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Model-based assessment of shading effect by riparian vegetation on river water quality

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ABSTRACT

Shading by riparian vegetation affects incident solar radiation and water temperature in small- and moderate-size streams, and is thus an important component in the influence of forested riparian buffers on streams. The water quality effects of riparian shading are largely unknown. A simulation study was carried out to evaluate the effect of shading on six water quality variables in a moderate-size Belgian river stretch. A dynamic modelling approach making use of the River Water Quality Model No. 1 was chosen to represent the system. The scenarios developed indicate that shading may be an effective tool in controlling stream eutrophication (44% reduction in phytoplankton productivity in the simulated stretch) but has a limited effect on dissolved oxygen, chemical oxygen demand, nitrates, ammonium nitrogen, and phosphates. Results suggest that shading can effectively be implemented as a direct management strategy to improve water quality conditions in small and moderate-size watercourses that are exposed to excessive algal growth during summer periods.

1. Introduction

The core issue of the European Water Framework Directive (Council Directive 2000/60/EC establishing a Framework for Community action in the field of water policy) is the achievement of a good ecological and chemical status in European waters. This water quality objective is to be achieved by means of an innovative approach that combines limits on emissions from pollution sources with quality standards to be achieved in the receiving natural water bodies. As a consequence of this approach, new strategies will become available to the water quality managers. Interventions will not necessarily be limited to the sources of pollution but may also include direct measures in the receiving watercourses to contribute to the achievement of the required water quality standards.

Restoration of forested riparian buffers is one of such direct watercourse management strategies and has been advocated as a best management practice for controlling water quality of surface streams (US EPA, 2005). The important role played by riparian vegetation in retaining nitrogen (Anbumozhi et al., 2005; Dodds and Oakes, 2006; Ensign and Mallin, 2001; Hefting et al., 2005; Mander et al., 1997; Vought et al., 1995) and phosphorus inputs (Borin et al., 2004; Ensign and Mallin, 2001; Mander et al., 1997; Meals, 2001; Vought et al., 1995) from upland diffuse and point sources of pollution is well documented and generally acknowledged (Muscutt et al., 1993).

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Filtration by vegetated riparian buffers significantly abates the concentration of other relevant water quality indicators and contaminants, such as suspended solids (Ensign and Mallin, 2001; McKergow et al., 2003), chloride (Anbumozhi et al., 2005), and pesticides (Borin et al., 2004; Lin et al., 2002; Muscutt et al., 1993; Sweeney et al., 2004). Furthermore, it may play an important role in controlling the microbiological quality of the stream water (Ensign and Mallin, 2001; Meals, 2001).

Buffer vegetation strips are important determinants of both stream ecology and morphology. They represent sources of particulate organic matter such as litter and wood debris (Casatti et al., 2006; Collier and Halliday, 2000) and terrestrial prey for carnivorous fish (Edwards and Huryn, 1996). Furthermore, well-established riparian vegetation contributes to control erosion processes and stabilize stream banks (Boothroyd et al., 2004; McKergow et al., 2003). Newly afforested streams may, however, temporarily display higher bank erosion (Quinn et al., 1997), which can result, in the long-term, in widening of the stream channel (Sweeney et al., 2004).

Shading of the water column by riparian vegetation represents an important component of forested riparian buffers, as it potentially affects two key parameters for stream water quality processes: the water temperature and the intensity of the incident light (Beschta et al., 1987; Collier et al., 2001; Poole and Berman, 2001; Quinn et al., 1997).

A reduction in water temperature influences both chemical and biological processes. A lower temperature increases the saturation concentration of dissolved oxygen and affects the growth and development rates of aquatic organisms, thus resulting in smaller risk for dissolved oxygen depletion and increased capacity of a stream to assimilate organic wastes. Lower water temperature may mitigate the impact of thermal pollution, which is currently experienced in many urban and peri-urban contexts (Eaton et al., 1995; Kinouchi et al., 2006). Boothroyd et al. (2004) observed that stream temperature in selected harvested river stretches in New Zealand were in the range expected to cause mortality for a variety of common fish and invertebrate species in mid-summer, whereas water temperature at shaded sites was just below this level.

Besides lowering the water temperature, shading reduces the intensity of light that reaches the water surface. Light penetrating the water column and reaching the bed is a control factor for the growth of algae and macrophytes, together with nutrient availability, hydraulic mixing and scouring, and grazing by invertebrates and fish. Light interception by vegetation canopy is likely the main determinant of the observed smaller periphyton biomass (Boothroyd et al., 2004; Ensign and Mallin, 2001; Towns, 1981) and macrophytes growth (Fritz et al., 2004; Kohler et al., 2000; Mander, 1995; Wilcock et al., 2004) in shaded streams with respect to unshaded control sites. As such, it can represent an effective tool to control the growth of algae in streams that are particularly exposed to the risk of eutrophication and reduce flooding risks induced by higher flow resistance in streams that are rich in macrophytes. Other effects of shading may include reduction of the richness of local aquatic fauna and flora species (Dawson and Haslam, 1983; Mander, 2005), accumulation of leaves and organic matter (Dawson and Kern-Hansen, 1979), and shading out of riparian grasses (Collier et al., 2001; Parkyn et al., 2005), which may result in temporarily higher bank erosion and, in the long term, channel widening. The provision of shaded conditions is often a restoration goal for streams where this resource has been lost due to removal of riparian vegetation or reduced retentiveness through channel straightening and wood removal.

The intensity of the shading effect on water temperature and light availability is controlled by a series of factors. Shading is more marked in small and moderate streams with width of 5 m or less (Boothroyd et al., 2004; Quinn et al., 1993; Whitledge et al., 2006) and becomes very small or negligible at channel widths of 10 m or more (Rounds, 2007; Washington Forest Practices Board, 1997; Davies-Colley and Quinn, 1998). Besides stream width, the shading effect is affected by characteristics of the vegetation (i.e., type of vegetation, canopy height, width of the vegetated buffer, foliage density), geographic and climatic aspects (i.e., ambient solar irradiance, sun angle, channel orientation), and optical properties of the water column (i.e., turbidity (Boothroyd et al., 2004) and yellow substance (Davies-Colley and Smith, 2001)).

Despite acknowledging the potential impact of shading on stream water quality, previous research failed to fully characterize the shading effect on common water quality parameters and to clearly separate it from other water-quality-related processes occurring in vegetated riparian buffers, such as nutrient retention.

This paper aims at filling this knowledge gap by quantifying the water quality changes due to riparian shading as induced by a reforestation programme in a moderate-sized river stretch in the Nete river basin in Belgium. A dynamic, model-based assessment of the efficiency of different riparian vegetation management strategies in improving river water quality through shading effect is performed. The effect of shading on primary productivity and other commonly monitored water quality parameters is evaluated.

2. Materials and methods: study site and modelling approach

2.1. *The Nete river basin*

The Nete River basin is located within the Flemish Region of Belgium and belongs to the basin of the Schelde River. It covers a surface of 1673 km^2 of non-calcareous lowland plain. The dominant soil type is sand. The main pressures on surface water quality in the Nete river basin are exerted by households, industrial and agricultural activities. A population of about 600,000 people lived in the river basin at the end of 1997. At that time, only 58% of the population was connected to sewerage with treatment. Since then, the degree of connection has significantly increased, thanks to the investment programmes of the Flemish Region that resulted in an expansion of the regional collector system on one hand and of the municipal sewage systems on the other. Some important industrial sites are situated within the river basin. In particular, heavy pollution from industrial effluents caused a serious degradation of the ecological status of the tributary Grote Laak. In the last years, the trend in the Nete basin has been to disconnect the largest industries from public WWTPs and require

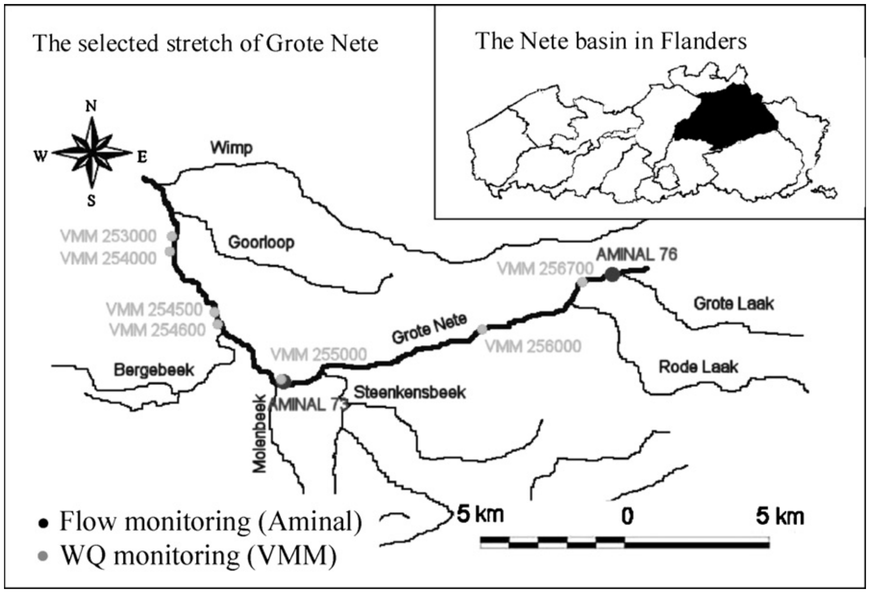

Fig. 1 – The Nete River basin in Flanders, the selected river stretch, and the location of water quantity and quality monitoring stations.

local wastewater treatment prior to discharge into the surface water. Agricultural land-use contributes significantly to the total amount of nutrients entering the water system of the Nete River since agriculture represents the main land-use in the basin (Benedetti et al., 2006).

The main watercourses within the river basin are the Grote Nete and the Kleine Nete. The Grote Nete is 85 km long, its average width is 12 m, and the total surface of its catchment is 730 km^2. The geomorphology of the Grote Nete was heavily modified by human intervention. Its natural tendency to meander was largely limited by canalization and embankments that took place in the 1970s. Nowadays the biological diversity in large part of the river floodplains is strongly reduced.

For this study, a stretch halfway upstream of the Grote Nete, with a length of 19.5 km was selected (see Fig. 1). The boundaries of the stretch are the confluence with the Grote Laak (upstream) and the Wimp (downstream). The average water depth and yearly flow measured in 2003 in proximity of the upstream section were 0.6 m and 6.2 m^3/s respectively (Hydronet, 2008). The minimum monthly average depth was measured in April (0.37 m) and the maximum in January (0.95 m). Four wastewater treatment plants (WWTPs) are located along the stretch. Heavy industrial pollution affects the Grote Laak and other sections of the Grote Nete upstream of the selected stretch. Despite a reduction in recent years in the emissions of heavy metals such as cadmium and zinc, accumulation in the sediments was observed at several locations (Reynders et al., 2008) and their concentrations in the water are still close or above the Belgian water quality objectives. Metal pollution negatively affects fish and macrophytes diversity (Meire et al., 2007) and results in accumulation in fish tissues (Bervoets et al., 2005). A field excursion in May 2004 did not reveal the presence of an abundant vegetation of emergent or free-floating macrophytes in the selected stretch. In the same occasion it was observed that the currently available shading of riparian trees along the selected stretch is localised and very limited.

The physical-chemical status and the water flow in surface waters in Flanders are monitored respectively by the Flemish Environmental Agency (VMM) and the Environment, Nature, Land, and Water Management Administration (AMINAL) through a well-developed network of measuring stations. A range of water quality parameters—i.e., dissolved oxygen (DO), chemical and biochemical oxygen demand (COD, BOD), pH, nutrients, heavy metals, and pesticides—is monitored. Fig. 1 illustrates the location of the seven VMM water quality measurement stations and two AMINAL water flow monitoring stations located along the stretch selected for this study. The average concentrations (and standard deviations) of COD and suspended solids measured in 2002 at VMM station 253000—i.e., in proximity of the downstream section of the selected stretch—were 26.3 mg/L (±9.2 mg/L; $N = 18$) and 20.6 mg/L (±15.2 mg/L; $N = 18$) respectively. The pH in the same period was relatively constant (7.4 ± 0.3; $N = 19$). The concentration of DO and chlorofill *a* varied in the range 5–11.6 mg/L ($N = 17$) and 1–7 μg/L ($N = 5$) respectively. The transparency of the water as measured with the Secchi method on August 30 was 0.8 m.

The physical-chemical water quality in the Nete was evaluated in the General Water Quality Plan (Paelickx et al., 2001) by means of the Oxygen-Prati Index (Prati et al., 1971), which is based on measurements of DO. Water is reported to be “polluted” or “moderately polluted” in the stretch selected for this study. Oxygen shortage due to summer planktonic algal blooms is reported as one of the water quality issues in the selected stretch of the Grote Nete. The biological water quality in the Nete is evaluated by VMM on the basis of the Belgian Biotic Index (De Pauw and Vanhooren, 1983). This index is based on the presence of freshwater macro-invertebrates in the water. The General Water Quality Plan of the Nete basin reports a “poor” and “fair” biological water quality in different subsections of the selected stretch.

2.2. The water quantity model

Since water flow measurements are available with hourly frequency in two locations along the selected stretch of the Grote Nete, no hydrologic model was specifically developed for this study. The flow rate at the upstream section of the stretch—i.e., in correspondence of AMINAL measurement station 76 in Fig. 1—is taken as input for the simulation. The hydraulic behaviour of the Nete River in the selected stretch is modelled with a cascade of tanks-in-series. The contribution of the base-flow was estimated—based on expert judgement—equal to 10% of the water volume in each tank. The rating curve available for the upstream flow measurement station is assumed to be applicable to all tanks since topography and morphology are uniform in the river stretch. The limits in the applicability of the tanks-in-series approach, including the difficulty to take into account backwater effects (Solvi et al., 2005), were of little relevance in the present study.

2.3. The biochemical model

A biochemical model of the selected stretch was implemented in the modelling and simulation platform WEST® (MOSTforWATER NV, Kortrijk, Belgium) (Vanhooren et al., 2003) using a simplified version of the River Water Quality Model No. 1 (RWQM1) (Reichert et al., 2001; Vanrolleghem et al., 2001).

The complete RWQM1 contains a large number of state variables (24) and processes (23) that assure the applicability of the model to a very large number of water quality modelling situations. In practice, however, few modelling conditions require the implementation of the full model and there is rather a need to identify the simplest model that is adequate to the problem at hand without involving spurious detail complexity (Reichert et al., 2001).

As such, the implemented RWQM1 sub-model does not include pH-dependent reactions, the state variable "consumers" and the connected processes. Although some modelling studies have shown that macrophyte photosynthesis may have a strong influence on pH (Simonsen and Harremoes, 1978), neglecting pH-related processes appears justified in the present study by the observed low abundance of macrophytes, low variability shown by pH measurements in all stations during the entire simulation period (pH ranging between 6.8 and 7.8), and small impact of riparian vegetation on pH, as highlighted by several studies (Blackburn and Wood, 1990; Ensign and Mallin, 2001). The same RWQM1 sub-model used in this study has been successfully tested on a South African basin (Deksissa et al., 2004) and on an Italian basin (Benedetti et al., 2007). Table 1 summarizes state variables and processes included in the biochemical model. The biochemical model used in this study includes 16 state variables and 16 processes. A detailed description of the state variables and the governing equations of the water quality processes is given in Reichert et al. (2001).

In the model only one class of algae is introduced, namely phytoplanktonic algae. The growth of algae is assumed to depend on three main factors: light intensity, water temperature, and nutrients concentration. Following Steele (1962), the dependence of algal growth on light intensity is characterized by direct proportionality up to a level of optimum irradiance beyond which an inhibitory effect occurs. Light attenuation due to background mineral turbidity and algal self-shading is not accounted for in the RWQM1 model as reduced water depths and vigorous mixing are assumed to minimize the spatial heterogeneity of light distribution over depth.

Table 1 – State variables and processes included in the biochemical model.

State variable		Process
Symbol	Description	
S_I	Inert soluble COD	Aerobic growth of heterotrophs with ammonia
S_S	Readily biodegradable COD	Aerobic growth of heterotrophs with nitrate
S_{O_2}	Dissolved oxygen	Anoxic growth of heterotrophs with nitrate
S_{NO_2}	Nitrite nitrogen	Anoxic growth of heterotrophs with nitrite
S_{NO_3}	Nitrate nitrogen	Aerobic endogenous respiration of heterotrophs
S_{PO}	Phosphate phosphorus	Anoxic endogenous respiration of heterotrophs
S_{NH}	Ammonia nitrogen	Growth of 1st stage nitrifiers
S_{ALK}	Alkalinity	Aerobic endogenous respiration of 1st stage nitrifiers
X_I	Particulate inert COD	Growth of 2nd stage nitrifiers
X_S	Particulate organic matter	Aerobic endogenous respiration of 2nd stage nitrifiers
X_H	Heterotrophic biomass	Growth of algae with ammonia
X_{N1}	First stage nitrifying bacteria	Growth of algae with nitrate
X_{N2}	Second stage nitrifying bacteria	Aerobic respiration of algae
X_{ALG}	Algae	Death of algae
X_P	Phosphate adsorbed to particles	Hydrolysis
X_{II}	Particulate inorganic matter	Aeration

The dependence of the algal growth kinetics on water temperature is modelled by means of the following exponential function:

$$K_T = K_{20^\circ C}\, e^{\beta(T-20)}$$

which relates the reaction constant K at a temperature T to its value at the reference temperature of 20 °C through the constant β. The default value for β in the RWQM1 is 0.046 °C^{-1}. The optimum level of irradiance is not assumed to depend on the water temperature (Renk et al., 2000; Michio et al., 1995).

The effect of nutrients concentration on algal growth is expressed by means of Michaelis–Menten equations. Two distinct sub-processes are implemented in the model to describe growth with ammonia and nitrate as nitrogen source. In both sub-processes, the dependence on phosphate concentration is included as a limitation term in the Michaelis–Menten equation.

2.4. The shading effect model

The shading effect of riparian vegetation is simulated in the model by a reduction of the light intensity reaching the stream, which, on one hand, directly affects the growth of algae and, on the other hand, influences the water temperature through a heat balance model.

The reduction of light intensity in shaded conditions was estimated by means of direct measurements during field excursions. Global solar radiation was measured during summertime (May 2004), at different times of the day (11:00 a.m., 02:20 p.m., and 06:00 p.m.) and in different shading conditions. Measurements were made in the shade of tree species that are commonly found on river banks in the Nete River basin—i.e., common alder (*Alnus glutinosa*), willow (*Salix* spp.), and poplar (*Populus nigra*)—and are indicated as best suited for river banks reforestation projects by the Ministry of the Flemish Community of Belgium (2000).

Measurements indicate a degree of spatial variation in the solar radiation filtering through the vegetation due to the structure of the canopy. While in open-space the highest measured light intensity was equal to about 4000 mV, in the shade of a fully developed tree light intensity decreased by more than 90% (360 mV). Where sunlight penetrates through small openings in the canopy ("partial shade") intermediate conditions occur: light intensity is reduced less than in fully shaded spots (highest intensity equal to 900 mV, 78% reduction) and shows little variations in intensity during day-time. Such values are well within the range indicated in the literature for small and moderate size streams. Hill et al. (1995) argue that the shade of a fully developed riparian tree canopy can block up to 95% of the incoming radiation in small streams. Boothroyd et al. (2004) measured a 97.6% decrease in the median diffuse non-interceptance lighting at the water level for small (<5.0 m width) streams in New Zealand. In the same study, an 80% decrease was measured in slightly larger streams (width ranging between 5.0 m and 11.2 m). Caissie et al. (2007) predicted a shading factor due to riparian forest cover and site topography equal to 55% for a 9-m wide stream and equal to 8% for an 80-m-wide river in Canada.

The shading effect was simulated in the model based on the reduction factor measured in conditions of partial shade, i.e., a reduction of 78% of the solar radiation reaching the water column was applied. Measurements of the solar radiation in open-space and with hourly frequency were available for the whole simulation period from the nearby meteorological station in Ukkel. The shading effect on solar radiation was modelled by applying a reduction factor to the open-space radiation as measured in Ukkel in order to derive the dynamic input for the model. The shading effect of the currently scarce riparian vegetation was neglected in both model calibration and scenario analysis and the orientation of the channel was not assumed to influence the shading effect. No decrease in light intensity with the depth is simulated in the model due to the shallow depth of the Grote Nete in the selected stretch.

The effect of shading on water temperature was derived by means of a heat balance model based on the model by Talati and Stenstrom (1990). The heat balance model includes the effects of solar radiation, atmospheric radiation, surface evaporation, and surface convection as a function of radiation intensity, air temperature, wind speed, relative humidity, and water surface of the river (for each of the tanks). In addition, the contribution of the base-flow to the heat balance was accounted for, assuming a groundwater temperature equal to 11 °C. This temperature represents the yearly average in the Nete river basin, as reported in the measurement database in Flanders (DOV, 2005).

The reduction in the daily maximum temperature and the daily average temperature as resulting from the heat balance model, based on light intensity as occurring in conditions of partial shade, amounted to 7 °C and 3 °C respectively. Such values are consistent with previous research findings. Quinn et al. (1992) report a 4–10 °C lower maximum water temperature for small watercourses in New Zealand. In an experimental study involving artificial shading of a small Canadian stream, Johnson (2004) observed a 4–5 °C reduction in maximum water temperature but unchanged minimum and mean temperature values. Lovett and Price (1999) report a 3–5 °C lower mean temperature and three times lower daily temperature fluctuations for forested stream in comparison to cleared stream sites in Australia. Boothroyd et al. (2004) found a 2.7–3.2 °C lower mean water temperature in small shaded streams in New Zealand in comparison to recently harvested sites. In the same study also differences in the diurnal range of variability of temperature are reported (3–4 °C vs. 10–12 °C). Binkley and Brown (1993) in a review of North American studies, report 2–6 °C higher temperatures in harvested sites compared to streams where riparian vegetation is retained. Brosofske et al. (1997) found no relationship between width of riparian buffer vegetation and stream temperature and concluded that, in the analysed small streams in Western Washington, the latter was mainly affected by soil temperature in the riparian zone.

2.5. Simulation period and other input data

The warm season of year 2002 is taken as simulation period. The model is calibrated based on the water quality measurements available between April 1 and October 31. Since the growth of leaves and their abscission may affect the shading effect and the organic matter dynamics during the early and late parts of this period, only the dynamics occurring between May 1 and September 30 are investigated in the shading scenario analysis. Due to reduced foliage density, absence of algal blooms, and lower radiation intensity, other seasons were considered to be of less significance for the objectives of the study.

The flow rate from the upstream section of the stretch during the simulation period is available from an automated measurement station. To calibrate the model, the flow in a section 12.2 km downstream the initial section—i.e., monitoring station AMINAL 73 in Fig. 1—as well as water quality measurements collected by VMM at the seven measurement locations along the stretch were used.

Four sources of pollution are considered in the model: agriculture, untreated industrial effluents, untreated households effluents, and effluents of WWTPs. VMM provided data on both pollution from agricultural land use and untreated household effluents. The impact of diffuse nutrient pollution from agricultural land-use has been estimated by VMM by means of the SENTWA model (System for the Evaluation of Nutrient Transport to Water), which takes into account the partial losses through atmospheric deposition, the losses into groundwater, the direct impact of mineral and organic fertilisers, the effect of natural drainage, erosion, and run-off (VMM, 2004). Data concerning the number of households not connected to sewage treatment for each of the hydrographic zones of the Nete basin were similarly provided by VMM. For untreated domestic effluent, pollutant loads were estimated on the basis of loads per inhabitant per day that are typical of the Flemish region of Belgium (60 g/day of BOD; 110 g/day of COD; 90 g/day of suspended solids; 12 g/day of total nitrogen; 2 g/day of total phosphorus). Data concerning industrial effluents and effluents from WWTPs were provided respectively by VMM and Aquafin NV, the latter being responsible for the construction and operation of sewage collection and treatment systems in the Flanders region.

The available frequency of measurement of the input data varies significantly. Hourly measurements were available for flow, water level, temperature, and solar radiation. WWTP effluent flow is measured daily, water quality parameters in the WWTP effluents twice per week, while nutrient loads from agriculture and water quality parameters in the monitoring stations were available with a monthly frequency. For untreated households and industrial effluents a steady state value was assumed during the whole simulation period, due to lack of more detailed information. The input file for the model was created with hourly time-steps. The loads of pollutants were assumed to remain constant between two measurements.

2.6. *Sensitivity analysis*

Due to the limited amount of data available for model calibration, a global sensitivity analysis was undertaken in order to identify the most influential parameters of the water quality model. This helped in reducing to a minimum the number of parameters whose values were manually adjusted in the calibrated model.

The method used for the sensitivity analysis is a regression and correlation technique (Saltelli et al., 2000) with Latin Hypercube Monte Carlo sampling (McKay, 1995). The regression was performed between the model parameters under consideration and two critical outputs of the model, namely the amount of hours that the DO concentration drops below the critical threshold of 5 mg/L and the average DO concentration. The choice of the critical model outputs was motivated by the relatively frequent oxygen shortages due to planktonic algal blooms experienced in the selected stretch of the Grote Nete.

The mean, minimum, and maximum values of the parameters used in the Monte Carlo sampling are given in Table 2 together with the assumed variation and distribution. The mean value correspond to the parameter values calibrated by Reichert et al. (2001). The range of variability of the parameter values is assumed equal to either ±50% or ±20% of the mean parameter value. Some parameters in Table 2—i.e., atmospheric radiation factor (β_{ar}), groundwater quantity (GW_q), and groundwater temperature (GW_t)—are not included in the biochemical model but influence the results of the simulation as parameters of the heat balance model. For these parameters, ranges of variations from the literature have been applied. For the sensitivity analysis, 45 parameters were considered and 500 Monte Carlo runs were performed.

The goodness of the linear approximation was assessed based on the coefficient of determination (R^2) and the adjusted coefficient of determination (R^2_{adj}). The variance inflation factor (VIF)—defined as the *i*th diagonal element of the inverse of the correlation matrix—was introduced in order to account for possible correlations between model parameters. Table 3 illustrates some statistics concerning the goodness of fit of the performed regression.

As the largest value of the VIF index is lower than 5, the correlation between model parameters is not high enough to reject the linear approximation. The regression analysis using average DO concentration as critical output, however, gives better results than with the exceedance frequency of low DO values, which suggests that for extremely low DO values, the model reacts more nonlinearly to parameter changes. In particular, the value of R^2_{adj} of the regression that uses exceedance frequency as critical output, is lower than 0.7. According to Saltelli et al. (2000), this indicates that the goodness of fit of the regression is insufficient to provide quantitative results about the sensitivity of the model with respect to the model parameters. To address this issue, following Saltelli et al. (2000), we performed a substitution of the values of the parameters and results by their ranks in order to obtain ranked standardised regression coefficient (RSRC) as a sensitivity measure.

The analysis of the resulting RSRCs shows that the parameters that more strongly affect the frequency of events with DO concentration below 5 mg/L are, in descending order of importance, Kla_{base}, Pow_h, α_1 and $K_{gro,ALG}$. For what concerns the second critical output, namely the average value of DO, the model is most sensitive to (in descending order of importance) $K_{gro,ALG}$, α_1, Kla_{base} and $K_{HPO_4,het}$. Besides the mentioned model parameters, both critical outputs were significantly affected by the input parameters. As these are based on direct measurements in the river, however, it can be expected that their value is known with relatively good precision.

2.7. *Scenario analysis*

Four scenarios, representing different shading conditions along the modelled stretch, were evaluated. All scenarios refer to the shading conditions induced by fully developed trees and take into account the effects of shading on both incident solar radiation and water temperature.

The first scenario was designed to represent conditions of alternate shading along the selected river stretch. This is reflected in the model by alternating shaded and not-shaded conditions in the tanks that simulate the river stretch. Since the ten tanks used in the model represent stretches of river characterized by different lengths, the total length of the shaded sections is greater than half of the total length of the stretch and amounts to 11.9 km (61% of the total length). The second scenario represents conditions of upstream shading. In this scenario, the first three tanks-in-series are modelled with shaded conditions, while conditions in the remaining tanks reflect the current, not-shaded conditions. The three

Table 2 – Parameter values with frequency distribution characteristics for the Monte Carlo sampling.

Parameter	Description	Mean	Var.	Min.	Max.	Dist.
α_1	Coefficient of rating curve in AMINAL 73	9.6859	20%	7.7487	11.6231	T
α_2	Coefficient of rating curve in AMINAL 76	4.8226	20%	3.8581	5.7871	T
β_1	Exponent of rating curve in AMINAL 73	1.1613	20%	0.9290	1.3936	T
β_2	Exponent of rating curve in AMINAL 76	1.6157	20%	1.2926	1.9388	T
Kla_{base}	Re-aeration coefficient	1	50%	0.5	1.5	T
Pow_v	Parameter for calculation of re-aeration coefficient	0.97	50%	0.485	1.455	T
Pow_h	Parameter for calculation of re-aeration coefficient	–1.67	50%	–0.835	–2.505	T
$K_{gro,H,anox}$	Maximum anoxic specific growth rate of heterotrophs	1.6	50%	0.8	2.4	U
$K_{gro,ALG}$[a]	Maximum specific growth rate for algae	2	50%	1	3	U
$K_{gro,N1}$	Maximum specific growth rate of 1st stage nitrifiers	0.8	50%	0.4	1.2	U
$K_{gro,N2}$	Maximum specific growth rate of 2nd stage nitrifiers	1.1	50%	0.55	1.65	U
K_{hyd}	Hydrolysis rate constant	3	50%	1.5	4.5	U
$K_{resp,ALG}$[a]	Maximum specific respiration rate of algae	0.1	50%	0.05	0.15	U
$K_{resp,H,aer}$	Maximum aerobic specific respiration rate of heterotrophs	0.2	50%	0.1	0.3	U
$K_{resp,H,anox}$	Maximum anoxic specific respiration rate of heteroptrophs	0.1	50%	0.05	0.15	U
$K_{resp,N1}$	Maximum specific respiration rate of 1st stage nitrifiers	0.05	50%	0.025	0.075	U
$K_{resp,N2}$	Maximum specific respiration rate of 2nd stage nitrifiers	0.05	50%	0.025	0.075	U
$K_{O,H,aer}$	Saturation/inhibition coefficient for aerobic endogenous respiration of heterotrophs	0.2	50%	0.1	0.3	U
$K_{HPO_4,H,aer}$	Saturation coeff. aerobic growth of heterotrophs on HPO_4	0.02	50%	0.01	0.03	U
$K_{N,H,aer}$	Saturation coeff. aerobic growth of heterotrophs on N	0.2	50%	0.1	0.3	U
$K_{NH_4,N1}$	Saturation coeff. growth of 1st stage nitrifiers on NH_4	0.5	50%	0.25	0.75	U
$K_{HPO_4,ALG}$[a]	Saturation coefficient for growth of algae on phosphate	0.02	50%	0.01	0.03	U
$K_{O,N1}$	Saturation/inhibition coefficient for aerobic endogenous respiration of 1st stage nitrifiers	0.5	50%	0.25	0.75	U
$K_{NO_2,N2}$	Saturation/inhibition coefficient for aerobic endogenous respiration of 2nd stage nitrifiers	0.5	50%	0.25	0.75	U
$K_{NO_3,H,anox}$	Saturation coeff. for anoxic growth of heterotrophs on NO_3	0.5	50%	0.25	0.75	U
$K_{NO_2,H,anox}$	Saturation coeff. for anoxic growth of heterotrophs on NO_2	0.2	50%	0.1	0.3	U
$K_{N,ALG}$[a]	Saturation coefficient for growth of algae on N	0.1	50%	0.05	0.15	U
$K_{HPO_4,H,anox}$	Saturation coeff. anoxic growth of heterotrophs on HPO_4	0.02	50%	0.01	0.03	U
K_I[a]	Saturation coefficient for growth of algae on light	30	50%	15	45	U
$K_{O,ALG}$[a]	Saturation/inhibition coefficient for endogenous respiration of algae	0.2	50%	0.1	0.3	U
$K_{death,ALG}$[a]	Specific death rate for algae	0.1	50%	0.05	0.15	U
K_{ads}	Phosphate adsorption rate constant	0.5	50%	0.25	0.75	U
β_{ar}[a]	Atmospheric radiation factor	0.85		0.8	0.9	U
GW_q[a]	Groundwater quantity	0.1		0.08	0.12	U
GW_t[a]	Groundwater temperature	11		10	12	U

[a] Parameter used for difference analysis in the uncertainty analysis. Var., variation; Dist., distribution; T, triangular distribution; U, uniform distribution.

upstream tanks account for approximately half the length of the stretch (9.6 km, 49% of the total length). The third scenario is chosen to represent conditions of downstream shading. Only the seven downstream tanks of the stretch are in shaded conditions. This corresponds to shading along 9.9 km, i.e. 51% of the total length. The fourth and last scenario represents conditions of shading along the entire stretch.

3. Results and uncertainty analysis

3.1. *Model calibration*

The calibration procedure involved two distinct steps: hydraulic calibration and calibration of the biochemical trans-

Table 3 – Regression statistics for DO concentration below critical threshold (5 mg/L) and average DO concentration.

	Original parameter values	Ranked parameter values
DO concentration <5 mg/L		
R^2	0.65	0.76
R^2_{adj}	0.61	0.73
Max VIF	1.20	1.15
Average DO concentration		
R^2	0.92	0.92
R^2_{adj}	0.92	0.91
Max VIF	1.16	1.14

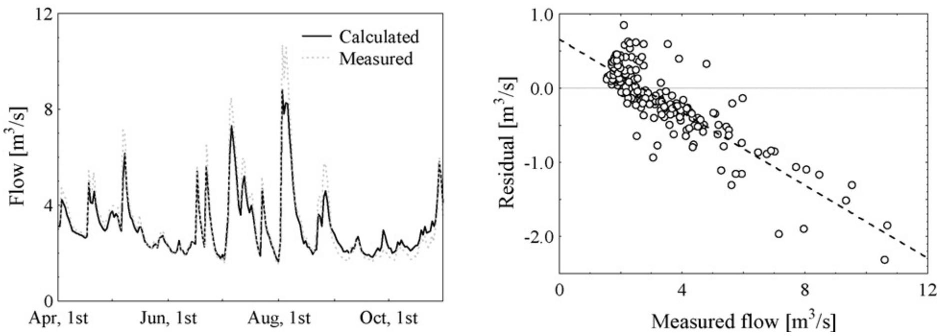

Fig. 2 – Comparison between measured and calculated flow time series (*left*) and plot of residuals (difference between measured and calculated flow) (*right*).

formations. The simulation period consisted of 100 days of steady state input for the initialisation of the model, followed by a dynamic input with hourly time-step for the period from April 1 to October 31, 2002. The steady state inputs were obtained by averaging influent flows and pollutant concentrations during April 2002.

The hydraulic model was calibrated by adjusting the number of tanks-in-series in which to subdivide the selected river stretch. Fig. 2 illustrates the results of the hydraulic calibration 12.2 km downstream of the initial section.

The hydraulic routing of the Nete is in general well reproduced by the model with the exception of peak flows, which are underestimated. This may be due to the fact that the contribution to the flow of several minor tributaries of the Nete is neglected in the hydraulic model, while this contribution may become significant during periods of high rainfall. A number of 10 tanks-in-series with different lengths and variable volumes was chosen as an acceptable compromise between calibration results and computation time.

The biochemical model was calibrated at different sections of the stretch based on six water quality variables: DO, chemical oxygen demand (COD), nitrates, ammonium nitrogen, phosphates, and phytoplanktonic algal biomass. Following the results of the sensitivity analysis, only few significant parameters were tuned for the calibration: the re-aeration coefficient (Kla_{base}), Monod's constants for the death of algae, the growth of algae, the algal uptake of HPO_4, and the uptake of nitrogen by algae, and the concentration of heterotrophic (X_H) and autotrophic (X_{N1} and X_{N2}) bacteria in the upstream section. In this procedure, X_H, X_{N1}, and X_{N2} were treated as model parameters rather than state variables since, in the absence of direct measurements, they had to be estimated on the basis of literature indications. The best fit was obtained for upstream concentrations equal to 4 mgX_H/L, 0.3 mgX_{N1}/L, and 0.1 mgX_{N2}/L. Fig. 3 shows the best-fit results in the downstream section of the modelled stretch. The points represent the measurements made by VMM in the same section.

Because of data scarcity, the evaluation of the calibration procedure had to be limited to a visual assessment of the fit between the model output and the few available measurements. The predictive power of the model output with respect to the measured values is also reported, although the information provided is limited by the small sample size and was not used in the calibration procedure. The model correctly predicts a decreasing trend in the concentration of dissolved oxygen ($R^2 = 0.20$, $N = 11$) and nitrates ($R^2 = 0.32$, $N = 11$) over the simulation period. Also, the observed increase in the concentration of algae during the hottest months—roughly from July to September—is predicted ($R^2 = 0.99$, $N = 3$). The few observations available lie within the predicted daily range of variation of algal concentration during the measurement days. Overall, the concentration of ammonium ($R^2 = 0.21$, $N = 11$) and COD ($R^2 = 0.08$, $N = 11$) are well represented. In judging the results concerning phosphates ($R^2 = 0.12$, $N = 11$), it should be noted that during the simulation period VMM often reported concentrations below the detection limit: all points in Fig. 3 that indicate a concentration of 0.15 mg/L must be interpreted as indicating a concentration less or equal to 0.15 mg/L.

Due to the data scarcity issue, it was not possible to validate the model on a separate set of data. Because of the lack of model validation, the study must be indicated as semi-hypothetical and its results cannot be interpreted as supporting a specific riparian management strategy along the Grote Nete.

3.2. *Results of scenario and uncertainty analyses*

The effect of shading on the water quality in the four scenarios is evaluated comparing the output of the scenarios with that of the calibrated model, which represents the current reference shading conditions along the stretch. The effect on primary production and algal biomass was evaluated both in terms of the change in the algal concentration in the closing section and of the algal growth within the stretch.

The results of the scenario analysis indicate that the reduction in the concentration of algae at the downstream section of the selected stretch, averaged over the whole simulation period, ranges between 8.0% and 18.7%. The highest effect on the algal concentration is given, as expected, by the scenario with complete shading (18.7%), while alternate shading (10.6%) and downstream shading (11.1%) induce a slightly higher reduction than upstream shading (8.0%). The sum of the effects of upstream and downstream shading corresponds approximately to the effect of complete shading. The effect of alternate shading is analogous to the effect of downstream shading, although in the corresponding scenarios different lengths of the stretch are in shaded conditions (61% for alternate against 51% for downstream shading).

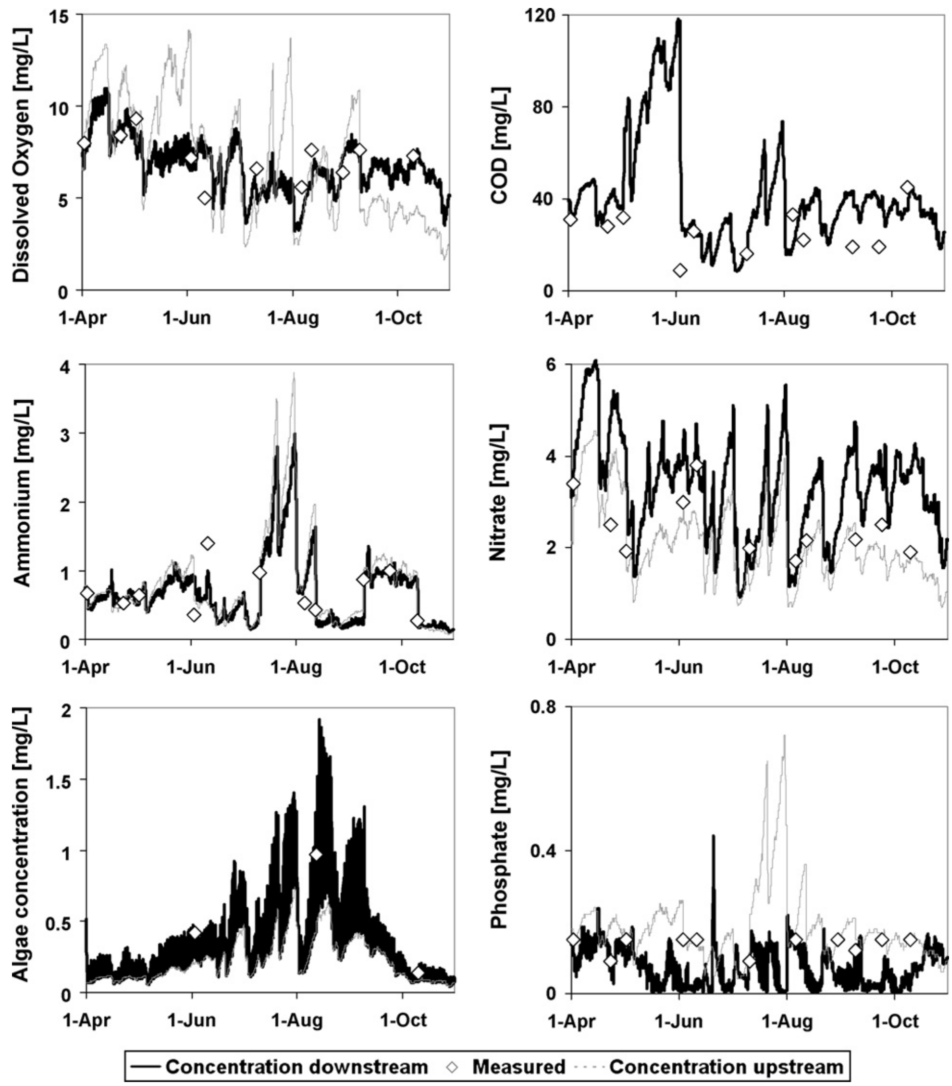

Fig. 3 – Calibration of biochemical model in the closing section for six water quality parameters.

The effects of shading are more significant when differences in algal growth are considered. The growth of algae during the whole dynamic period is reduced by 44.2% in the scenario with complete shading. As for algal concentrations, the comparison of the results of the different scenarios indicates that algal growth is more affected by downstream shading (26.3% reduction) than by upstream shading (18.9%), and that downstream and alternate shading (25.0%) produce similar results on algal growth despite different lengths of shaded conditions between the two scenarios.

The effects of shading on the concentrations of DO, COD, phosphates, and ammonium nitrogen are small when looking at minimum, maximum, and average concentrations over the whole simulation period. Significant changes are found, however, in the number of hours that the concentration of the selected water quality variables exceeds (or falls below in the case of DO) a certain critical threshold value (Table 4). The critical values assumed in Table 4 were chosen according to the basic water quality legislation in Belgium.

Table 4 – Number of hours of exceedance of water quality variables under different shading scenarios.

Scenario	DO < 5 mg/L	PO_4 > 0.3 mg/L	NH_3 > 5 mg/L
0. No shading	394	11	0
1. Alternate shading	373	11	0
2. Upstream shading	372	11	0
3. Downstream shading	397	11	0
4. Complete shading	377	12	0

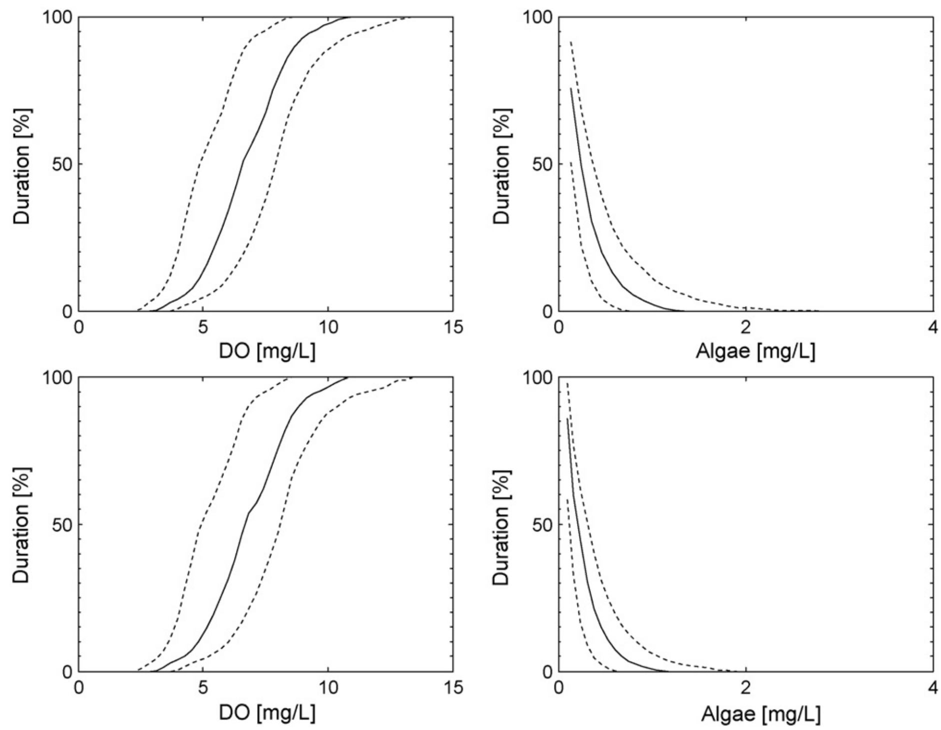

Fig. 4 – Concentration-duration curves of DO and algae for the current shading conditions (*above*) and for the fully shaded scenario (*below*).

According to the analysis of Table 4, the time the critical concentration dissolved oxygen is exceeded is reduced in the case of upstream, alternate and complete shading. Downstream shading, however, induces slightly worse conditions for DO, its concentration being longer under the critical level of 5 mg/L. Exceedance of critical phosphate concentration is not affected by shading. Ammonium did not exceed the critical value in any circumstance.

To verify how much the results are influenced by the uncertainty in the parameters, a concentration-duration curve for DO and algae with 5th, 50th, and 95th percentiles is shown in Fig. 4 for the non-shaded scenario and the completely shaded one. The curves express the percentage of time that DO is below a given concentration. Only these two scenarios are shown as they represent the extreme cases: the other scenarios fall in between those two. The uncertainty is calculated with Monte Carlo simulations where parameter ranges are taken from Table 2.

The uncertainty bounds for DO and algal concentrations are wide. From this analysis it is therefore not possible to reach precise conclusions about the outcome of a scenario. However, not all the parameter and input uncertainties need to be taken into account for scenario analysis under uncertainty. Uncertainty on parameters that influence the outcomes in both scenarios in the same way, so called 'fully dependent parameters' (Reichert and Borsuk, 2002), will only shift the results in one direction and with the same magnitude. Therefore, an analysis on the difference of a variable in two scenarios was performed in order to see whether the 5th and 95th percentile uncertainty bounds around the difference include or not the zero. Only the independent parameters and inputs are used in such analysis. They are indicated with asterisks in Table 2. The results of the described uncertainty analysis on the difference between scenario 0 and scenario 4 for DO concentration and average growth of algae at the end of the stretch are shown in Figs. 5 and 6.

From Figs. 5 and 6, it can be inferred that shading does not significantly reduce or increase the lowest oxygen levels in this specific stretch of the river. The uncertainty concerning the difference between the scenarios contains both positive and negative values. A clear trend is visible in the concentration of algae, which is lower in the shaded scenario

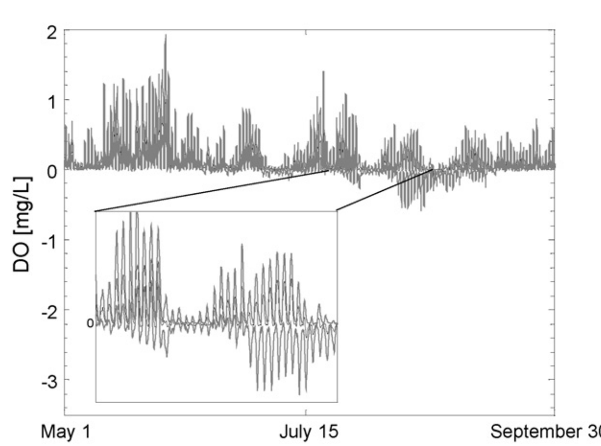

Fig. 5 – Time profile of difference between current conditions and full shading scenario for DO; the *full line* represents the 50%-ile, while the *dotted lines* are the 5%-ile and 95%-ile.

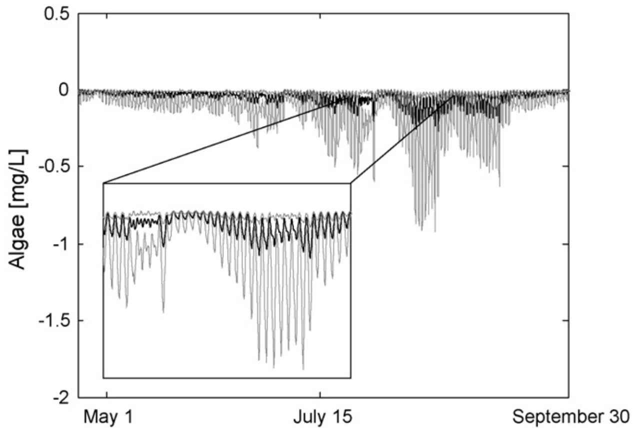

Fig. 6 – Time profile of difference between current conditions and shading scenario for algae; the *full line* represents the 50%-ile, while the *dotted lines* are the 5%-ile and 95%-ile.

particularly during warmer periods, when most algal growth occurs.

4. Discussion and conclusions

The present study makes use of a dynamic modelling approach to predict water quality reactions within a moderate-size river and predicts water quality changes in the selected river stretch as induced by riparian shading of forested river banks. The approach followed differs significantly from previous studies on shading effect, which focused on field measurements campaigns (Boothroyd et al., 2004; Ensign and Mallin, 2001; Towns, 1981; Quinn et al., 1992, 2004; Stone et al., 2005; Sweeney et al., 2004; Fritz et al., 2004; Wilcock et al., 2004; Dawson and Haslam, 1983) or steady-state models (Parkyn et al., 2005) accounting for only a small sub-sample of the complex chemical, physical, and biological interactions occurring in a watercourse. Due to its semi-hypothetical character, it is suggested that the results of this study have a qualitative validity that extends beyond the specific case-study to watercourses with similar characteristics. The results of the study are thus meant to be complementary to field measurement campaigns and possibly contribute to direct future research inasmuch as they allow to characterize the shading effect independently from other processes that occur in riparian buffers—e.g., nutrient retention—and thus avoid the often prohibitive costs required for extensive field measurement campaigns.

According to the analysis performed, the shading effect induced by reforestation of the river banks in small and moderate-size streams can effectively influence water quality, in particular in streams that suffer from excessive algal growth during summer months. The reduction in phytoplanktonic algal productivity due to shading was estimated to be 44.9% at the outlet of a 19.5-km long stretch of the Nete River in Belgium. The average algal concentration at the outlet was reduced by 18.6%. Such values are relatively low if compared to the results of previous studies. Boothroyd et al. (2004) and DeNicola and McIntyre (1991) reported strong (concentrations up 80–100 times lower) decreases in periphyton biomass concentration in non-shaded streams with respect to comparable vegetated control sites. Towns (1981) observed that shading under an artificial cover, intercepting 94% of the incoming radiation, was sufficient to cause the disappearance of all visible attached periphyton in a small stream in New Zealand.

Possible explanations for the lower shading effect on algal concentration found in this study include the relatively small length of the simulated stretch, the short time of travel between upstream and downstream section, and the short simulation period, which was limited to one season (summer 2002). Furthermore, previous research suggests that shading is not the only factor that determines algal reduction in forested streams. In a comparative study between vegetated and not vegetated streams with small and moderate widths, Boothroyd et al. (2004) found no significant size effect on the reduction in periphyton biomass: since the shading effect is strongly dependent on stream width, this suggests other processes occurring in vegetated riparian buffers—such as filtration of orthophosphates and nitrogen compounds from shallow subsurface flow (Borin et al., 2004)—represent other important factors in controlling stream eutrophication.

In this study, no evidence was found for the shading effect on minimum, maximum, and average concentrations of DO, COD, phosphates, ammonium nitrogen, and nitrates in the water. Previous research findings in this respect are contradictory. Sweeney et al. (2004) identified higher uptake rates per unit length of ammonium nitrogen and phosphate phosphorus in forested reaches than in deforested ones. Parkyn et al. (2005), however, predicted an increase in nutrient export in shaded, moderate size streams due to the reduced nutrient uptake by macrophytes and algae. With respect to such results, it should be reminded that the model developed in this study does not account for the long-term tendency of reforested streams to widen and thus offer a wider streambed area per unit length of stream, which is shown to play a critical role in the nutrient dynamics in the streams analysed by Sweeney et al. (2004). On the other hand, the active uptake of nitrogen compounds by heterotrophic bacteria is included among the model processes considered in this study but is not accounted for in the model developed by Parkyn et al. (2005). Uptake by heterotrophic microbial communities attached to bottom substrata may represent one of the major removal processes of dissolved nutrients (Sweeney et al., 2004).

In Summary, the results of this study concerning the reduction of the concentration of phytoplanktonic algae in the selected stretch of the Grote Nete River suggest that shading can effectively be implemented as a direct management strategy to improve water quality conditions in small- and moderate-size watercourses that are exposed to excessive algal growth during summer periods.

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