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF}_o)H(Z,Z_{o1}) + \alpha_Z - \delta_Z Z \quad (9)\end{aligned}$$

Bivalves (B)

$$\begin{aligned}\partial B/\partial t =\ & \beta_{PB}B(1-B/B_o)H(P,P_{o1}) + \beta_{DB}B(1-B/B_o)H(D,D_{o1}) \\ & - \beta_{BCF}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF}_o)H(B,B_{o1}) + \alpha_B - \delta_B B \quad (10)\end{aligned}$$

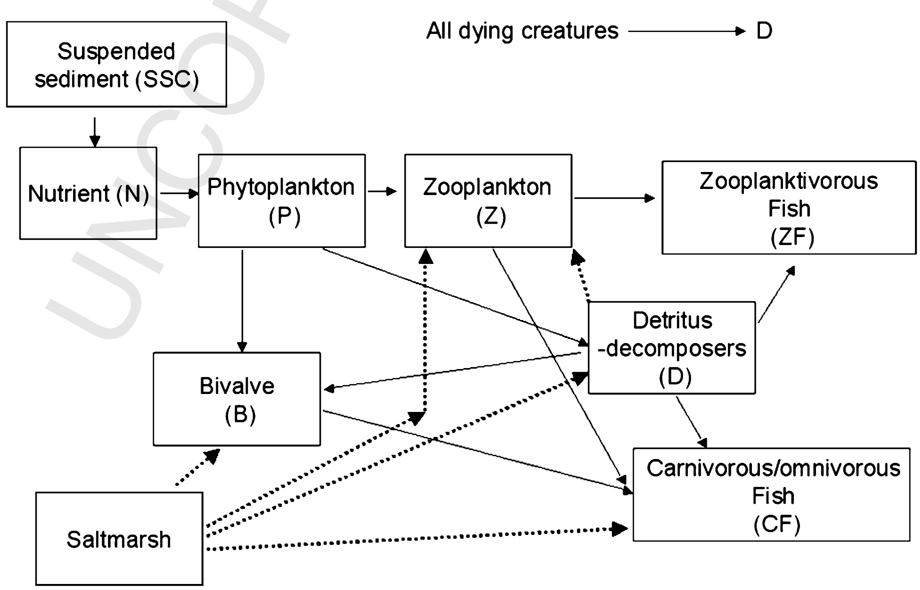

Fig. 4. Sketch of the estuarine food web in the ecohydrology model.

Carnivorous/omnivorous fish (CF)

$$\begin{aligned}\partial \mathrm{CF}/\partial t = {} & \beta_{\mathrm{BCF}}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF_o})H(B,B_{\mathrm{o1}}) \\ & + \beta_{\mathrm{PCF}}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF_o})H(P,P_{\mathrm{o1}}) \\ & + \beta_{\mathrm{ZCF}}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF_o})H(Z,Z_{\mathrm{o1}}) \\ & + \beta_{\mathrm{DCF}}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF_o})H(D,D_{\mathrm{o1}}) \\ & + \alpha_{\mathrm{CF}} - \delta_{\mathrm{CF}}\mathrm{CF} \end{aligned} \tag{11}$$

Zooplanktivorous fish (ZF)

$$\begin{aligned}\partial ZF/\partial t = {} & \beta_{\mathrm{ZZF}}\mathrm{ZF}(1-\mathrm{ZF}/\mathrm{ZF_o})H(Z,Z_{\mathrm{o1}}) \\ & + \beta_{\mathrm{DZF}}\mathrm{ZF}(1-\mathrm{ZF}/\mathrm{ZF_o})H(D,D_{\mathrm{o1}}) + \alpha_{\mathrm{ZF}} - \delta_{\mathrm{ZF}}\mathrm{ZF} \end{aligned} \tag{12}$$

Detritus (D)

$$\begin{aligned}\partial D/\partial t = {} & -\beta_{\mathrm{DB}}B(1-B/B_{\mathrm{o}})H(D,D_{\mathrm{o1}}) \\ & - \beta_{\mathrm{DZF}}\mathrm{ZF}(1-\mathrm{ZF}/\mathrm{ZF_o})H(D,D_{\mathrm{o1}}) \\ & - \beta_{\mathrm{DCF}}\mathrm{CF}(1-\mathrm{CF}/\mathrm{CF_o})H(D,D_{\mathrm{o1}}) \\ & - \beta_{\mathrm{DZ}}Z(1-Z/Z_{\mathrm{o}})H(D,D_{\mathrm{o1}}) + \alpha_{\mathrm{D}} + \alpha_{\mathrm{D}} + \delta_{\mathrm{B}}B + \delta_{\mathrm{P}}P \\ & + \delta_{\mathrm{Z}}Z + \delta_{\mathrm{CF}}\mathrm{CF} + \delta_{\mathrm{ZF}}\mathrm{ZF} \end{aligned} \tag{13}$$

In these equations the subscripts denote either constituent or the interaction between two constituents. For instance, δ_{ZF} is the death-excretion rate of ZF, and β_{DZF} is the growth rate of ZF from detritus, i.e. the rate of mass transfer rate from detritus to ZF.

In the nutrient equation, a new parameter was introduced, γ_{SSCN}, it denotes the leaching rate of nutrients from the particulate phase (i.e. absorbed on the fine sediment) to the dissolved phase.

In Eq. (2), because the model is run at a time step of 1 day, $Q = Q_{\mathrm{f}}$. There is thus no need to calculate the tidal dynamics; these are parameterised by the term E.

When applying Eq. (2) to the zooplanktivorous fish equation, Q is modified to incorporate the horizontal swimming by the fish as fish swim, by kinesis or taxis following environmental clues (Wolanski et al., 1997; Humston et al., 2000). This velocity is assumed to be proportional to S. Thus the fish in the model is able to swim, following taxis or kinesis, along environmental gradients.

3. Results and discussion

3.1. Application to the Guadiana Estuary

In the Guadiana Estuary, field data of fine sediment concentration during low flow conditions ($Q_{\mathrm{f}} < 50\ \mathrm{m^3\,s^{-1}}$) suggest the presence of a weak turbidity maximum zone near the salinity intrusion limit, with a maximum SSC value of 114 $\mathrm{mg\,l^{-1}}$ at $S = 12$ while SSC is about 30 $\mathrm{mg\,l^{-1}}$ in the freshwater reaches of the estuary (Portela, unpubl. data). Therefore, the model assumes a suspended sediment concentration (SSC) that is determined as follows:

$$\begin{aligned} &\mathrm{SSC} = 30 \quad \text{if } S < 1 \\ &\mathrm{SSC} = 30 + 7S \quad \text{if } 1 < S < 12 \\ &\mathrm{SSC} = 100 - 3.5(S - 15) \quad \text{if } S > 12 \end{aligned}$$

The model needs the knowledge of all the ecological parameters. The parameter α varies along-channel to correspond to the location of the salt marshes and riparian/terrestrial vegetation. The approximate values of the parameters are known from a number of studies and from comparison with other estuaries. The final values were selected as a result of a best-fit between observed and predicted values. The results of this calibration are shown in Fig. 5 for nutrients, zooplankton, bivalve, and fish, respectively. Table 1 lists the adopted values of the parameters.

While the calibration appears successful, it is important for the user to also judge whether the solution is realistic and stable (Hilborn and Mangel, 1997). This may be done by undertaking a sensitivity test to judge whether the model is unrealistically sensitive to a specific parameter, making it potentially unstable and unrealistic. A number of sensitivity runs were carried out, each one involving changing one parameter. Calculations were performed for $Q_{\mathrm{f}} = 2\ \mathrm{m^3\,s^{-1}}$, which is the environmental flow for the Guadiana River, i.e. the post-dam river discharge during the dry season. The list of sensitivity runs is summarised in Table 2.

The sensitivity tests show that phytoplankton (Chl *a*) is most sensitive in cases 2, 4 and 5, i.e. to β_{NP}, β_{PB} and δ_{P} (Fig. 6a).

These results suggest that bivalves play a more important role in filtering phytoplankton than zooplankton. This can result from the fact that bivalves are benthic and sessile organisms, being able to resist currents as opposite to zooplankton populations, although some develop strategies to resist displacement forces (Simenstad et al., 1994).

The most important parameter for zooplankton is δ_{Z} (the death-excretion rate of zooplankton), and to a lesser degree β_{NP} (uptake rate of nutrient by phytoplankton) and β_{PZ} (uptake rate of phytoplankton by zooplankton). The model zooplankton (Z) is most sensitive in case 7 and to a lesser degree in cases 2 and 3 (Fig. 6). As detritus can also be included in zooplankton diet, if $\beta_{\mathrm{DZ}} = 0.1\ \mathrm{day^{-1}}$. In fact, in a situation of low inflow – as the one tested in the sensitivity runs (Fig. 6c) – the expected decrease in detritus input caused by the reduction of inflow will affect the zooplankton biomass in the estuary, which highlights the importance of detritus as food source for estuarine zooplankton (Edwards, 2001; Kibirige et al., 2002).

The model also shows that zooplanktivorous fish (ZF) is most sensitive to β_{ZZF}, β_{DZF}, and δ_{ZF} (i.e. respectively, the uptake rate of zooplankton by zooplanktivorous fish, the uptake rate of detritus by zooplanktivorous fish, and the death rate of the zooplanktivorous fish; cases 6, 11 and 12) (Fig. 6d). In fact, salinity changes caused by modification of freshwater/seawater balance may affect zooplankton–prey distribution and impact zooplanktivorous fish species distribution. Moreover, it suggests that the export of detritus from the salt marsh does not seem to be the most important source of food for these fish.

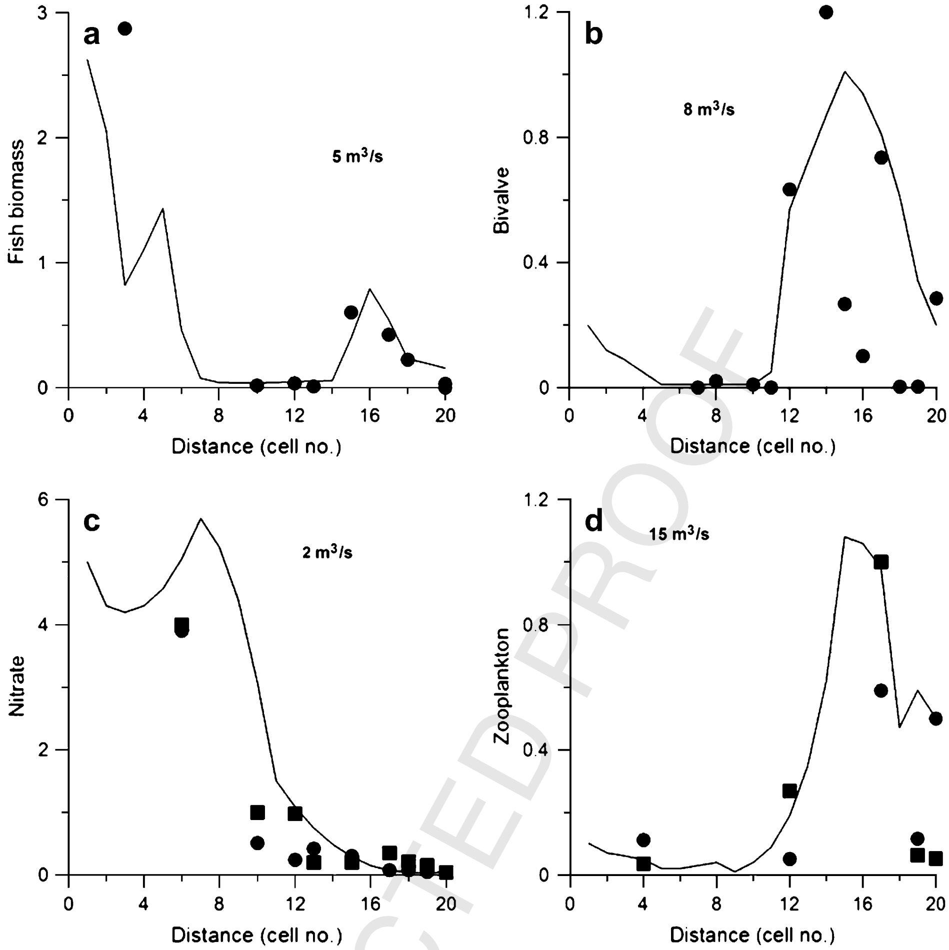

Fig. 5. Along-channel distribution in the Guadiana Estuary of (a) observed (dots) and predicted (line) total fish biomass, (b) observed (dots) and predicted (line) bivalve biomass, (c) observed (dots) and predicted (line) nutrients' (nitrate) mass, and (d) observed (dots) and predicted (line) zooplankton biomass. To convert biomass to concentration, for fish 2.8–2.87 g cm^{-2}, for bivalve 1.2–24 m^{-2}, for nitrate 4–15.5 μM, and for zooplankton 1–54 m^{-3}.

Table 1
Final values of the parameters. Rates are expressed as day^{-1}

β_{NP}	0.2
β_{PZ}	0.1
β_{PB}	0.1
δ_{P}	0.05
β_{ZZF}	0.1
δ_{Z}	0.1
β_{BCF}	0.1
δ_{B}	0.1
δ_{CF}	0.1
β_{DZF}	0.1
δ_{ZF}	0.1
β_{SSCN}	0.3
β_{ZCF}	0.03
β_{DCF}	0.03
β_{DB}	0.1
β_{DZ}	0.05
α_{B}	0.15 (=0 in freshwater reaches)
α_{D}	0.05 (=0 in freshwater reaches)
α_{CF}	0.05 (=0 in freshwater reaches)
α_{Z}	0.05 (=0 in freshwater reaches)

Table 2
Model sensitivity runs. All the runs were carried out for a steady, freshwater discharge $Q_f = 2$ m^3 s^{-1}. Rates are expressed as day^{-1}

Run 1:	Standard run
Run 2:	β_{NP} decreased from 0.1 to 0.05
Run 3:	β_{PZ} decreased from 0.1 to 0.05
Run 4:	β_{PB} decreased from 0.1 to 0.05
Run 5:	δ_{P} decreased from 0.05 to 0.025
Run 6:	β_{ZZF} decreased from 0.1 to 0.05
Run 7:	δ_{Z} decreased from 0.1 to 0.05
Run 8:	β_{BCF} decreased from 0.1 to 0.05
Run 9:	decreased from 0.1 to 0.05
Run 10:	δ_{CF} decreased from 0.1 to 0.05
Run 11:	β_{DZF} decreased from 0.1 to 0.05
Run 12:	δ_{ZF} decreased from 0.1 to 0.05
Run 13:	β_{SSCN} decreased from 0.3 to 0.15
Run 14:	β_{ZCF} decreased from 0.03 to 0.015
Run 15:	decreased from 0.03 to 0.015
Run 16:	β_{DB} decreased from 0.1 to 0.05
Run 17:	α_{Z} and α_{B} decreased from 0.15 to 0.075
Run 18	α_{D} in the freshwater zone increased from 0 to 0.05
Run 19:	β_{DZ} increased to 0.1 day^{-1} (run 19) and 0.05 day^{-1} (run 19a, open circles)
Run 20:	α_{D} increased to 0.15 in the saline region and 0.05 in the freshwater region
Run 21:	β_{PCF} increased to 0.1; α_{z} and α_{B} increased to 0.15

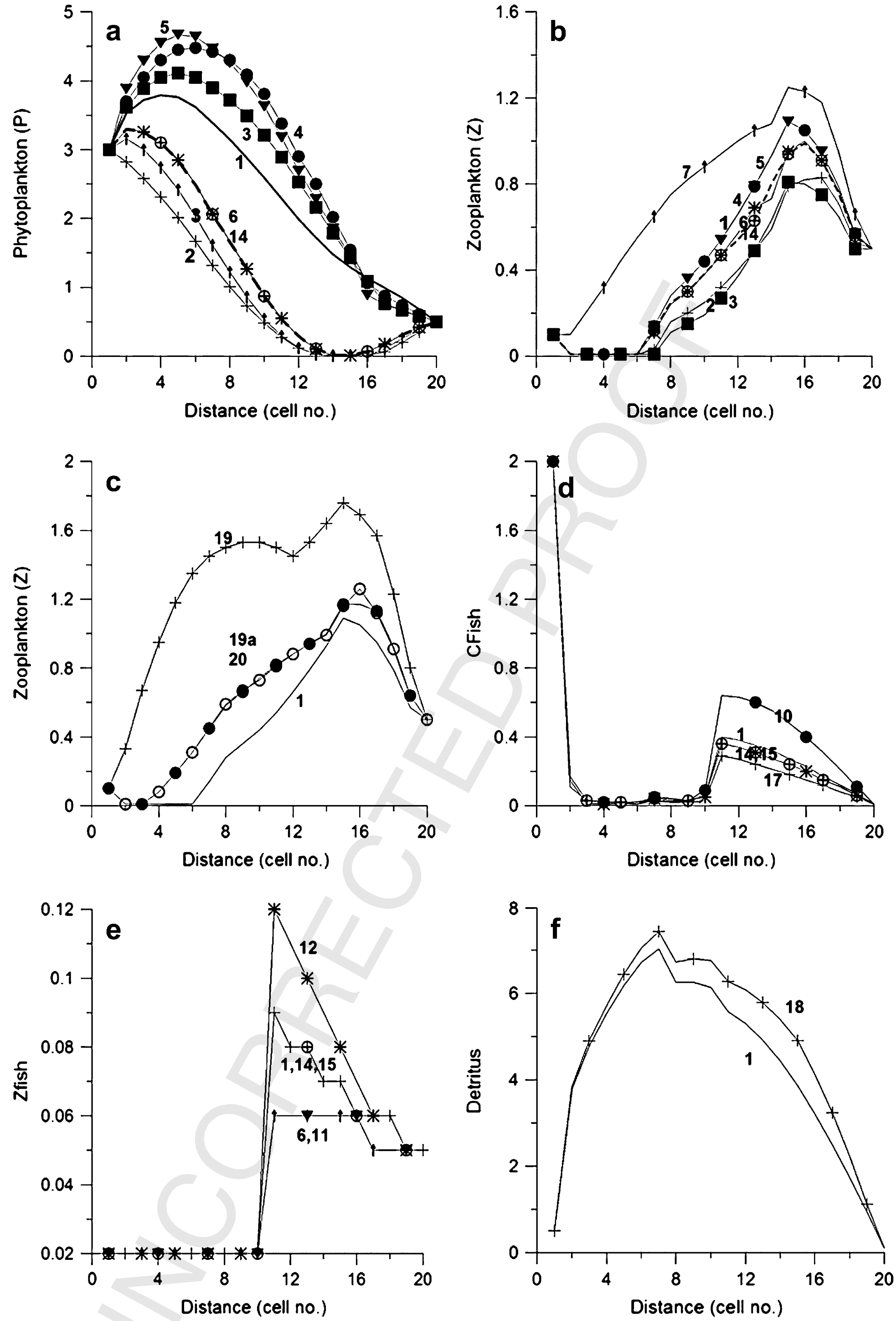

Fig. 6. Along-channel distribution of predicted variables in the Guadiana Estuary for various sensitivity runs shown as numbers (see Table 2). (a) Phytoplankton biomass (Chl *a*), (b) zooplankton biomass, (c) zooplankton (cont), (d) zooplanktivorous fish biomass, (e) carnivorous/omnivorous fish biomass, and (f) detritus biomass. To convert biomass to concentration, see Fig. 5 and for Chl *a* 3.5–7.8 $\mu g\,l^{-1}$.

In the model carnivorous/omnivorous fish is measurably sensitive only to δ_{CF} (the natural death rate of carnivorous/omnivorous fish; case 10, Fig. 6e). The model suggests that no other parameter than the natural death rate significantly influences the omnivorous fish. These fishes are mainly freshwater *Barbus* spp. These species are located mostly in the upper reaches of the estuary and the model suggests that this fish is highly vulnerable to a salinity increase, as a result of reduction in river inflow. The model also suggests that the lower estuary has more detritus than it can consume, thus the additional detritus from salt marshes is unimportant. In the upper areas, detritus mainly originates from the decomposition of riparian vegetation, this source of detritus seems more important in the middle and lower estuary (Fig. 6f).

The model sensitivity tests are useful because they show that:

1. the model appears robust because large, but reasonable, changes in the parameters do not lead to instabilities such as the destruction of trophic layers;
2. the biomass of organisms is directly affected by its consumption of prey or being consumed by predators the next level up in the food chain. Indirect effects across two trophic levels are generally small; for instance if we compare ZF from runs 1 and 5, i.e. there is no impact of the death rate of phytoplankton on carnivorous fish.

3.2. Examples of management application of the model

The ecological sub-model is also simple, though still realistic. It incorporates the dominant six state variables. The model integrates physical, chemical and biological processes in the estuary; it predicts the ecosystem health as determined by the following variables: nutrients, suspended particulate matter, phytoplankton, zooplankton, bivalves, zooplanktivorous fish and carnivorous/omnivorous fish. Thus the model is simpler than a number of other models (e.g. Flindt and Kamp-Nielsen, 1997 – this comprises 12 state variables) that are often too complex and unwieldy for practical applications, especially when data are unavailable or insufficient.

The model can readily be used to test management scenarios when querying the impact of developments and disturbances to land-use and water-resources in the river catchment. For instance, the model predicts (Fig. 7) the impact of doubling the nutrient concentration in the Guadiana River as a result of irrigation farming downstream of the Alqueva dam. Such farming is indeed planned. The phytoplankton concentration is predicted to increase, particularly in the phytoplankton maximum zone located in the upper reaches of the estuary. This suggests that the system is becoming eutrophicated and the risk of toxic algae blooms has increased.

The model can also predict the impact of the salt marshes being destroyed by developments. The model predictions for phytoplankton are shown in Fig. 7. Clearly the risk of eutrophication and of toxic algae blooms would be further increased.

The model was used to assess the influence on the estuarine ecosystem health of the Alqueva dam that in 2002–2003 substantially decreased the river discharge Q_f (Fig. 8a). The predictions (Fig. 8b, c) show that without the dam the system was highly variable during a freshwater pulse, while with the dam the system was at steady state. The predicted influence of the Alqueva dam is particularly dramatic for the carnivorous/omnivorous fish (Fig. 8d, e) because without the dam the fish was able to spread over much of the estuary for up to a month after a freshet, while with the dam the fish is restricted to the uppermost region of the estuary. Zooplankton and zooplanktivorous fish also are predicted to decrease in the presence of the

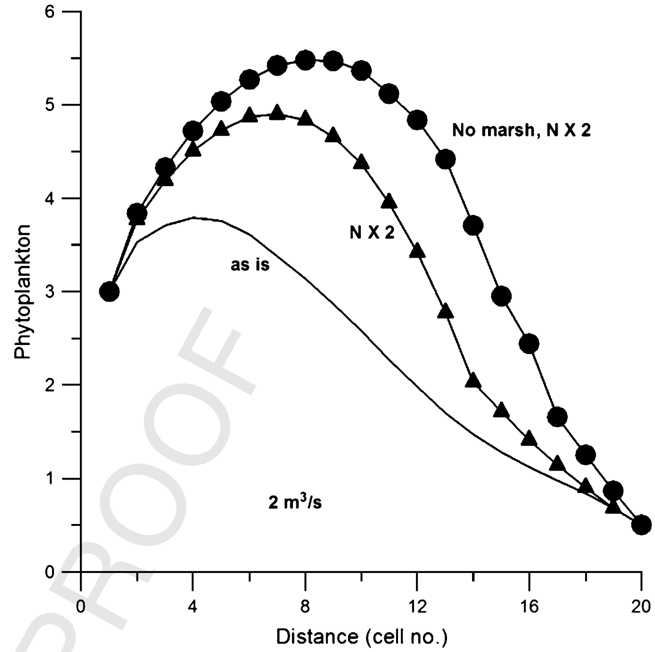

Fig. 7. Along-channel distribution of predicted phytoplankton (Chl *a*) biomass in the Guadiana Estuary for the standard run ('as is'), for a doubling of nutrient concentration in the river ('$N \times 2$'), and for the additional impact of removing the salt marshes ('No marsh, $N \times 2$') for a freshwater discharge equal to $2\ m^3\ s^{-1}$. To convert biomass to concentration for Chl *a* 3.5–7.8 $\mu g\ l^{-1}$.

Alqueva dam because their renewal and distribution depend on freshets.

Moreover, the model can also be used for finding solutions for practical existing environmental problems in the Guadiana Estuary such as toxic algal blooms and eutrophication risk. After the dam construction the estuary reached a man-made quasi-steady state characterised by poor productivity and low biomass in all communities (Fig. 8). Indeed, the fluctuations in river discharge – as freshets – as occurred historically, increased diversity and variability in plankton and nektonic communities (Fig. 8b–e), and promoted ecosystem dynamics. This model prediction is supported by the observations of Roelke (2000) in the Nueces Delta, Texas. This ecosystem response to freshwater discharge pulses can be used as a management solution for toxic algal blooms or eutrophication in the Guadiana. In the Guadiana, the model suggests that increasing Q_f to $50\ m^3\ s^{-1}$ for 5 days will flush the estuary and promote the development of a diverse phytoplankton and zooplankton communities.

The model is restricted to the estuary. It cannot predict impacts on the coastal zone. Studies are needed to determine if longer-duration and possibly higher intensity freshets may be needed to maintain coastal marine ecosystem health, as suggested by Doornbos (1982), Quiñones and Montes (2001), Chícharo et al. (2002) and Simier et al. (2004).

Thus the estuarine ecohydrology model is able to provide answers to a number of practical questions. These answers must always be taken carefully because the model, like any ecosystem model, over-simplifies reality, and the data set is inadequate for a detailed calibration. In that sense, the model predictions are somewhere between quantitative and qualitative. Detailed field studies are needed to better understand,

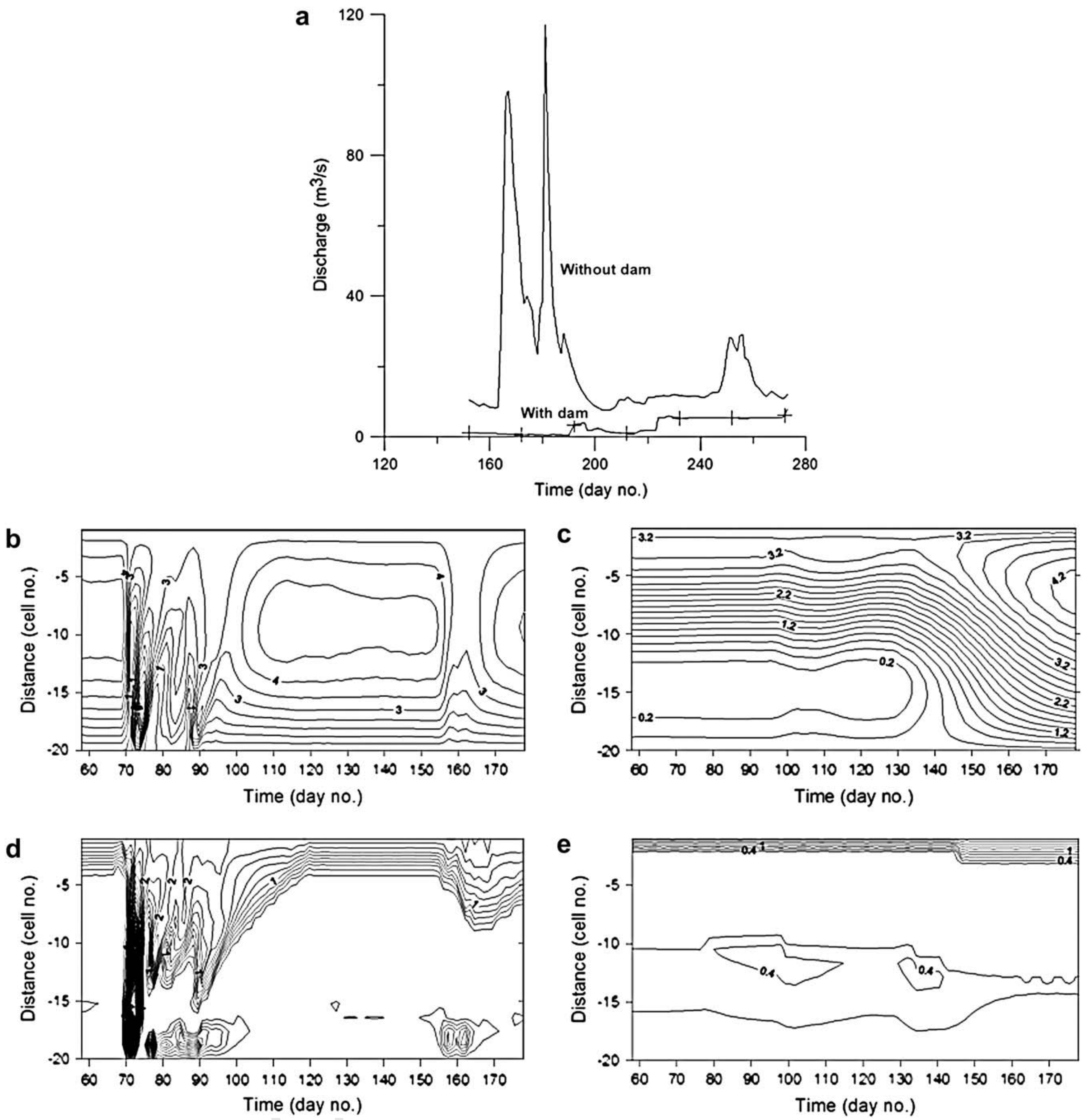

Fig. 8. (a) Time series plot of the Guadiana River discharge entering the estuary in the dry season of 2003 in the presence of the Alqueva dam, and the predicted river discharge if the dam had not been constructed (middle). Time series plot of the predicted distribution of phytoplankton biomass in the Guadiana Estuary in 2003 (b) without and (c) with the Alqueva dam. Time series plot of the predicted distribution of carnivorous/omnivorous fish biomass in the Guadiana Estuary in 2003 (d) without and (e) with the Alqueva dam. To convert biomass to concentration, see Figs. 5 and 6.

and hence better parameterise in the model, the various processes driving the ecosystem. The model should be seen as a living model – it has been written using subroutines that are readily edited, so that the new knowledge on individual processes can readily be incorporated in the model. For the model to remain a useful tool, it is suggested that its complexity should be increased only as fast as additional physical, chemical and biological processes can be quantified through new field and laboratory studies. For example, the import rate α from salt marshes and riparian ecotones, which is presently set as a constant, is probably varying seasonally and possibly stochastically – data on this are missing and are needed. Also, as the new data become available, the model should be improved by subdividing the phytoplankton compartment into the main classes (Domingues et al., 2005).

For science, the model provides a tool to enable the exchange of information between oceanographers, biologists, ecologists, engineers, sociologists, economists and water-resources managers at regional and national government levels, and the community.

It is hoped that the model can also be useful for management. The model shows that it is possible to predict – within likely error bounds provided by the sensitivity tests – the consequences on the estuary ecosystem health of human activities throughout the river catchment. The model does show that, to maintain the ecosystem services provided by the estuary, integrated coastal management needs to take the whole river catchment as the fundamental planning unit. It is necessary to bring together land-use managers, water-resources managers, and coastal and fisheries managers. The model offers thus a tool for using ecohydrology as a holistic approach to the management of rivers, estuaries and coastal zones within entire river catchments.

4. Conclusions

The ecohydrology model is original in that it links physical, chemical and biological processes over the entire estuary for the entire food web as a function of catchment output and the oceanic open boundary condition. Despite the fact that a number of simplifications are made, the model is encouraging in that it reproduces satisfactorily the observations in 2001–2003. These data are still sparse and the model may need improvements as additional data become available.

The model can readily be used to assess future impact on the Guadiana Estuary ecosystem health caused by urbanisation or other factors that reduce the salt marsh area, by an increase in nutrient loads as a result of changes in agriculture practices in the catchment area due to increase in water availability by the Alqueva dam, by extreme high freshwater discharges, e.g. due to release of high volume of water storage in the dam, and by the introduction of exotic species.

The model can also be used to predict the efficiency of remedial measures, such as creating wetlands, creating freshets by releasing water from the Alqueva dam, managing bivalve species in the freshwater part of the estuary, and removing nutrients from the river.

Acknowledgements

It is a pleasure to thank Philippe Pypaert and UNESCO – ROSTE for supporting the development of this model through contract No. 875.767.4 in estuarine and coastal zone ecohydrology. Also thanks are due to the University of Algarve for providing logistical support. The collection of the biological data was made possible by the financial support of the projects 'Effects of River Flow changes on the fish communities of the Douro, Tejo and Guadiana Estuaries and coastal areas: ecological and socioeconomic predictions (ERIC) (FCT/P/MAR/15263/1999)', 'General characterisation of the Guadiana Estuary ecosystem (FCT/P/15 REG/II/6/96)', 'Valorization of Aquatic resources of the Guadiana Estuary (Portugal) (Feder/PPDR/CCRA/ODIANA)', and the PhD scholarship of P. Morais '*Engraulis encrasicolus* population dynamics in the Guadiana Estuary and adjacent coastal area (SFRH/BD/5187/2001)'.

References

Ackroyd, D.R., Bale, A.J., Howland, R.J.M., Knox, S., Millward, G.E., Morris, A.W., 1986. Distributions and behaviour of dissolved Cu, Zn and Mn in the Tamar estuary. Estuarine, Coastal and Shelf Science 23, 621–624.

Alveirinho, J.M.A., Gonzalez, R., Ferreira, Ó., 2004. Natural versus anthropic causes in variations of sand export from river basins: an example from the Guadiana river mouth (Southwestern Iberia). Polish Geological Institute Special Papers 11, 95–102.

Balls, P.W., 1994. Nutrient inputs to estuaries from nine Scottish east coast rivers: influence of estuarine processes on inputs to the North sea. Estuarine, Coastal and Shelf Science 39, 329–352.

Billé, R., Mermet, L., 2002. Integrated coastal management at the regional level: lessons from Toliary, Madagascar. Ocean and Coastal Management 45, 41–58.

Boorman, L.A., Hazelden, J.H., Loveland, P.J., Wells, J.G., 1994a. Comparative relationships between primary productivity and organic and nutrient fluxes in four salt marshes. In: Mitsch, W.J. (Ed.), Global Wetlands. Old World and New. Elsevier, Amsterdam, pp. 181–189.

Boorman, L.A., Hazelden, J., Andrews, R., Wells, J.G., 1994b. Organic and nutrient fluxes in four north-west European salt marshes. In: Dyer, K.R., Orth, R.J. (Eds.), Changes in Fluxes in Estuaries: Implications from Science to Management. Olsen and Olsen, Fredensborg, pp. 243–248.

Brauer, F., Castillo-Chavez, C., 2001. Mathematical Models in Population Biology and Epidemiology. Springer, Berlin, 416 pp.

Chícharo, L., Chícharo, M.A., Esteves, E., Andrade, P., Morais, P., 2002. Effects of alterations in fresh water supply on the abundance and distribution of *Engraulis encrasicolus* in the Guadiana Estuary and adjacent coastal areas of south Portugal. Journal Ecohydrology and Hydrobiology 1, 195–200.

Domingues, R.B., Barbosa, A., Galvão, H., 2005. Nutrients, light and phytoplankton succession in a temperate estuary (the Guadiana, south-western Iberia). Estuarine, Coastal and Shelf Science 64, 249–260.

Doornbos, G., 1982. Changes in the fish fauna of the former Grevelingen estuary, before and after the closure in 1971. Hydrobiological Bulletin 16, 279–283.

Edwards, A.M., 2001. Adding detritus to a nutrient–phytoplankton–zooplankton model: a dynamical-systems approach. Journal of Plankton Research 23, 389–413.

Erzini, K., 2005. Trends in NE Atlantic landings (southern Portugal): identifying the relative importance of fisheries and environmental variables. Fisheries Oceanography 14, 195–209.

Esteves, E., Pina, T., Chicharo, M.A., Andrade, J.P., 2000. The distribution of estuarine fish larvae: nutritional condition and co-occurrence with predators and prey. Acta Oecologica 21 (3), 1–13.

Flameling, I.A., Kromkamp, J., 1994. Responses of respiration and photosynthesis of *Scenedesmus protuberans* (Fritsch) to gradual and steep salinity increases. Journal of Plankton Research 16, 1781–1791.

Flindt, M.R., Kamp-Nielsen, L., 1997. Modelling of an estuarine eutrophication gradient. Ecological Modeling 102, 143–153.

Fischer, H.B., List, E.Y., Koh, R.C.Y., Imberger, J., Brooks, N.H., 1979. Mixing in Inland and Coastal Waters. Academic Press, New York, 483 pp.

Fortunato, A.B., Oliviera, A., Alves, E.T., 2002. Circulation and salinity intrusion in the Guadiana Estuary. Thalassas 18, 43–65.

Gonzalez, J.A.M., 1995. Sedimentologia del estuário del Rio Guadiana (SO Portugal). CEP. Biblioteca Universitária, Universidade de Huelva.

Goosen, N.K., van Rijswijk, P., Brockmann, U., 1995. Comparison of heterotrophic bacterial production rates in early spring in the turbid estuaries of the Scheldt and the Elbe. Hydrobiologia 311, 31–42.

Haward, M., 1996. Institutional framework for Australian ocean and coastal management. Ocean and Coastal Management 33, 19–39.

Hilborn, R., Mangel, M., 1997. The Ecological Detective. Confronting Models with Data. Princeton University Press, Princeton, 315 pp.

Humston, R., Ault, J.S., Lutcavage, M., Olson, D.B., 2000. Schooling and migration of large pelagic fishes relative to environmental cues. Fisheries Oceanography 9, 136–146.

Jorgensen, S.E., Bendoricchio, G., 2001. Fundamentals of Ecological Modelling. Elsevier, 530 pp.

Kibirige, I.R., Perissinotto, R., Nozais, C., 2002. Alternative food sources of zooplankton in a temporarily-open estuary: evidence from $\delta^{13}C$ and $\delta^{15}N$. Journal of Plankton Research 24, 1089−1095.

Knox, G.A., 1986. Estuarine Ecosystems: A Systems Approach, vol. II. CRC Press, Florida, 230 pp.

Kot, M., 2001. Elements of Mathematical Ecology. Cambridge University Press, Cambridge, 464 pp.

Lau, M., 2005. Integrated coastal zone management in the People's Republic of China − an assessment of structural impacts on decision-making processes. Ocean and Coastal Management 48, 115−159.

Lefeuvre, J.C., 1996. Effects of environmental change on European salt marshes. In: Structure, Functioning and Exchange Potentialities with Marine Coastal Waters, vols. 1−6. University of Rennes, France.

Lindeboom, H., 2002. The coastal zone: an ecosystem under pressure. In: Field, J.G., Hempel, G., Summerhayes, C.P. (Eds.), Oceans 2020: Science, Trends, and the Challenge of Sustainability. Island Press, Washington, pp. 49−84, 365 pp.

Michel, D., 1980. Synthese des donnes physiques mesures dans le Rio Guadiana. Evaluation de l'intrusion saline dans l'estuaire. PhD thesis, Laboratoire d'oceanographie, Université de Bruxelles.

Nixon, S.W., 2003. Replacing the Nile: are anthropogenic nutrients providing the fertility once brought to the Mediterranean by a great river? Ambio 32, 30−39.

Pickaver, A.H., Gilbert, C., Breton, F., 2004. An indicator set to measure the progress in the implementation of integrated coastal zone management in Europe. Ocean and Coastal Management 47, 449−462.

Pinto, G.V., 2000. Caracterização Dos Bancos Naturais De Bivalves Com Interesse Comercial No Estuário Do Guadiana. Tese de licenciatura Biologia Marinha, Universidade do Algarve.

Quiñones, R.A., Montes, R.M., 2001. Relationship between freshwater input to the coastal zone and historical landing of the benthic/demersal fish in Central-South Chile. Fisheries Oceanography 10, 311−328.

Roelke, D.L., 2000. Copepod food quality threshold as a mechanism influencing phytoplankton succession and accumulation of biomass, and secondary productivity: a modelling study with management implications. Ecological Modelling 134, 245−274.

Salomons, W., Forstner, U., 1984. Metals in the Hydrocycle. Springer-Verlag, Berlin, 349 pp.

Simenstad, C.A., Morgan, C.A., Cordell, J.R., Baross, J.A., 1994. Flux, passive retention, and active residence of zooplankton in Columbia river estuarine turbidity maxima. In: Dyer, J., Orth, M. (Eds.), Changes in Fluxes in Estuaries: Implications from Science to Management. University of Plymouth, pp. 473−482 (ECSA22/ERF Symposium).

Simier, M., Blanc, L., Aliaume, C., Diouf, P.S., Albaret, J.J., 2004. Spatial and temporal structure of fish assemblages in an ''inverse estuary'', the Sine Saloum system. Estuarine, Coastal and Shelf Science 59, 69−86.

Tagliani, P.R.A., Landazuri, H., Reis, E.G., Tagliani, C.R., Asmus, M.L., Sánchez-Arcilla, A., 2003. Integrated coastal zone management in the Patos Lagoon estuary: perspectives in context of developing country. Ocean and Coastal Management 46, 807−822.

Thomann, R.V., 1980. Deterministic and statistical models of chemical fate in aquatic systems. In: Levin, S.A., Hallam, T.G., Gross, L.J. (Eds.), Applied Mathematical Ecology. Springer-Verlag, Berlin, pp. 322−351, 491 pp.

Uncles, R.J., Stephens, J.A., Smith, R.E., 2002. The dependence of estuarine turbidity on tidal intrusion length, tidal range and residence time. Continental Shelf Research 22, 1835−1856.

Wolanski, E., Doherty, P., Carleton, J., 1997. Directional swimming of fish larvae determines connectivity of fish populations on the Great Barrier reef. Naturwissenschaften 84, 262−268.

Wolanski, E., Boorman, L.A., Chícharo, L., Langlois-Saliou, E., Lara, R., Plater, A.J., Uncles, R.J., Zalewski, M., 2004. Ecohydrology as a new tool for sustainable management of estuaries and coastal waters. Wetlands Ecology and Management 12, 235−276.

Zalewski, M., 2002. Ecohydrology − the use of ecological and hydrological processes for sustainable management of water-resources. Hydrological Sciences Bulletin 47, 823−832.