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Modelling water discharges and nitrogen inputs into a Mediterranean lagoon Impact on the primary production

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Abstract: The Soil and Water Assessment Tool (SWAT model, 2001 Version) has been applied to the Thau lagoon catchment area in order to simulate water discharges and nutrient inputs into the lagoon over a 10 years period (1989–1999), and to provide routing inflows to a previously developed lagoon ecosystem model. The watershed model has been calibrated and validated using measured data available for the two main rivers. The results indicate that the mean annual nitrogen inputs into the Thau lagoon is 117 ± 57 tons y−1, with the two main rivers, contributing for 80% of total annual nitrogen export. The variations of outputs to the lagoon are nonetheless important from 1 year to another. Due to the local agricultural practices and a reduced in-stream natural depuration, point sources seem to be the main factor affecting the fresh water quality.

The coupling with the lagoon model allowed to estimate the impact of those terrestrial inputs on the lagoon nitrogen cycling and primary productivity. Influence of river discharges makes itself felt essentially near the river outlets. The northern bordure of the lagoon is then characterised by highly variable dissolved inorganic nitrogen concentrations, especially during flood events, while more stable and lower concentrations were simulated in the southern part of the lagoon. Simulated chlorophyll a ranged 1–15 µg l−1, with maximums in March. Mean annual phytoplankton production was 364 ± 142 gC m−2. The simulations showed that maximum annual productions are due to macrophytes (up to 1300 gC m−2 y−1), but at the whole lagoon scale, annual phytoplankton production resulted greater. From our results it also appeared that the greatest part of primary producers nitrogen requirements is satisfied by nutrient regeneration within the lagoon.

Keywords: Watershed model; Lagoon model; Integrated modelling; Primary production; Thau lagoon

1. INTRODUCTION

Coastal lagoons represent a tiny part (less than 1%) of the surface covered by oceans, but they are nonetheless characterized by high biodiversity and intense primary productions, from 200 to 400 $gC.m^{-2}.y^{-1}$ according to Nixon (1982), that lead to both ecological and economical considerable importance. As transition media between land and ocean, they play a filtering role with respect to materials coming from their catchment area, which makes these ecosystems particularly sensitive to upstream dysfunctions. Moreover, the close link between human population and coastal zones, and consequently coastal lagoons, is particularly evident since approximately 2/3rd of human total population live in these areas. Indeed, terrestrial human activities - agriculture, tourism, industry - generally meet marine human activities - fishing, bathing, yachting, aquaculture -, and this pressure is likely to increase with population growth. As a matter of fact, since several decades, increasing human wastes have created important problems of diffuse and point source pollution of chemical (pesticides, heavy metals, etc.), bacterial or trophic (nutrients, organic matter, etc.) origins. In order to allow a sustainable development of coastal zones, the quantity and fate of inputs coming from the catchment has to be evaluated, as a first step to manage these systems.

In this work we focused on a Mediterranean area including a coastal lagoon (Thau lagoon) and its watershed, located on the French south coast around the Lion Gulf. The lagoon has a notable economic importance due to shellfish cultivation (annual oyster production of about 13 000 tons). Several models have been already developed, involving hydrodynamics (Millet, 1989, Lazure, 1992), plankton food web and nutrient cycles (Chapelle, 1995, Chapelle et al., 2000), shellfish biomass and growth (Bacher et al., 1997, Gangnery et al., 2003), macrophyte populations (Plus et al., 2003 a and b). On the contrary, very few models calculating riverine inputs to the Thau lagoon are available in the literature since rivers have been generally studied from an experimental point of view (Guilbot and Tournoud, 1991, Monna et al., 1995, Ben Othman et al., 1997, Anonymous, 1997, Luck and Ben Othman, 1998). As far as we know, two models have been developed on Thau lagoon watershed: one considers the Pallas river basin and calculates total annual nitrogen and phosphorus outputs (Payraudeau et al., 2001) and the other one focuses on phosphorus export for the whole Thau catchment area, also on a yearly basis (La Jeunesse, 2001). These models are useful when considering an annual nutrient budget but they do not allow the simulation of nutrient exports at a time scale compatible with the dynamics of the lagoon ecosystem (hourly or daily).

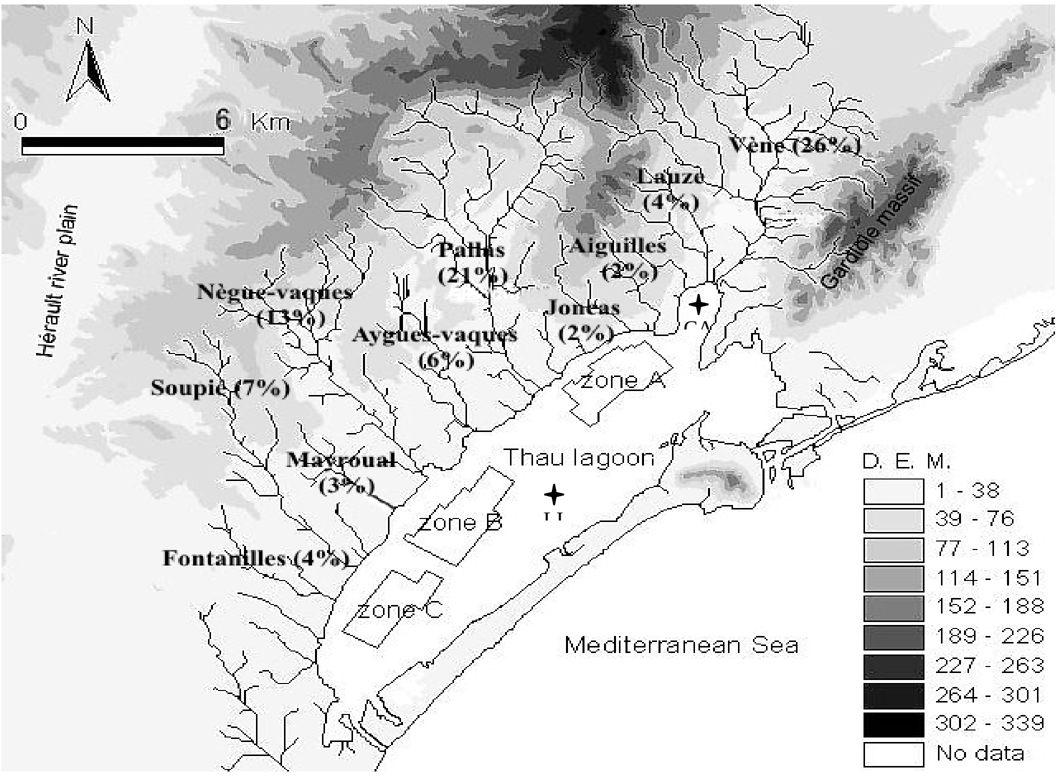
The aim of the present study was to model the Thau catchment area in order to simulate the freshwater and nutrient inputs in the lagoon at space scales and time steps compatible with some previously developed lagoon models (Chapelle et al., 2001, Plus et al., 2003 b) and to provide the lagoon model (LM) with simulated riverine inflows in order to assess their impact on the primary production.

2. STUDY SITE

The Thau lagoon (surface: 70 km^2) is one of the *c.a.* thirty lagoons located on the French Mediterranean coast (Languedoc region). It is a shallow semi-confined ecosystem (mean depth 4 m), with two narrow openings towards the sea located at its extremities (figure 1). The Mediterranean climate imposes a wide range of temperatures and salinities (5-29°C and 27-40, respectively) and the wind plays an important role in the lagoon hydrodynamics (Lazure, 1992). The nitrogen was recognized as the limiting factor as it is usually the case in coastal lagoons, whilst phosphorus habitually controls the primary production in fresh waters

(Hecky and Kilham, 1988). In fact, this point was validated for the Thau lagoon in several studies focusing on phytoplankton (Picot et al., 1990, Souchu et al., 1998, Chapelle, 1995) as well as on macrophytes (Plus et al., 2005). Picot et al. (1990) showed for example that the ratio of total inorganic nitrogen to phosphate is always very low in the Thau waters (mean ratio 1.6) and that the development of phytoplankton is positively correlated to the quantity of ammonium present in the water column. Moreover, the phytoplankton biomass is not correlated with phosphate concentrations and the phosphate levels do not fall below 0.1 µmol l^{-1} in the Thau lagoon while the maxima habitually encountered in Mediterranean surface waters are *c. a.* 0.05 µmol l^{-1}.

Figure 1. General map of the Thau lagoon and its watershed: digital elevation model (D.E.M., 50 m grid, values expressed in m), and river network. Names of rivers are reported as well as their catchment area (expressed as a % of total watershed surface). Zones A, B and C represent the shellfish farming areas and diagnostic points LI and CA are reported at their exact positions in the lagoon.



The Thau watershed extends over about 280 km^2 and is drained by numerous little streams (3-13 km) with intermittent flows. It is delimited by the Aumelas massif (altitude, *c.a.* 300 m above sea level) to the north, the Gardiole massif to its eastern rim and by the Hérault river basin to the west (figure 1). Toward the south, a narrow sandy strip separates the lagoon from the sea. This area, called Lido, accounts for only 5% of total catchment area , with very low mean slope and sandy sediments, which renders the inputs extremely diffuse.

Following Ben Othman et al. (1997) and Anonymous (1997), two different zones can be differentiated on the catchment area, using geomorphologic criteria : on one hand the eastern part of the catchment area is composed of strongly karstic Jurassic limestone, overlaid by

Miocene marls in its central part. This area corresponds more or less to the Vène river watershed, which is fed by two karstic resurgences, the Vène and Issanka springs. On the other hand the western part is mainly composed of Eocene marls and Miocene molasses. This area concerns the Pallas river and all other little streams to the west. The river response delays to rainy events are supposed to be shorter in this area than in the karstic zone, where more water infiltrations are likely to occur (Anonymous, 1997).

The land cover map (figure 2) has been drawn up on the basis of aerial colours photographs taken in 1996 at a 1/25000 scale (French National Geographic Institute), and validation has been performed by field observations (La Jeunesse et al., 2002). Natural areas spread over a large part of the catchment area, covering 103 km^2. They are mostly located in the north east, and represent about 35% of the total surface, with the natural Mediterranean sclerophyllous vegetation (low scattered bushes with bare patches of rock or stony ground between) locally called 'garrigue', as the main component (93 km^2). The remaining natural vegetation (10 km^2) is composed of salt marshes, mostly located on the Lido and near the south-west extremity of the lagoon. The typical crop landscape on the Thau watershed is composed of vineyards, which covers 40% (109 km^2) of total catchment area and represent on a surface basis 67% of arable land. Other crops are mainly composed of durum wheat (7% of total watershed surface, 20 km^2) as well as some scarce parcels of fruit trees (in total 2 km^2). The fallow lands represent in fact about 10% of the total watershed surface, as well as urban areas, composed of small villages and little cities distributed all over the catchment area (figure 2 and table 1). In summer the population increases sensibly due to tourism, with an increase of population ranging 10% - 770% during June, July and August (table 1). The largest city in the watershed is Sète, and it is also where the main industrial activity takes place. The pollutant emissions caused by this city are treated by a plant having its outlet in the Mediterranean Sea. Thus, it seems that these discharges do not concern the rivers of the watershed (Monna et al., 1995). On the contrary, the vine industry concerns directly the rivers located on the Thau watershed. In each village, the winegrowers cooperative collecting the grapes from the surroundings vineyards, discharge a large amount of organic wastes. Until the seventies all discharges were made directly in the rivers, but progressively, cooperatives have been connected to the existing urban wastewater treatment plants.

Figure 2. Land cover map (1996) of the Thau catchment area.

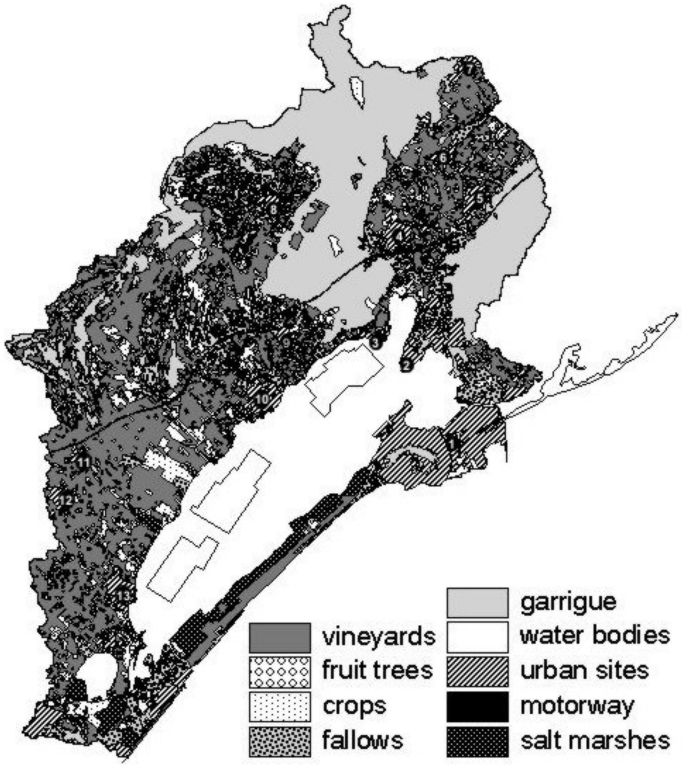


Table 1. List and importance of urban sites located on the Thau lagoon watershed. Identification numbers (Id) are reported on figure 2.

Id	Name	Surface (km^2)	Permanent population†	Summer-increase
1	Sète	10.2	42 738	120 %
2	Balaruc	2.8	6 962	300 %
3	Bouzigues	0.7	1 014	180 %
4	Poussan	1.1	3 563	150 %
5	Gigean	0.7	2 847	110 %
6	Montbazin	0.7	2 490	110 %
7	Cournonsec	0.6	1 569	110 %
8	Villeveyrac	0.6	2 026	150 %
9	Loupian	0.4	1 399	180 %
10	Mèze	1.6	6 977	160 %
11	Pinet	0.4	944	110 %
12	Pomérols	0.7	1 837	150 %
13	Marseillan	1.6	5 432	870 %

† Number of inhabitants in 1998 (La Jeunesse, 2001)

3. MODEL DESCRIPTION

3.1 The watershed model (WM)

For our purpose, a continuous and distributed model was needed in order to estimate total daily loads as a function of climatic seasonal variations as well as of spatial variations of soil type, topography and land cover. We chose to use the Soil and Water Assessment Tool (SWAT) model (Srinivasan and Arnold, 1994), that simulates on a daily time step, sediment erosion, surface runoff and subsurface flow, nutrient loads, crop growth and agricultural management. Point sources are also taken into account, and can be inserted in the model as constant or varying inputs. An interface between SWAT and Arcview® allows running the model on a Geographical Information System that contains a Digital Elevation Model (D.E.M.), a map on land cover/land use, a soil map and a stream network map.

SWAT uses the SCS curve number methodology (Soil Conservation Service, 1972) and the infiltration method of Green and Ampt (1911) to calculate the water surface runoff. Then, sediments (based on the Modified Universal Soil Loss Equation, Williams, 1975, a modified version of USLE developped by Whischmeier and Smith, 1965 and 1978) and nutrients (based on soil organic matter content and classical biochemical cycles) may be transported through surface runoff and lateral subsurface flows towards the stream network. SWAT uses a plant growth module in order to simulate properly the uptake of water and nutrients from the root zone as well as evapotranspiration and biomass production. Then, the water and sediments/nutrients loads are routed (dissolved and particle transport) towards the outlet of the watershed taking into account the following in-stream processes: dilution, diffusion, evaporation, sedimentation/resuspension, sorption onto sediment particles, biodegradation (algal growth). A complete and detailed description of the SWAT model has been already published and can be consulted on the following web site :

www.brc.tamus.edu/swat/swatdoc.html

Ten streams have been selected in order to take into account the major contributions to total inputs into the lagoon (figure 1). One can notice that two river basins (the Vène and the Pallas) cover about the half of the whole catchment area, while other river basins are much smaller. Required climatic data (daily precipitation, minimum and maximum air temperature, solar radiation, wind speed and relative humidity) were provided by Météo-France and Hydrosciences (Montpellier University) and concerned six different weather gauges: five located on the Thau watershed and the last one at the Fréjorgues Airport, about 20 km from the north-east.

Point sources calculation

Wastewater treatment plants (WWTP) are the principal point sources (PS) that have to be considered on the Thau catchment. They collect the wastes coming from the cities as well as the extra-loads due to wine production. In order to take properly into account the seasonal variations of PS (summer increasing population and grape-harvesting period), three periods in the year have been distinguished: from November to May, the permanent population, from June to August, the permanent population plus the summer population and from September to October, the permanent population plus the wine cooperatives. The calculation of waste quantities that exit from a wastewater treatment plant is based on the inhabitant equivalent (IE) concept. According to the EEC N. 91/271 Directive, one IE is the *organic biodegradable load having a five-day biochemical oxygen demand (BOD5) of 60 g of oxygen per day, i. e. the quantity of substances for which the oxygen consumption in aerobic degradation equals that of the wastewater of one inhabitant per day*. The IE has then been converted into mean quantities of water, organic matter, nitrogen etc… using values that are usually admitted for a European inhabitant in rural areas (summarised in table 2). For the extra loads, we assumed that the production of 10 hectolitres of wine generates 1 IE wastes during the vinification period (La Jeunesse, 2001). Finally, a nitrogen removal efficiency factor (RE, expressed in %) has been applied to the total IE that enters a WWTP, in order to estimate the output flowing into the rivers. RE depends on whether the treatment applied is lagooning (LAG) or activated sludge (AS): 70% in LAG-WWTP, 80% in AS-WWTP (La Jeunesse, 2001).

Table 2. Mean daily quantities of wastes rejected by one inhabitant (after Ondeo-Degrémont, 2000).

Water	150 l
Total suspended matter (SM)	90 g
% mineral	30%
Total Organic matter (OM)	57 g
Reduced nitrogen (Kjeldahl nitrogen, N_K)	14 g
% ammonia in N_K	50%
Nitrate - Nitrite	≈ 0 g
Total phosphorus (Ptot)	3 g
% mineral phosphorus in Ptot	75%

Soil parameterisation

Physico-chemical soil characteristics are needed by SWAT to calculate the movements of water and air through the soil profile: soil hydrologic group (*Hyd*), soil depth profile, moist bulk density (*BD*), available water capacity (*AWC*), saturated hydraulic conductivity (*Ks*), organic carbon content (*OC*), clay (*C*), silt (*S*), sand (*SA*) contents and soil erodibility factor (USLE_K). The soil typology was provided by the INRA-Montpellier soil database (Jamagne et al., 1996, Bornand et al., 1998), in which each soil type can be linked to a soil profile description containing: number and thickness of each soil layers, organic matter, silt, clay and sand contents. Soil hydrologic groups (A, B, C or D) have been defined by the Soil Conservation Service (1972). Group A soils have low runoff potential due to high infiltration rates, groups B and C are intermediates and group D soils have a high runoff potential caused by very low infiltration rates. All other parameters were deduced from these data using the following pedotransfer functions:

$$BD = 1.8 + 1.236 \times \frac{OM}{100} - 2.91 \times \sqrt{\frac{OM}{100}}$$ Bollen et al. (1995) (1)

with *OM*, the organic matter content expressed in %.

$$AWC = \frac{\sigma_s - \sigma_r}{\left[1 + (\alpha h_{50})^n\right]^m} - \frac{\sigma_s - \sigma_r}{\left[1 + (\alpha h_{15000})^n\right]^m}$$ Van Genuchten (1980) (2)

$$Ks = e^{(Ks^*)}$$ Wösten et al. (1999) (3)

with σ_s and σ_r, the saturated and residual soil-water contents ($cm^3\ cm^{-3}$) respectively, α and Ks^*, two Mualem – Van Genuchten parameters (Wösten et al., 1999, expressed in cm^{-1}), h_{50} and h_{15000}, the pressure head (cm) at the field capacity and at the permanent wilting point, respectively.

$$USLE\_K = f_{cSA} \times f_{C-S} \times f_{OC} \times f_{hiSA}$$ Williams (1995) (4)

$$\text{with}\ \begin{cases} f_{cSA} = 0.2 + 0.3 \times e^{\left[-0.256 \times SA \times \left(1 - \frac{S}{100}\right)\right]} \\ f_{C-S} = \left(\dfrac{S}{C+S}\right)^{0.3} \\ f_{OC} = 1 - \dfrac{0.25 \times OC}{OC + e^{(3.72 - 2.95 \times OC)}} \quad ; \quad OC = \dfrac{OM}{1.72} \\ f_{hiSA} = 1 - \dfrac{0.7 \times \left(1 - \frac{S}{100}\right)}{1 - \frac{S}{100} + e^{\left[-5.51 + 22.9 \times \left(1 - \frac{S}{100}\right)\right]}} \end{cases}$$

Land use parameterisation

Each time it was possible parameter values for plant growth and management, runoff coefficients etc… were kept unchanged from SWAT databases. Only two of the land covers that are present on the Thau watershed, the garrigue and the vineyards, were not already included in the SWAT plant growth database. Table 3 summarises the main parameters that have been added to the database for vineyards and garrigue. The management of vineyards in the model has been kept as simple as possible. Growth begins the 15th of February and the grape harvest is done on the 15th of September. The runoff curve coefficient (CN2, depending on the hydrologic soil group) has been set with the values provided by the Soil Conservation Service (1986): CN2 = 72, 81, 88, 91 for soils A, B, C and D respectively (default values given for "row crops, straight row"). Furthermore, a survey made by the Chambre d'Agriculture de l'Hérault (2000 a and b) concluded to the very high heterogeneity in management practices for vineyards. It appeared that some cultivators do not fertilize and some others use relatively low quantities of fertilizers in comparison with what is spread on orchards or wheat fields for example. We thus chose to consider the vineyards as non fertilized in the present study. For garrigue land cover the SWAT default values given for "arid, semiarid range, piñon-juniper” were taken (CN2 = 75, 75, 85, 89 for soils A, B, C and D respectively).

Table 3. Main plant growth parameters for vineyards and garrigue, other parameters were kept equal to those given respectively for "Row Crops, straight row" and "Range Brush". For a detailed description of parameters, refer to the SWAT User's Manual, Chapter 14. References: 1=tentative; 2=Castellan-Estrada (2001); 3=Castellan-Estrada et al. (2002) ; 4=La Jeunesse (2001).

Parameter		Value (unit)	Ref
Vineyards			
BIO_E	Radiation use efficiency	24 (kg/ha)/(MJ/m^2)	2
HVSTI	Harvest index for optimal growing conditions	0.6 (-)	3
BLAI	Maximum potential leaf area index	2 (-)	2
CNYLD	Normal fraction of nitrogen in yield	0.01 (kg N/kg Yield)	2
CPYLD	Normal fraction of phosphorus in yield	0.005 (kg P/kg Yield)	1
USLE_C	Minimum value for water erosion applicable to the land cover/plant	0.1 (-)	1
RDMX	Maximum root depth	1 (m)	2
T_OPT	Optimal temperature for plant growth	22 (°C)	1
T_BASE	Minimum temperature for plant growth	7 (°C)	1
OV_N	Manning's "n" value for overland flow	2 (-)	1
Garrigue			
T_OPT	Optimal temperature for plant growth	22 (°C)	1
T_BASE	Minimum temperature for plant growth	5 (°C)	1
USLE_C	Minimum value for water erosion applicable to the land cover/plant	0.009 (-)	4
OV_N	Manning's "n" value for overland flow	3 (-)	1

The growth parameters as well as the management practices (tillage, fertilization, etc.) for all other land uses (orchards, wheat, fallow lands etc.) were kept unchanged from the SWAT databases and can be consulted on the beforehand mentioned web site. As an example the urban areas parameters were taken from the "urban area – residential medium/low density" category (CN2 = 31, 59, 72, 79 for soils A, B, C and D respectively). The quantities of spread fertilizers were taken from a FAO et al. international survey (2002) : 80 kg N ha^{-1} - 80 kg P ha^{-1} and 50 kg N ha^{-1} - 20 kg P ha^{-1} for wheat fields and orchards respectively.

3.2 Calibration and validation

The calibration was made on the 1993-1994 period, for the two biggest rivers the Vène and the Pallas rivers. Even if the phosphorus cycle is calculated by the SWAT model, in the present study the focus was put on nitrogen since it has been recognized as the limiting nutrient in the Thau lagoon. All the parameters that have been tuned in order to reach the best fit, are listed in Annex 1. The only parameter for the Pallas river differing from those of the Vène river is the delay time for recharge. This modification expresses the differences between the two river basins' geomorphology. For all other little streams, the parameters were considered equal to the Pallas river ones. All parameters remain within the range of values usually accepted in the literature.

Model fit was tested using the method recommended by Mesplé et al. (1996) allowing to compare simulations to experimental measurements which can be also subjected to error : Major Axis Regressions were performed between simulated water flows and measurements (regression equation : $X_{observed} = a \times X_{simulated} + b + \zeta$). For all other variables dissolved inorganic nitrogen (DIN namely ammonia, NH_4 and nitrate, NO_3) and organic nitrogen (ON), the same direct analysis was not possible due to time scale discrepancies (measurements are instantaneous and simulated values are integrated over a daily period). We nevertheless transformed the data into daily fluxes using linear extrapolation, and compared with the simulations using the same method. Following Mesplé et al. (1996), the model may be regarded as perfectly simulating, on average, the observations when *a* and *b* are not significantly different from 1 and 0 respectively. Three other possibilities are : (i) *a* is not significantly different from 1, but *b* differs significantly from 0, then it exists a constant difference between observations and simulations, (ii) *a* differs significantly from 1 and from 0, but *b* is not significantly different from 0, then the discrepancy between simulations and observations is proportional to the variable value, and (iii) *a* is not significantly different from 0, then no relationship exists between simulations and measurements.

Simulations were run on a ten years period (1990-1999), and the results were taken into account only after 3 years of simulation, i. e. during 7 years from 1993 to 1999, in order to reach a "biogeochemically" acceptable status of the model (time for the model to “forget” the initial conditions). These simulations were used to provide routing daily inputs into the lagoon model (LM).

3.3 Sensitivity analysis

A sensitivity analysis was performed in order to know the impact of parameters on the model results. The value for each parameter was ± 10% modified before running the model, and the resulting simulation was compared to a standard simulation (Vène river, year 1994). For each tested parameter, a sensitivity index (SI) has been calculated as described below :

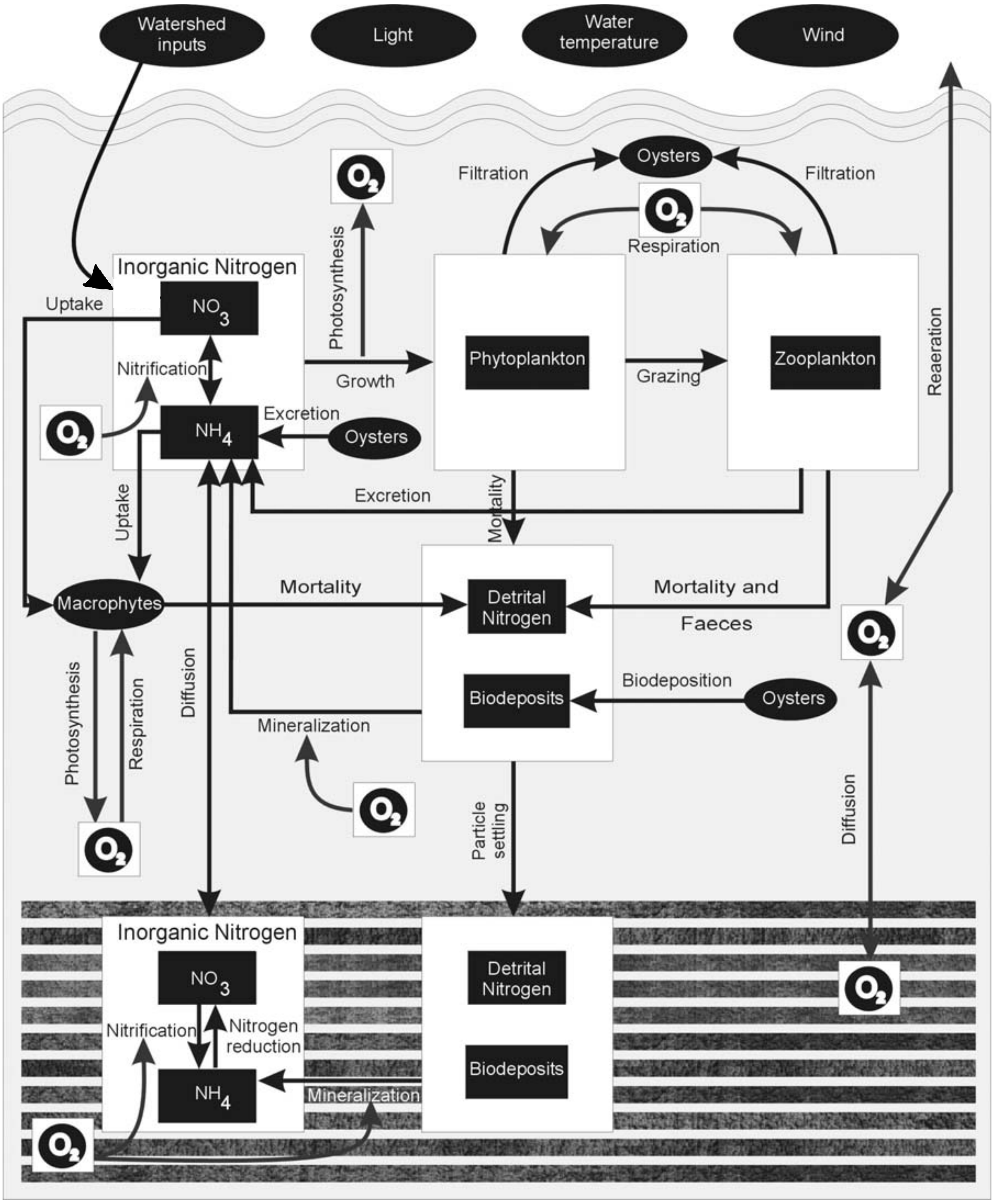
$$SI = \frac{100}{p} \times \frac{1}{n} \times \sum_{i=1}^{n} \frac{\left|X_i - X_i^{st}\right|}{X_i^{st}}$$

with p the % of parameter variation (10 % in our case), n the number of simulated days (365), X_i and X_i^{st} the new and the standard value for variable X respectively. The SI mean value for +10% and –10% variations was then calculated. Fifty parameters controlling main processes (water routing, surface runoff, evapotranspiration, percolation, soil water and ground-water, nutrients…) were tested.

3.4 The lagoon model

The LM is a three-dimensional model coupling hydrodynamics and biogeochemical processes. Spatial horizontal resolution is 400 m (squared meshes) and the water column is divided into ten layers. The model and its calibration and validation processes have been already described in details in Chapelle et al. (2000) and in Plus et al. (2003 a). Figure 3 presents the conceptual diagram of the biogeochemical module. As mentioned before, nitrogen was recognized as the limiting nutrient in the Thau lagoon, and this is the reason why the model was based on nitrogen fluxes. Seven state variables were considered : ammonia, nitrates, phytoplankton, zooplankton, detritus, oyster biodeposits and dissolved oxygen. The lagoon primary production is divided in the model into phytoplankton and macrophyte productions. Macrophyte communities are taken into account as forcing variables in the LM, *i. e.* the model simulates only the impact of macrophytes on the different state variables (production, respiration and nutrient uptake) whilst the seasonal variation of macrophyte biomass is fixed (for a more detailed description, refer to Plus et al., 2003 a). Other forcing variables are light, water temperature, wind, oyster farming and nitrogen inputs from the watershed.

Figure 3. Conceptual diagram of the biogeochemical model (after Chapelle et al., 2000, Plus et al., 2003 a). Rectangles represent state variables, arrows mean processes and the forcing functions are represented by black ovals.



Sensitivity analysis of the model parameters (Chapelle et al., 2000) as well as of the biomass of macrophytes (Plus et al., 2003 a) were previously performed and the detailed results will not be presented again here. Nonetheless, in order to sum up, these analyses enlightened the sensitivity of parameters linked to zooplankton state variable. As frequently observed in this kind of models, the higher trophic levels which are not controlled by other predators are difficult to model (Chapelle et al., 2000). Hence, further experimental studies would be necessary to ensure the precision of the related parameters (grazing, assimilation and excretion for instance). Other parameters linked to primary production (temperature coefficient, nitrogen mineralization rates, production rates, optimal light intensities, respiration rates etc.) remained less sensitive with a maximum of 5% induced variations when ±10% was applied to the different parameters. Furthermore, the variation imposed on macrophyte biomass (+10% and –10%) had little influence on pelagic primary productivity with respectively –2.4% and +0.2% induced variations (Plus et al., 2003 a).

Given the fact that the WM and the LM do not run with the same time step (1 day for the former and 200 to 500 seconds for the latter), the two models could not run at the same time. The WM was run beforehand on the entire period, providing to the LM at each time step, the correspondent calculated water flows and nitrate, ammonia and organic nitrogen daily loads.

4. RESULTS

4.1 Watershed model

Simulations have been compared with the measurements performed during the "Contrat pour l'Etang de Thau" (Anonymous, 1997) campaign (figure 4). As expected, the regime of the rivers is characterised by high values during the autumn and winter months while extremely low flows are found during summer. This, in turn, implies highly variable nitrogen discharges into the lagoon, with pikes usually encountered from September to February whilst summer period is always characterised by minimum loads. This general pattern is well simulated, as well as some sudden floods that usually occur under Mediterranean climates (see for example the rapid and short increasing flow, reaching 21.2 and 13.3 m^3/s for the Vène and the Pallas respectively in October 1994). The simulated mean annual watershed discharge into the lagoon, calculated on a seven years period (1993-1999), is $72 \cdot 10^6 \pm 42 \cdot 10^6$ m^3. Table 4 reports the results of the Major Axis Regressions performed between simulations and measurements. The best correlations were found for the nitrate loads and river flows and the worst for the ammonia loads, and in a general way, results for the Vène were found slightly better than those for the Pallas. It appears furthermore that the model tends to underestimate the Vène flow and overestimate the Pallas flow, and that these discrepancies tends to increase with the water flow. The model tends moreover to overestimate the daily ammonium loads.

No correlation was found between either the Vène or Pallas river flows and ON, NO_3 or NH_4 concentrations. Then, it can be supposed that the nitrogen discharges at the river outlets are mainly driven by the river water flows. Indeed, positive correlations were found between the simulated water flows and the ON, NO3 and NH4 discharges at the river outlets ($p<0.01$, r^2 = 0.67, 0.75 and 0.38 respectively for the Vène and $p<0.01$, r^2 = 0.18, 0.78 and 0.08 respectively for the Pallas).

Figure 4. Comparison between simulations (black line) and measurements (black rhombs) for at the two main river outlets : the Vène and the Pallas. Data (September 1993 to July 1996) are taken from Anonymous (1997).

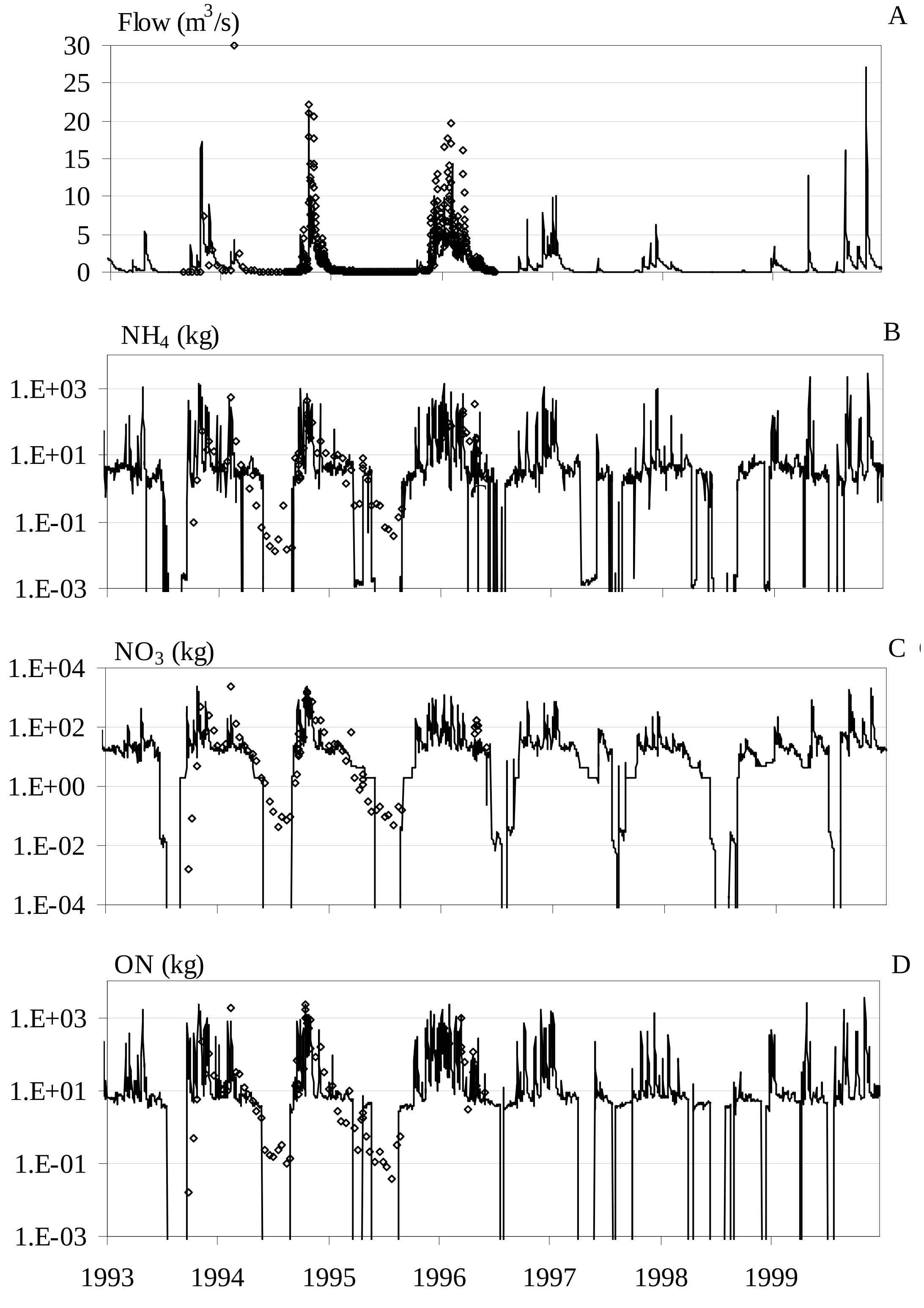


Table 4. Results of the Major Axis Regressions between simulated flows, nitrates (NO_3), ammonia (NH_4) and organic nitrogen (ON) freshwater concentrations and observed values (Anonymous, 1997) : a = slope, b = y-intercept of the regression line ($X_{obs} = a \times X_{sim} + b + \zeta$).

River	Variable	Parameter	Values	Lower limit (95%)	Upper limit (95%)	Conclusions
Vène	Water flow	a	1.59	1.53	1.66	$a \neq 0$; $a > 1$
		b	-0.05	-0.22	0.11	$b \approx 0$
		r^2	0.68	-	-	-
	NO_3	a	0.88	0.73	1.04	$a \neq 0$; $a \approx 1$
		b	10.10	-89.135	109.34	$b \approx 0$
		r^2	0.44	-	-	-
	NH_4	a	0.62	0.51	0.73	$a \neq 0$; $a < 1$
		b	-0.52	-17.75	16.70	$b \approx 0$
		r^2	0.31	-	-	-
	ON	a	1.019	0.89	1.15	$a \neq 0$; $a \approx 1$
		b	-25.04	-82.04	31.96	$b \approx 0$
		r^2	0.66	-	-	-
Pallas	Water flow	a	0.61	0.57	0.64	$a \neq 0$; $a < 1$
		b	-0.09	-0.14	-0.03	$b \neq 0$
		r^2	0.45	-	-	-
	NO_3	a	0.99	0.70	1.29	$a \neq 0$; $a \approx 1$
		b	-2.03	-22.18	18.12	$b \approx 0$
		r^2	0.27	-	-	-
	NH_4	a	0.22	0.17	0.28	$a \neq 0$; $a < 1$
		b	-4.22	-11.274	2.83	$b \approx 0$
		r^2	0.15	-	-	-
	ON	a	0.90	0.69	1.11	$a \neq 0$; $a \approx 1$
		b	-70.11	-128.61	-11.61	$b \approx 0$
		r^2	0.20	-	-	-

$\approx$: not significantly different from ; $\neq$: significantly different from.

Figures 5 and 6 present the results obtained for the two main rivers during 7 years from 1993 to 1999 and figure 7 presents some of the results simulated for the other little streams. These last simulations, unfortunately, could not be validated due to a lack of experimental data. The total annual riverine inputs into the Thau lagoon was calculated as the mean of the simulated total annual discharges for years 1993 to 1999, table 5 summarizes the obtained results. The first characteristic that can be deduced from those results is the very high variability of annual discharges, which is not surprising if we consider the Mediterranean climate characteristics (see for example the particularly dry year simulated during 1998, with a maximum Vène river flow of 1.8 m^3/s and a mean annual flow of 0.14 m^3/s). Nevertheless these simulated values still have to be validated, since no measurement was available for this year. The estimated total annual nitrogen and phosphorus inputs into the Thau lagoon are respectively 117 ± 57 tons/year and 24 ± 12 tons/year. The two largest rivers, the Vène and the Pallas, account for respectively 42% and 38% of total annual nitrogen export and 28% and 27% of total annual phosphorus export. Moreover, the ON:DIN and OP:DIP predicted ratios are smaller than 1 for the Vène and the Pallas whereas they are greater than 1 for all other smaller rivers. This points out the strong impact of the episodic character of smallest rivers on the in-stream mineralization of organic matter and thus that point sources will obviously have a considerable importance in the water quality at the river outlets.

Table 5. Simulated total annual river discharges (annual mean, $\overline{X}$, and standard errors, se, calculated on a seven years period from 1993 to 1999). ON = organic nitrogen, OP = organic phosphorus, DIN = dissolved inorganic nitrogen (nitrate plus nitrite plus ammonia), DIP = dissolved inorganic phosphorus. All values are expressed in tons/year.

	ON		OP		DIN		DIP	
	$\overline{X}$	(± se)	$\overline{X}$	(± se)	$\overline{X}$	(± se)	$\overline{X}$	(± se)
Vène	20.6	(8.1)	2.4	(1.0)	28.8	(10.5)	6.6	(2.2)
Pallas	14.9	(8.9)	2.1	(1.3)	17.6	(10.8)	4.4	(2.8)
Nègue-Vaques	4.7	(2.8)	0.8	(0.5)	2.3	(1.5)	0.7	(0.4)
Soupié	4.0	(2.0)	0.7	(0.4)	1.3	(0.8)	0.4	(0.2)
Aygues-Vaques	4.6	(2.5)	0.8	(0.4)	3.3	(1.8)	2.0	(1.0)
Lauze	5.9	(2.6)	1.0	(0.5)	1.9	(0.8)	0.5	(0.2)
Fontanilles	3.3	(2.2)	0.6	(0.4)	0.8	(0.5)	0.3	(0.2)
Mayroual	0.6	(0.4)	0.1	(0.1)	0.1	(0.1)	0.0	(0.0)
Joncas	0.8	(0.4)	0.1	(0.1)	0.2	(0.1)	0.0	(0.0)
Aiguilles	1.0	(0.5)	0.2	(0.1)	0.2	(0.1)	0.1	(0.0)
Total	60.4	(30.4)	8.8	(4.7)	56.5	(27.1)	15.0	(7.1)

Table 6. Simulated annual phytoplankton (Pk) and macrophytes (Mac) productions for years 1996, 1998 and 1999. Dissolved inorganic nitrogen (DIN) and organic nitrogen (ON) river discharges, total annual solar radiations (Qr), total annual precipitation (PCP) and mean air temperature (Temp.) for the three reported years are also shown.

	Unit	1996	1998	1999
Qr	MJ/m^2	5 286	5 699	5 439
Temp.	°C	14.8	14.9	15.3
PCP	mm	1 014	279	967
DIN discharges	$tN\ y^{-1}$	81	8	104
ON discharges	$tN\ y^{-1}$	107	6	97
Mac gross production	$tC\ y^{-1}$	24 344	24 169	26 031
Mac respiration	$tC\ y^{-1}$	24 445	24 140	25 936
Mac net production	$tC\ y^{-1}$	-101	29	95
Pk gross production	$tC\ y^{-1}$	24 859	24 481	27 644
Pk respiration	$tC\ y^{-1}$	7 600	7 408	8 033
Pk net production	t C	17 259	17 073	19 611

Figure 5. Long term simulation for Vène river water flow (A), ammonia (B), nitrates (C) and organic nitrogen (D) daily loads. B, C and D are in log scale, line breaks are due to extremely low values.

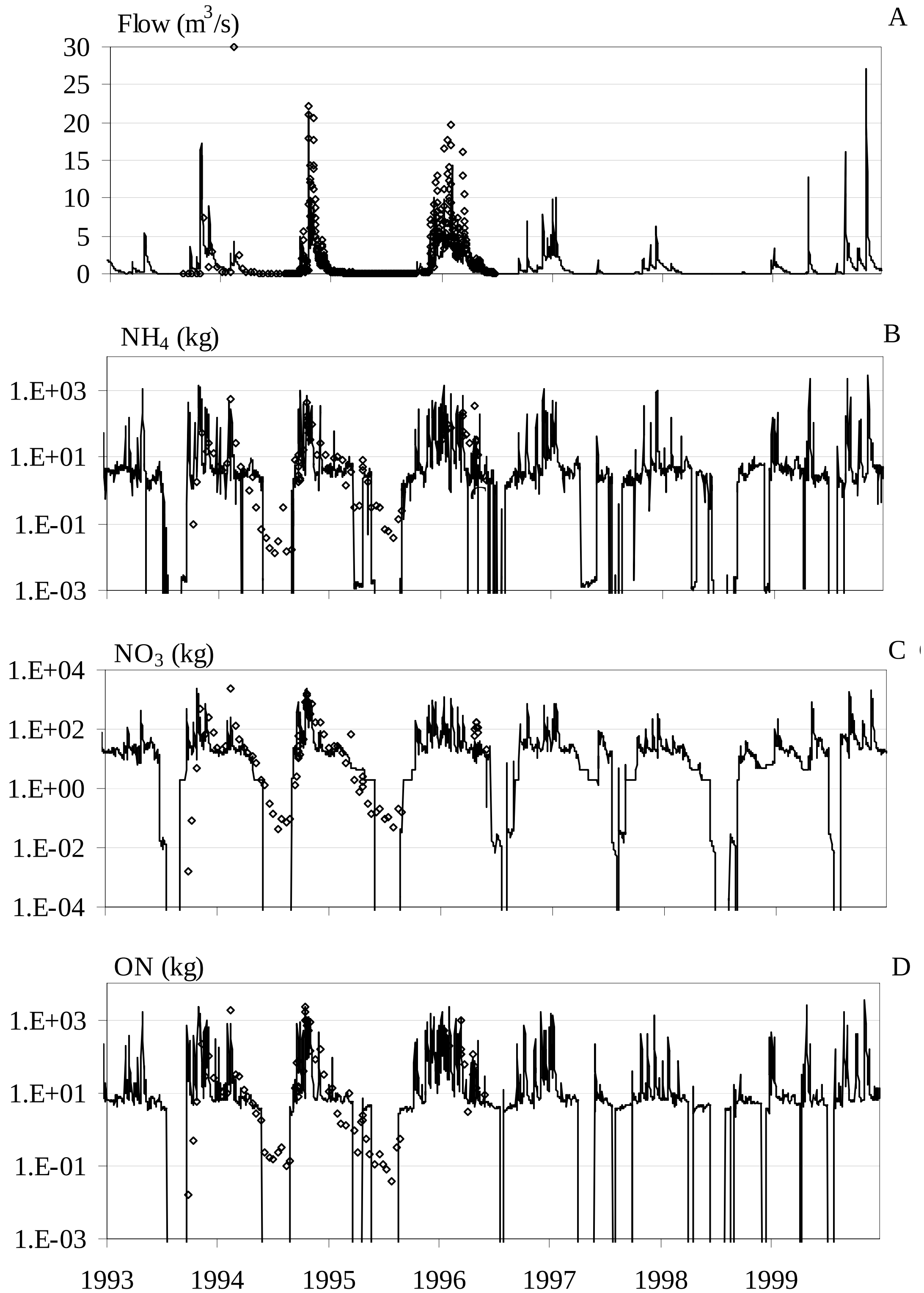


Figure 6. Long term simulation for Pallas river water flow (A), ammonia (B), nitrates (C) and organic nitrogen (D) daily loads. B, C and D are in log scale, line breaks are due to extremely low values.

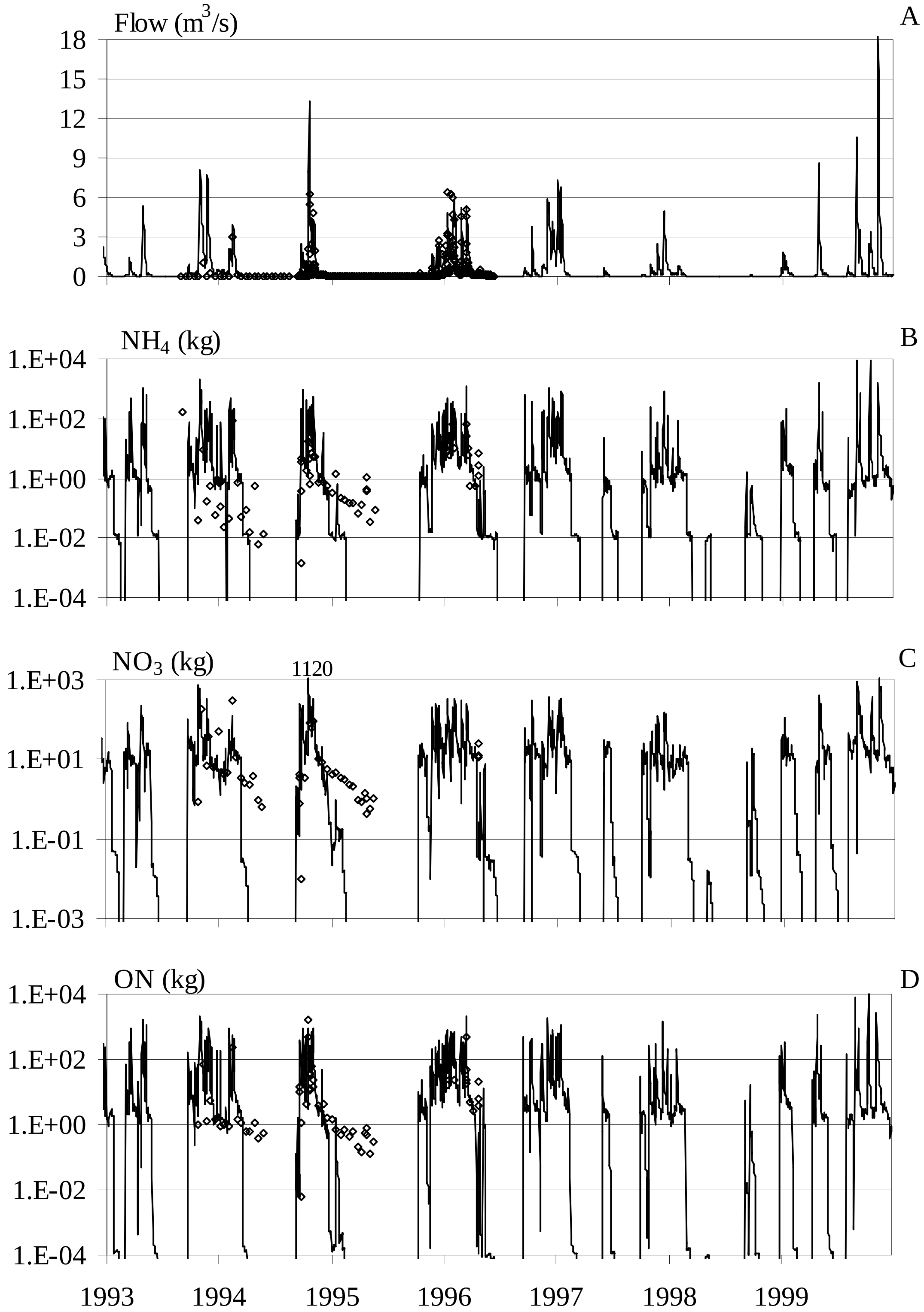
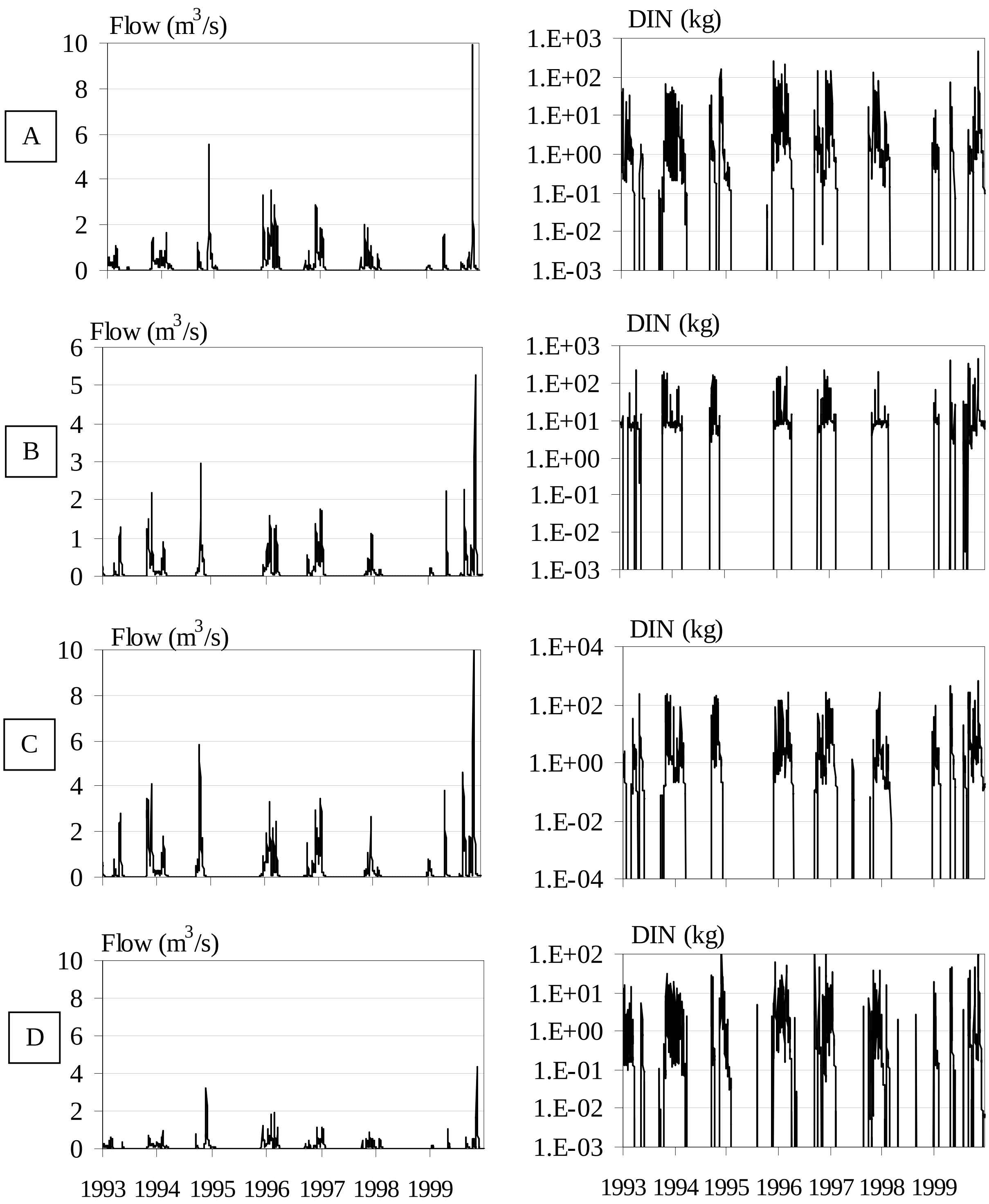
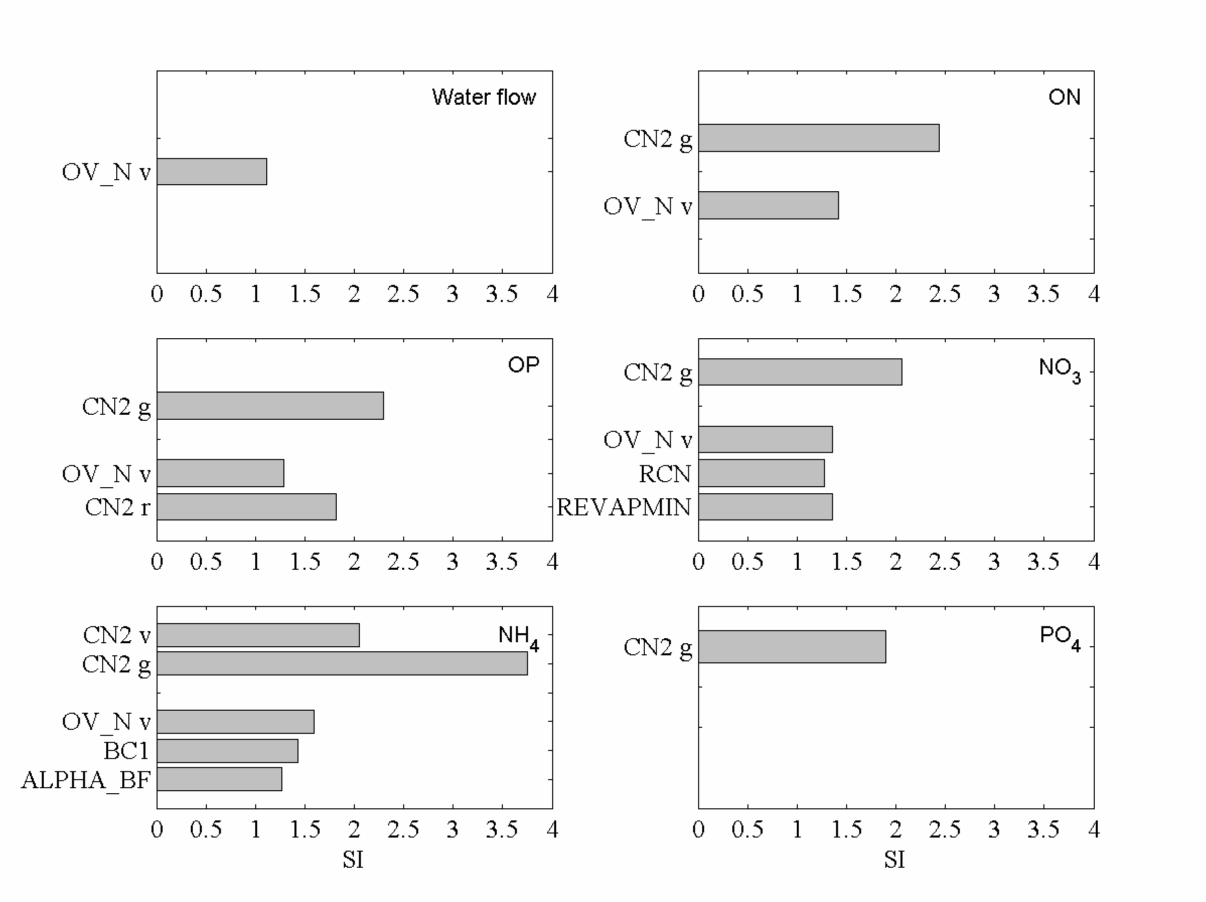


Figure 7. Simulated river flows and dissolved inorganic nitrogen (DIN) daily loads for Soupié (A), Aygues-Vaques (B), Nègue-Vaques (C) and Fontanilles (D) rivers.



The sensitivity analysis enlighten the relatively low sensitivity of the model to parameter variations (figure 8). The ±10% parameter variations never induced more than 4% variation of the variables. DIN variables are more sensitive to parameter variations than simulated water flow or inorganic phosphorus. It appears also that parameter linked to the SCS curve number methodology (CN2) or to the overland flow (OV_N) have a noticeable impact on the overall simulation results. Hence, if better simulations are needed, those parameters would need to be adjusted in priority using experimental studies on garrigue and vineyards, since they represent important surfaces on the Thau watershed.

Figure 8. Sensitivity analysis results for the main variables of the model. Only the parameters inducing more than 1% variability were reported on the graph. The meaning of parameter acronyms is reported in Annex 1.



4.2 The lagoon model

A long-term simulation was run (seven years, from 1993 to 1999) with the aim of catching a wide range of climatologic conditions and compare the total annual simulated phytoplankton and macrophyte productions. The simulation was performed with the outputs of the WM as forcing variables for the LM.

Figures 9 and 10 present the results obtained for two very contrasting years in terms of nitrogen inputs from the watershed : year 1998 was very dry and year 1996, very rainy. Main variables seasonal patterns as well as the evolution of limiting factors are presented for two different diagnostic points (report to figure 1 for location) :

- Station LI (depth 6.3 m, absence of macrophytes) is located in the middle of the lagoon, far from the river influence and with a relatively important depth that explain the absence of macrophytes.
- Station CA (depth 1.8 m, presence of macrophytes : 202 g dw m^{-2}) is located in the Crique de l'Angle, a very shallow area close to the Vène river outlet.

Figure 9. Simulated nutrient (NH_4 and NO_3), detrital nitrogen (Ndet) and chlorophyll *a* (Chl *a*) concentrations at LI (bottom) and CA (top) stations (surface waters).

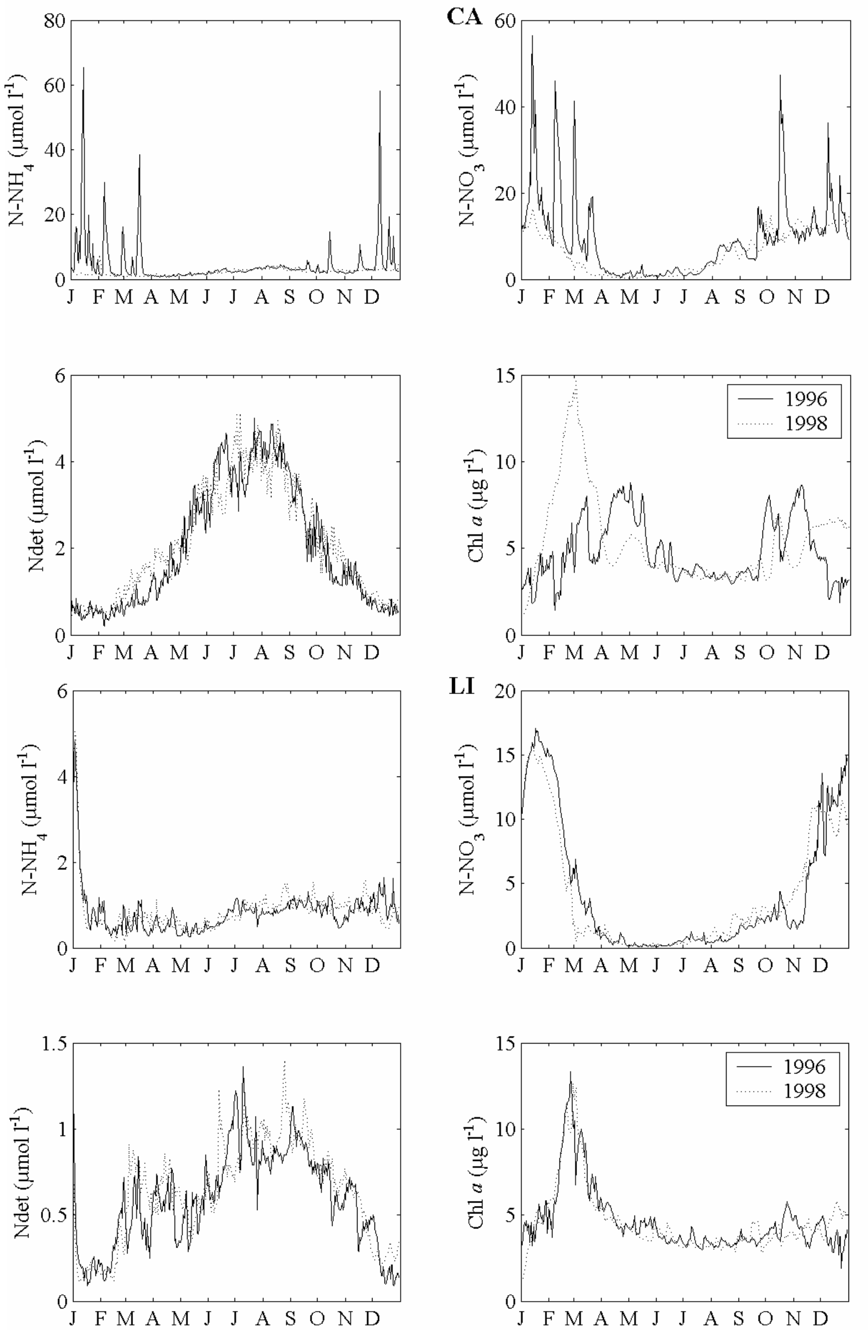
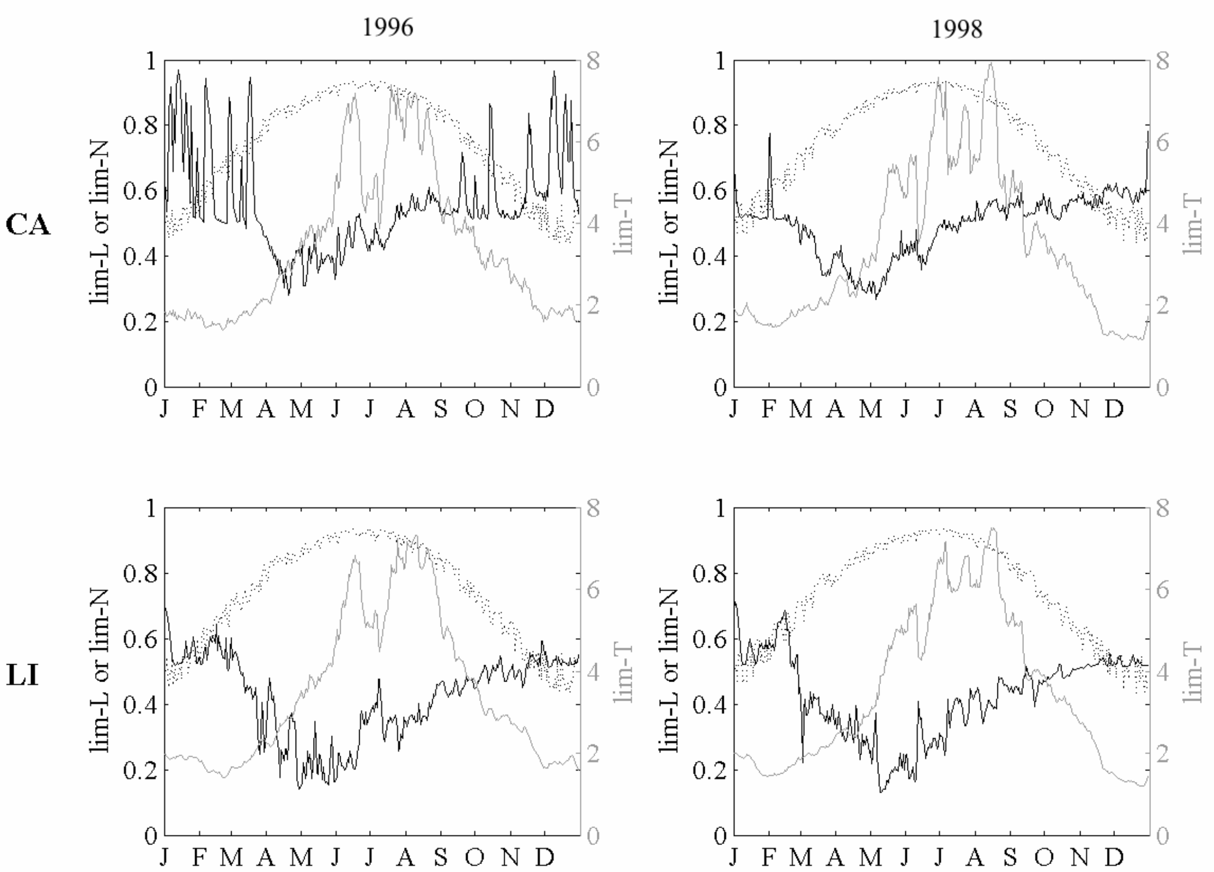


Figure 10. Simulated light, temperature and nitrogen limitations (respectively lim-L, dotted line, lim-T, grey line, and lim-N, black line) for phytoplankton growth during years 1996 (left) and 1998 (right) and at station CA (top) and LI (bottom).



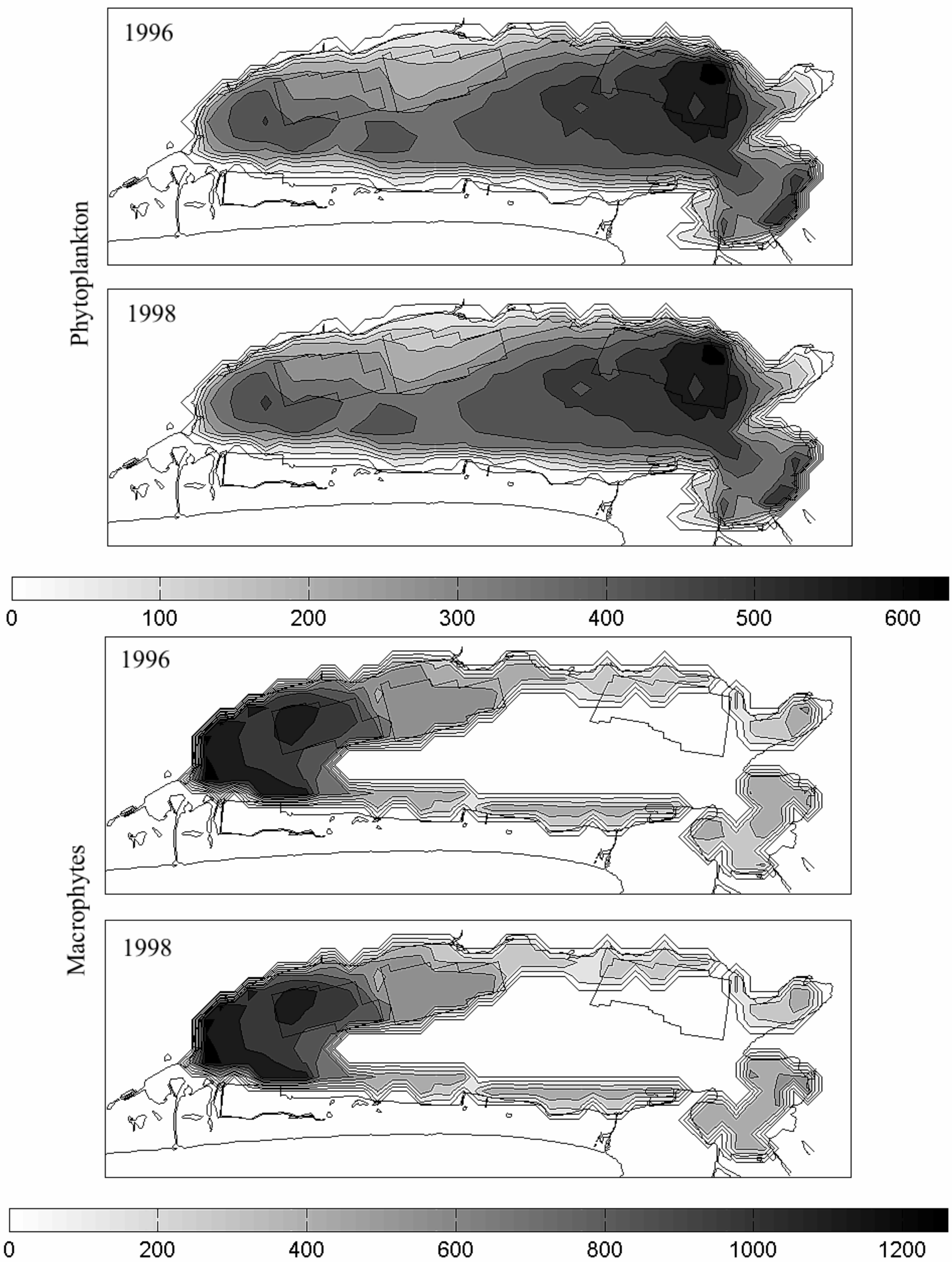
Stations LI and CA resulted very contrasted in terms of DIN concentrations. As expected, the influence of the main river flowing into the lagoon is very clear at station CA, with sudden pikes reaching c.a. 60 µmol l^{-1} concomitantly to river flooding periods. On the contrary, simulated concentrations remain always below 17 and 5 µmol l^{-1} for nitrates and ammonia respectively, at station LI. Furthermore, simulated NO_3 shows a clear seasonal pattern with high winter concentrations ranging 10-58 µmol l^{-1}, and concentrations remaining below 5 µmol l^{-1} during summer months. The influence of mineralization processes on DIN is clearer at station CA with an ammonia summer increase up to 5.8 µmol l^{-1} in 1996 (respectively 4.2 in 1998) while maximum summer NH_4 concentration remains below 1.5 µmol l^{-1} at station LI.

The simulated Ndet concentrations exhibit an unimodal seasonal pattern, with low values between November and April and higher values during summer months. This same general pattern was obtained whatever the considered year but simulations show different responses among the stations : detritus concentrations are higher near the Vène river outlet than at station LI (mean summer concentrations are 5.1 and 1.4 µmol l^{-1} respectively).

Simulated chlorophyll *a* concentrations range 1 – 15 µg l^{-1}, with maximums calculated in March. This spring phytoplankton bloom leads to a temporary DIN decrease in the water column. Afterwards, the simulated bacterial activity increases during summer and leads to a detritus consumption coupled with an increase of dissolved ammonia concentrations. Consequently, the DIN concentration in the water column becomes high enough to sustain the primary production during summer, with a mean chlorophyll *a* concentrations of *c.a.* 3.5 $\mu g.l^{-1}$. Furthermore, probably due to the revival of river discharges in autumn causing higher DIN concentrations, a second phytoplankton bloom was simulated in October – November at station CA. Classically, simulated light, temperature and nitrogen limitations for phytoplankton growth (figure 10) behaved in different ways : during winter light and temperature limitations are dominant whilst the contrary occurs during summer with a stronger nitrogen limitation.

The simulated distribution of annual primary productions (figure 11) resulted very similar from one year to the other, with a mean annual pelagic primary production of 367 ± 141 and 362 ± 143 gC m^{-2} for years 1996 and 1998, respectively. The annual macrophyte total production behaved in the same way as phytoplankton with very similar 1996 and 1998 productions. The simulations showed that maximum productions are due to macrophytes (up to 1300 gC m^{-2} y^{-1}), nonetheless, since macrophytes remain confined in shallow areas at the lagoon periphery, annual phytoplankton production resulted greater at the whole lagoon scale (table 6). Years 1996 and 1999 can be both considered as rainy years, but 1996 was probably more cloudy as shown by the lower incident solar radiation. The mean temperature was also slightly cooler in 1996 than in 1999. Contrasting with these two years, 1998 was dry and very sunny. The maximum phytoplankton production was calculated for year 1999, and the minimum for year 1998, with a net production remaining positive whatever the considered year. On the contrary, the model calculated a positive macrophyte net production for years 1998 and 1999 whilst it resulted negative for 1996.

Figure 11. Simulated spatial distribution of total annual phytoplankton (top) and macrophytes (bottom) gross productions (gC/m^2) for years 1996 and 1998.



5. DISCUSSION

Watershed model

The estimated annual riverine inputs into the Thau lagoon (117 ± 57 t N y^{-1}, 24 ± 12 t P y^{-1}) were found lower than in other studies : 244 t N y^{-1} and 48 t P y^{-1} for Anonymous (1997) and 198 t N y^{-1} and 31 t P y^{-1} according to Pichot et al. (1994). Nevertheless, Anonymous (1997) based their calculations on measurements performed between September 1993 and August 1996, while our estimation takes into account some very dry years as in 1998. This point could explain the discrepancy between those two estimations. Furthermore, Pichot et al. (1994) specify that their values remain "*probably overestimated, ... and should be taken as an order of magnitude*". Then, the values presented in this study, although lower than the mean total annual inputs previously published, remain close to those estimations. Annual freshwater coming from the watershed represent a significant input, 29 % of the lagoon volume ($250 \cdot 10^6$ m^3 according to Amanieu et al., 1989), but remain low comparatively to seawater exchanges through the Sète canals (diurnal exchanges range 0.7 - $35 \cdot 10^6$ m^3 d^{-1}, according to Palomares et al., 1993).

The WM is able to reproduce the general seasonal characteristics of the Thau lagoon watershed, despite of its highly variable character mainly due to the Mediterranean climate. Nonetheless, the statistical comparison performed between simulations and observations put into light some discrepancies concerning water flows and ammonium discharges specially during flood events. Further calibration could be performed to increase the simulations quality but the linear extrapolations we have done to transform observed instantaneous fluxes into daily loads is the source of inevitable errors. Then, in any case, the need for further measurements on the Thau watershed rivers is patent. We think nevertheless, that the main objective of this study has been reached, namely to provide the LM with suitable, even if not fully realistic, daily river flows and nutrient fluxes, over a long period of time. However, the following points, which underline some of the weaknesses of the model, have been raised, and will have to be improved if more accurate simulations are needed.

First of all, the simulations have pointed out the great impact of point sources on the streams water quality. Indeed, the in-stream natural epuration, i. e. mineralization of organic matter and algal growth, is extremely reduced in our case study because of the small size of the streams and their episodic character. Then, the organic matter and nutrient fluxes at the river outlet are directly dependent on the point sources discharges. This dependence obviously increases as the distance between the point source and the river outlet decreases. The point sources located on the Thau catchment area are mainly wastewater treatment plants that collect urban wastewaters and eventually winegrowers cooperative extra-loads. In the model, the point sources discharges are calculated using the removal efficiency factor (RE), which has been, as a simplification, considered as a constant value over the whole studied period. However, if RE can be considered in a first approximation as constant for the plants using activated sludge, for the lagooning treatment plants, RE varies seasonally as well as inter-annually, due to the progressive lagoon sifting (Picot et al., 1992, La Jeunesse, 2001). Generally speaking, point sources appeared to be a key factor in the model and some efforts should be directed on that particular point in order to improve the simulations. We would also like to remember that the main crop (vineyards) is considered as non fertilized in the model, following the survey made by the Chambre d'Agriculture de l'Hérault (2000 a and b), and hence that the relative importance of point sources on river nutrient discharges is higher. This would obviously have to be reviewed in case of change in crop surfaces or modification of fertilization processes.

Secondly, this WM is obviously not able to simulate the very short-term events (a few hours) that can happen under Mediterranean climatic conditions, and can impact highly the coastal ecosystem dynamics (Arhonditsis et al., 2002). The complex erosion mechanisms occurring during flood events is computed in the model using the Modified Universal Soil Loss Equation (Williams, 1975), that considers the properties of soils, the intensity of rainfalls and the land cover and soil management. The parameters used for this calculation should be reconsidered if some more accurate simulated values are needed. Some further measurements would nonetheless be necessary for a better calibration.

Finally, some discrepancies between the model time step and biogeochemical processes have been raised. For example, during the driest periods of the year, very low flow and high temperatures, increase the biochemical reactions' speed, the latter becoming incompatible with a daily time step and leads to temporal desoxygenation of the water and predominance of reduced forms of nitrogen (nil oxygen concentrations and high nitrite concentrations, data not shown). SWAT was not developed with as main objective the modelling of little watersheds with small reaches and, even if the results presented in this study remain satisfactory, water quality parameters and in-stream nutrient processes routines could be modified in order to cope with the problems raised by little streams with intermittent flows. Such modifications would be evidently limited by parameter availability and accuracy.

Lagoon model

Simulated DIN and chlorophyll *a* concentrations in the water column are within the range usually reported in the literature (Picot et al., 1990 ; Pichot et al., 1994, Souchu et al., 1998). The calculated phytoplankton gross production is also close to the mean value estimated by Vaquer and published in 2000 by Chapelle et al. (400 gC m^{-2}). Unfortunately, the simulated macrophyte production could not be validated since, to our knowledge, no estimation of the macrophyte production at the whole lagoon scale is available in the literature. Following our results, the seasonal phytoplankton dynamics in the Thau lagoon is mainly controlled by light and nitrogen. The simulated main phytoplankton bloom occurs in March, in spite of still low water temperature during this period, when DIN concentrations are high and incident light levels become sufficient. During summer lower nitrogen concentrations limit the phytoplankton growth and conversely during winter light and temperature limitations are predominant.

The model allowed moreover to compare calculated primary productivities under the influence of very different watershed inputs and climatologic conditions. For the simulated period, the model calculates an important annual net production for phytoplankton, and a respiration roughly balancing the production for macrophytes. Surprisingly, minimum net macrophyte production was simulated for year 1996, despite of important nitrogen watershed inputs. In fact, year 1996 was also characterised by a low total annual solar radiations in comparison with 1998 or 1999. Hence, this suggests that incident light may play a more important role than nitrogen riverine inputs in the control of macrophytes total annual production. Then, phytoplankton blooms, inducing lower water transparencies, have an indirect negative effect on macrophyte productions, and play probably a determining role in the macrophyte maximum depth distribution. Coming back to the nutrients, for the macrophyte populations, the vicinity with sediment as well as with the important detritus concentrations laying at the water column bottom could explain their possibility to easily derive benefit from the mineralization processes and be less dependent from the river discharges. Actually, these assumptions is confirmed by the spatial distribution of macrophytes in the Thau lagoon : maximum macrophyte biomasses were recorded in the south-western part of the lagoon, far from the main rivers influence (Gerbal and Verlaque, 1995, Plus et al., 2003 a). These preliminary results are nevertheless to be taken with precautions because, as we mentioned before, the macrophyte biomass is considered as a forcing function in the model, and its seasonal variation depends only on the time. A possible improvement of the LM could be the implementation of macrophytes as real state variables.

Phytoplankton growth and nutrient uptake characteristics allow a quick response to sudden DIN pikes in case of temporary river floods. The model simulates such temporary phytoplankton blooms, by they remain confined at the proximity of the river outlets (*e. g.* station LI was not directly impacted by river flood periods). Hence, extrapolating to the whole lagoon scale and on an annual basis, our results also state that the huge differences in nitrogen discharges that can occur from one year to another does not systematically imply high variations in total annual lagoon phytoplankton productivity. Actually, considering a C:N molar ratio of 106:16 for phytoplankton (Redfield, 1963), the model calculates that riverine inputs can only meet for 0.5 (in 1998) to 6.2 % (in 1996) of annual phytoplankton nitrogen requirements. Thus, and even if some other external nitrogen sources, *i. e.* direct rain and bacterial nitrogen fixation, are not taken into account in the actual model, the important lagoon production might be largely supported by regenerated nutrients coming from mineralization processes, animal excretion and nutrient exchanges between the sediment and the water column. Our results are in agreement with Mazouni (2004) who recently showed that, within the oyster culture of the Thau lagoon, the whole nitrogen phytoplankton requirement could be satisfied by nitrogen recycling. Other studies for the Thau lagoon (Pichot et al., 1994, Chapelle et al., 2000) as well as for other Mediterranean coastal zone (Arhonditsis et al., 2000) concluded in the same way.

The fact that the variation of watershed discharges seems to have a little influence on total annual primary production does not absolutely mean that eventual future enhancement of riverine nutrient inputs will not conduct in a modification of the lagoon functioning. We have seen that the north and north-east border of the lagoon, near the river outlets, are for example greatly impacted by nutrient inputs coming from the watershed and that simulated DIN concentrations show very high and short-term variations during the flooding periods. At the present time, this nutrient pattern profit essentially to phytoplankton cells able to quicker responses than macrophytes. A modification in discharges frequency or magnitude might totally change this equilibrium and be beneficial to green opportunistic algae for example. The WM results enlightened moreover the importance of point sources in the watershed discharges pattern. In the actual demographic tendency, more and more people are willing to live near the seashore (temporarily or not), the problem of waste water treatment is then fatally crucial.

This study is a first attempt in coupling a watershed model with a hydrodynamical-ecological three-dimensional model for the Thau lagoon. The WM outcomes were used to force the LM in order to assess the impact of varying climatologic conditions on the lagoon primary productivity and estimate the relative importance of nitrogen sources. We think that such integrated tools might be of real help in the implementation of the European Union Water Framework Directive. A lot of work remain nevertheless to be done before catching accurately the entire complexity of such an ecosystem, quantifying matter fluxes between all its different components and forecasting its evolution under varying anthropological and environmental pressures.

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Annex 1. Values of parameters deriving from the calibration process.

Parameter	Description	Value	Unit
CN2 u	Runoff coef. for urban zone	85	-
CN2 r	Runoff coef. for fallow land	77	-
USLE_P v	USLE support practice coef. for vineyard	0.15	-
" g	USLE support practice coef. for garrigue	0.1	-
" r	USLE support practice coef. for fallow	0.1	-
" u	USLE support practice coef. for urban zone	0.1	-
OV_N r	Manning's overland flow coef. for fallow	1.5	-
ALPHA_BF	Base flow recession constant	0.2	d
GW_DELAY	Delay time for ground-water recharge	20* and 2**	d
GW_REVAP	Revap. (evapotranspiration) coefficient	0.1	-
REVAPMIN	Threshold water level for revap.	0	mm
CH_K2	Effective hydraulic conductivity (streams)	150	mm/h
SURLAG	Surface runoff lag time	1	d
RCN	Nitrogen in rain	0.669	mg/l
BC1	Cst. rate for $NH_4 \rightarrow NO_2$	1	d^{-1}
BC2	Cst. rate for $NO_2 \rightarrow NO_3$	2	d^{-1}
RK1	CBOD (Carbonaceous biological oxygen demand) desoxygenation rate	0.02	d^{-1}
RK2	Oxygen reaeration rate	1	d^{-1}
RK4	Benthic oxygen demand rate at 20°C	0	$mg/m^2/d$
RS1	Algal settling rate	0.15	m/d
AI1	Nitrogen fraction in algae	0.09	mg/mg
AI2	Phosphorus fraction in algae	0.02	mg/mg
AI5	O_2 uptake/NH_3 oxidation	3	mg/mg
AI6	O_2 uptake/NO_2 oxidation	1	mg/mg
TFACT	Photosynth. active radiation	0.5	-
P_N	Algal preference for NH_4	1	-

MUMAX	Algal maximum growth rate	3	d^{-1}
K_L	Light limitation coef. for algae	0.223	$kJ/m^2/mn$
K_N	N limitation coef. for algae	0.01	mg/l
K_P	P limitation coef. for algae	0.001	mg/l

* Vène river ; ** Pallas river