CHAPTER 6

Ecological modeling as a scientific tool for assessing eutrophication and mitigation strategies for Belgian coastal waters

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6.1 Unresolved 'eutrophication' issues in Belgian coastal waters (BCZ) that can be addressed by ecological models

The role of model simulation in understanding system complexity and in decision-making is nowadays recognized. Models offer the ability to test conceptual understanding of how components of a given system are linked, and to simulate complex biogeochemical interactions in a quantifiable and repeatable manner. Modeling, however, is useful only if new insights subsequently lead to validation or rejection of hypotheses upon more detailed examination. Thus information extracted from model simulations is strongly linked to the chosen structure (trophic resolution) of the model and to the parameterization of the interactions between the components. When properly validated, models can be useful in both hindcast and forecast modes and used for testing environmental policy alternatives and their impact. Models are indeed the only tools which can be used to investigate future developments by simulating scenarios with different constraints.

Anthropogenic eutrophication in the BCZ results from the input of transboundary (SW- Atlantic waters enriched by the Seine, Somme, Authie and Canche, and Rhine) and local (IJzer, Leie and Scheldt) sources (Fig. 6.1) of land-based nutrients (N, P, Si). The relative importance of these different sources in the BCZ has transnational implications for eutrophication management but is not accurately known. The impact of each source will depend on the evolution of human activity on the watershed and on large scale climatic phenomena such as the North Atlantic Oscillation (NAO; Hurrell 1995; Ruddick & Lacroix, 2008), which determines the weather conditions over North-western Europe. The eutrophication problem in the BCZ is most visible as massive undesirable algal

blooms in spring. These blooms are composed mainly of ungrazable colony forms of the Haptophycea *Phaeocystis* that supplement diatoms (Rousseau *et al.*, 2008) and spread over the whole area along a SW-NE gradient (Lancelot *et al.*, 1987), impacting the ecosystem function and services (Lancelot, 1995, Rousseau *et al.*, 2004). In spite of more than 20 years' survey of phytoplankton blooms in BCZ and adjacent waters, the spatio-temporal coverage and magnitude of *Phaeocystis* colony blooms is not clearly known. Also unclear still today is the link between *Phaeocystis* colonies and nutrient loads of anthropogenic origin. Nowadays, these blooms are recurrent and occur after an early-spring diatom bloom controlled by dissolved silicate availability (Rousseau *et al.* 2002; Rousseau *et al.*, 2008). *Phaeocystis* colonies grow on nitrogen and phosphorus and form with diatoms the bulk of phytoplankton biomass during the growing season but their magnitude varies from year-to-year (Lancelot *et al.* 1998). Attempts have been made to relate long-term fluctuations of *Phaeocystis* blooms in the Southern Bight of the North Sea to changes either in climate (Owens *et al.* 1989; Gieskes *et al.*, 2007) or in anthropogenic nitrogen and phosphorus river loads (Cadée & Hegeman, 1991), but the outcome is unclear. These hypotheses were recently reconciled by Breton *et al.* (2006) who showed that in coastal waters like the BCZ large-scale variability correlated with the NAO interacts with the local influence of river nutrient loads.

Therefore, due to the complexity and temporal variability of controls on marine planktonic food webs, the link between nutrient change and the coastal ecosystem function cannot be understood by simple correlation between events. Mechanistic models which are based on chemical and biological principles and describe ecosystem carbon and nutrient cycles as a function of environmental constraints are ideal tools to handle this complexity. Constructing a mathematical tool for understanding and advising on mitigation strategies for combating eutrophication problems in the *Phaeocystis*-dominated BCZ and its adjacent waters is the long term purpose of AMORE (Advanced Modeling and Research on Eutrophication; http://www.ulb.ac.be/assoc/esa/AMORE), a multidisciplinary consortium of scientists focusing their research activity on coastal eutrophication since 1997. To achieve such a long term objective AMORE has developed an integrated research methodology that combines field observations, process-level studies and models development in an iterative way. In this approach model development has a key position as integrator and test of new knowledge. The main mathematical tool is the three-dimensional (3D)-MIRO&CO model (Lacroix *et al.*, 2007a) that combines the 3D hydrodynamical model COHSNS (Lacroix *et al.*, 2004) determining the physical transport of water and of ecosystem components, and the ecological model MIRO describing interactions between the ecosystem components as previously tested in a multibox frame (OD-MIRO; Lancelot *et al.*, 2005). In this chapter 0D-MIRO and 3D-MIRO&CO models are used to address scientific unresolved issues such as:

- The geographical extent of *Phaeocystis* colony blooms in the Channel and Southern Bight of the North Sea with a focus on the Belgian waters;

- The relative contribution of Channel/Atlantic inflow and local sources to the nutrient status of the BCZ;

- The relationship between *Phaeocystis* colony blooms in the BCZ and nutrient enrichment

As well as managerial concerns such as:

- The geographical limits of problem and non-problem maritime areas in terms of eutrophication as defined by the Oslo and Paris Commission for the Prevention of Marine Pollution (OSPAR, 2005);

- The level of nutrient reduction needed to decrease *Phaeocystis* blooms in the BCZ to ecologically acceptable biomass and the achievement to be expected in 2015 after implementation of the European Union's Water Framework Directive (2000) in the Great North Basin.
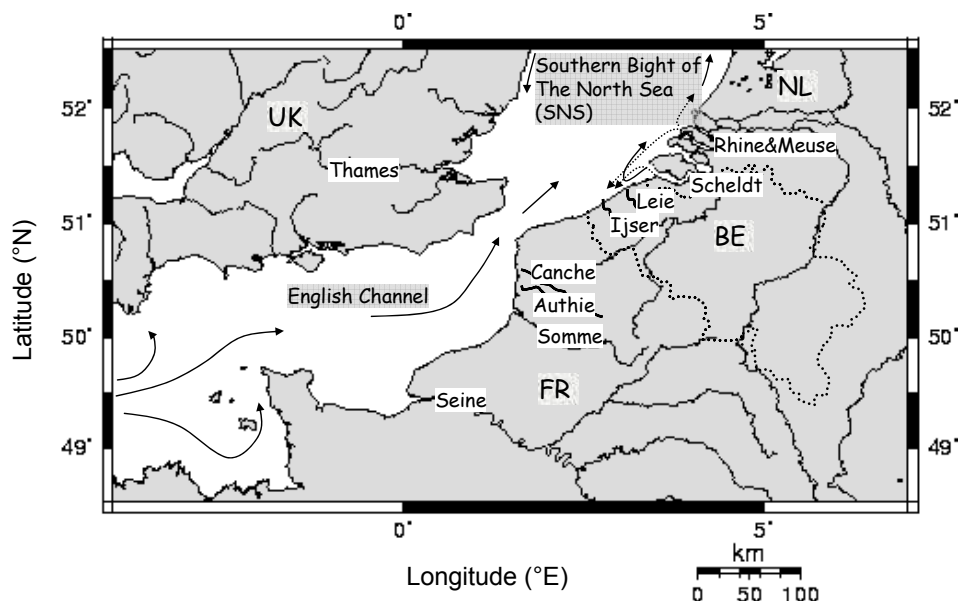
Figure 6.1. Map showing the modeled domain with river inputs and the mean general circulation

6.2 Model tools

Figure 6.2 shows the methodological approach chosen to construct the ecological model MIRO as well as the different implementation frames (0D-MIRO and 3D-MIRO&CO) used for tests and applications. The trophic resolution of MIRO was chosen based on the current mechanistic understanding of the eutrophication problem in the BCZ. MIRO describes C, N, P and Si cycling through aggregated components of the planktonic and benthic realms of *Phaeocystis*-dominated ecosystems (see justification of chosen state variables in Lancelot *et al.*, 2005). Its structure includes thirty-eight state variables assembled in four modules describing the dynamics of phytoplankton (three groups: diatoms, nanoflagellates and *Phaeocystis* colonies), zooplankton (two groups: copepods and microzooplankton), dissolved and particulate organic matter (each with two classes of biodegradability) degradation and nutrients [NO_3, NH_4, PO_4 and $Si(OH)_4$] regeneration by bacteria in the water column and the sediment. Equations and parameters were formulated based on current knowledge of the kinetics and factors controlling the main auto- and heterotrophic processes involved in the functioning of the coastal marine ecosystem. These are fully documented in Lancelot *et al.* (2005) and www.int-res.com/journals/suppl/appendix_lancelot.pdf. MIRO was first calibrated in a multi-box frame (0D-MIRO) delineated on the basis of the hydrological regime and river inputs. In order to take into account the cumulated nutrient enrichment of Atlantic waters by the Seine and Scheldt rivers, two successive boxes, assumed to be homogeneous, were chosen from the Seine Bight (French Coastal Zone, FCZ) to the BCZ (Fig. 6.2). Each box has its own morphological characteristics and river inputs (Table 1 in Lancelot *et al.*, 2005) and is treated as an open system, receiving waters from the upward adjacent box and exporting water to the downward box. In the current applications, river nutrient inputs are either extracted from national data bases (Lancelot *et al.*, 2005; Gypens *et al.*, 2007; Lacroix *et al.*, 2007a, b) or provided by RIVERSTRAHLER simulations (Lancelot *et al.*, 2007).

The RIVERSTRAHLER model establishes the link between the biogeochemical functioning of large river systems and the constraints set by the meteorological conditions, the morphology of the drainage network and the human activity on the watershed (Billen *et al.*, 1994, 1999; Garnier *et al.*, 1995). It is therefore an excellent tool to assess historical trends of eutrophication in BCZ.

The 3D version of MIRO, 3D-MIRO&CO, resulting from the implementation of MIRO in the 3D hydrodynamical COHSNS model (Lacroix *et al.*, 2004), simulates the transport and dynamics of inorganic and organic nutrients, phytoplankton, bacterioplankton and zooplankton biomass in the Western Channel and Southern Bight of the North Sea. For this purpose the 3D-MIRO&CO model has been set up for the region between 4° W (48.5°N) and 52.5°N (4.5°E) with the bathymetry shown in Figure 6.2, using a 109 by 97 horizontal grid with resolution 5' longitude (approx. 5.6 km) by 2.5' latitude (approx. 4.6 km) and with 5 vertical sigma coordinate layers. All details of its implementation and the description of forcing, boundary and initial conditions can be found in Lacroix *et al.* (2007a).
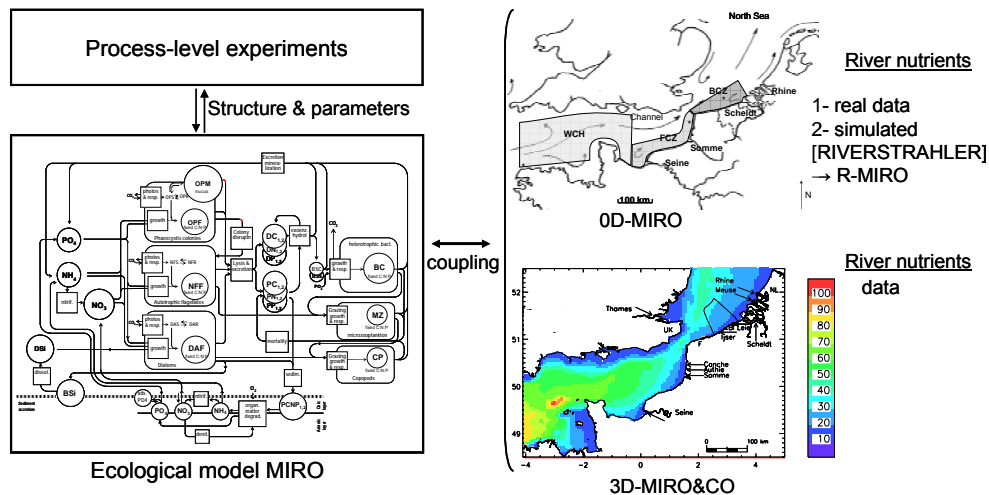
Figure 6.2. Schematic representation of the ecological model MIRO and its implementation frame (0D-MIRO and 3D-MIRO&CO) and river forcing.

6.3 Improved understanding of bloom dynamics and nutrient sources

6.3.1 Present-day geographical extent of *Phaeocystis* colony blooms

Figure 6.3 shows 3D-MIRO&CO simulations of biomass distribution of diatoms and *Phaeocystis* colonies in respectively early- (Fig. 6.3a) and end- (Fig. 6.3b) April 2004, *i.e.* at the time of their own blooming. In agreement with observations the simulated diatoms are blooming earlier in spring compared to *Phaeocystis* colonies. Both phytoplankton taxa show the highest biomass in the vicinity of the river mouths and correspond with areas of elevated nutrient stocks illustrated by the simulated distribution of nutrients in end-January (Fig. 6.3c), *i.e.* when remineralization is achieved and biological uptake is the lowest due to temperature and light limitation. In the eastern part of the 3D-MIRO&CO domain *Phaeocystis* colonies are simulated in the whole area extending from the Seine Bay up to the Northern limit of the 3D-MIRO&CO domain. According to 3D-MIRO&CO simulations the whole BCZ area is invaded by *Phaeocystis* colonies but the highest concentrations are to be found inshore especially in the vicinity of the Scheldt mouth (Fig. 6.3b). Nevertheless, the highest *Phaeocystis* concentrations in the geographical domain are simulated in the Dutch coastal waters due to SW-NE cumulated fluxes of nutrients to which are added land-based nutrients discharged by the river Rhine. In general the simulated *Phaeocystis* colony presence (Fig. 6.3b) agrees with what is known from sporadic monitoring sampling except for the Seine Bay where colonies have never been recorded. This discrepancy is intriguing and suggests that there is some aspect of ecosystem functioning that is not well understood and therefore not represented by the model equations. The reason for this is under investigation.

The analysis of 3D-MIRO&CO simulations over 12 years (1993-2004) indicates significant interannual variations in the timing, the magnitude and the geographical extent of *Phaeocystis* blooms. This is visible on Figure 6.4 which compares 3D-MIRO&CO simulations of *Phaeocystis* colonies in the investigated domain as obtained on 3rd May in 1995, 1998, 2002 and 2004. Such interannual difference in the spreading of *Phaeocystis* colonies is due to different meteorological conditions that determine the onset of the bloom and the timing of the maximum bloom development (Fig. 6.5). However, as shown on Figure 6.5 that maps the *Phaeocystis* colony distribution at the time of their maximum spreading in 1995, 1998, 2002 and 2004, there are also significant difference between years in magnitude and geographical extent. These are explained both by the varying meteorological conditions and nutrient river loads which together will determine the spreading of nutrients in the coastal area.
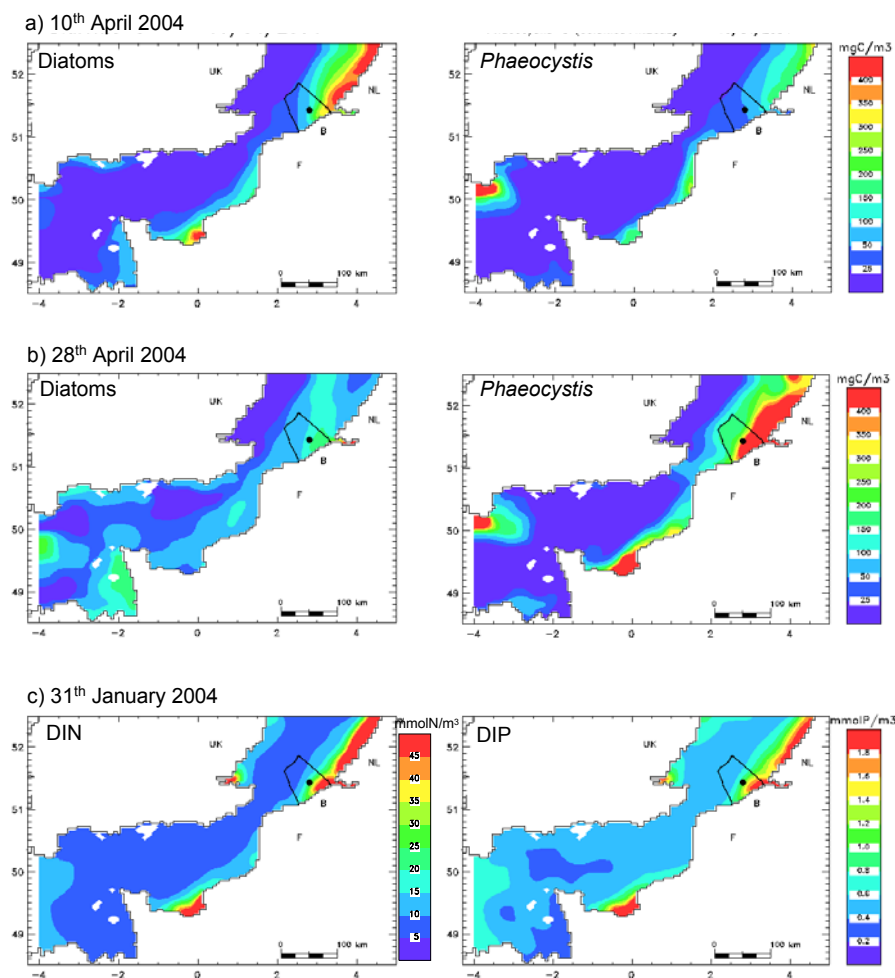

Figure 6.3. 3D-MIRO&CO simulations of diatom and *Phaeocystis* blooms in early (a) and end- (b) April 2004 and of nutrient concentrations in January 2004 (c).
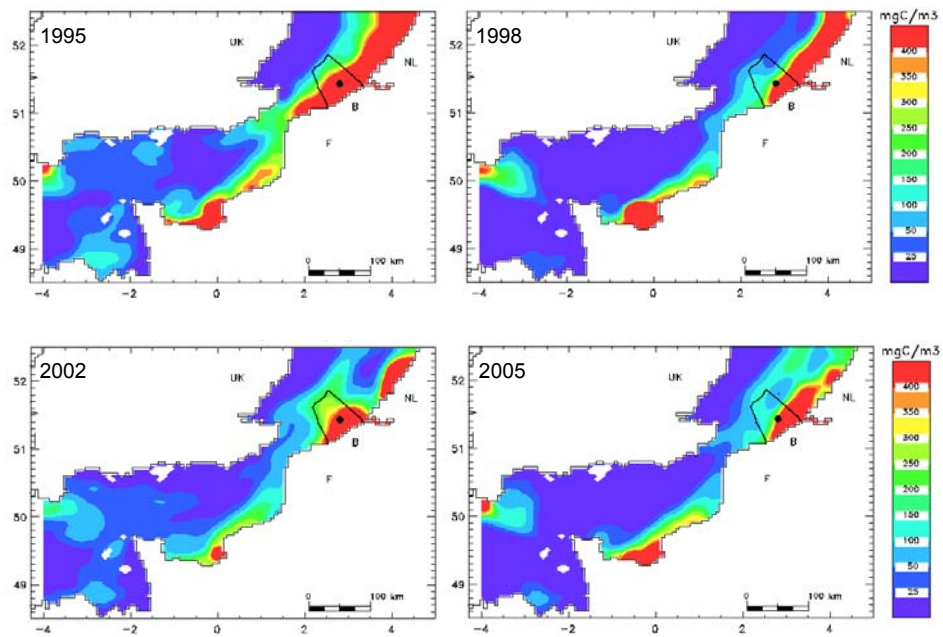
Figure 6.4. 3D-MIRO&CO simulations of *Phaeocystis* blooms on 3rd May in 1995, 1998, 2002 and 2004.
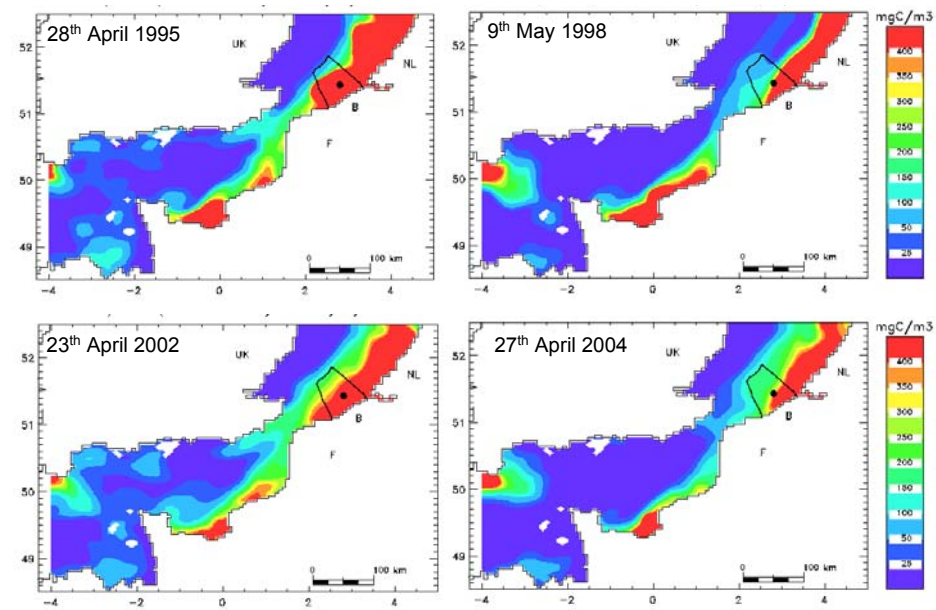

Figure 6.5. Maximum spreading of 3D-MIRO&CO *Phaeocystis* blooms in 1995, 1998, 2002 and 2004.

6.3.2 Contribution of different rivers (Seine, Scheldt and Rhine/Meuse) to the BCZ nutrient pool

The origin of inorganic nutrients (Dissolved inorganic nitrogen DIN and inorganic phosphate DIP) available for phytoplankton growth in the BCZ is indirectly investigated by running 3D-MIRO&CO sensitivity scenarios with decreasing nutrient inputs from inflowing Channel waters, Scheldt/Leie/IJzer and Rhine/Meuse by respectively 1% (Lacroix *et al.*, 2007b). Such a small decrease is chosen in order to ensure a quasi linear ecosystem response, effectively tracing the fate of nutrients from different sources in the current situation. The effect of this reduction is estimated by comparing the obtained average 1993-2003 field concentrations with those obtained with the non-perturbed simulation. Figure 6.6 illustrates which river contributes the most to the surface nutrient relative difference. Clearly, the nutrient reduction from the Seine has the strongest impact on surface DIN and DIP in the whole 3D-MIRO&CO domain, except the Belgian and Netherlands coastal zones (Fig. 6.6). Interestingly, the Scheldt has a larger effect than the Seine in the BCZ where eutrophication is most severe. The region of highest influence by the Rhine/Meuse (NE area of the model domain) extends more southward (until the Scheldt mouth) for DIN than that for DIP (Fig. 6.6). The fate of nutrients is thus somewhat different from the fate of freshwater (Lacroix *et al.*, 2004), where the Rhine/Meuse discharge was found to be more important in the BCZ. This difference can be attributed to the longer time scale for dispersion of freshwater, which is effectively conserved over decadal time scales whereas nutrients are transported as a conserved, dissolved constituents only for a few months (winter).
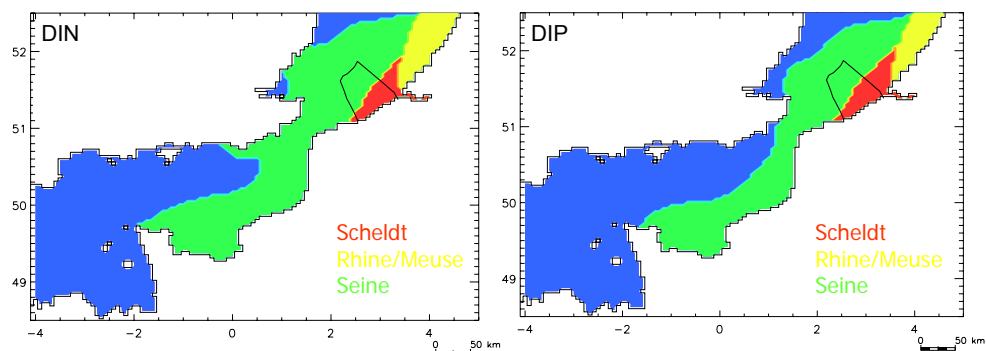

Figure 6.6. 1993-3003 highest river contribution (Seine, Scheldt, Rhine-Meuse) to the surface nutrients in the 3D-MIRO&CO domain.

6.3.3 Relationship between *Phaeocystis* colonies and nutrient enrichment in the BCZ: R-MIRO simulations over the past 50 years

Recent statistical analysis of fourteen-years (1988-2001) of intensive phytoplankton monitoring in the central BCZ (Breton *et al.*, 2006) in relationship with basin scale and local constraints indicates that the current interannual variability of *Phaeocystis* colony blooms is determined by river-based nitrate pulses modulated by NAO which affects the local hydrology, *i.e.* the relative contribution of Scheldt and Atlantic waters to the BCZ water budget. The variability of the hydrodynamics makes it difficult to establish links between *Phaeocystis* colonies and direct and indirect anthropogenic sources of nutrients when based on time series at fixed grid stations (Breton *et al.*, 2006). An additional difficulty is the lack of long-term time series of river nutrient inputs.

Long-term model simulations over periods of sustained human pressure provide a useful tool to analyze the link between blooms and nutrient enrichment. In this section, the link between *Phaeocystis* colony blooms in the BCZ and direct (Scheldt) and transboundary (Atlantic waters flowing into the BCZ) inputs of land-based nutrients is explored by performing 0D-MIRO simulations. The model is forced with nutrient fluxes simulated by the RIVERSTRAHLER model applied to the Seine and the Scheldt watersheds (Billen *et al.*, 2001; 2005) over the past 50 years, a period of intensive changes in land use and human activity. For this application targeting nutrients, an average meteorological year (global solar radiation, temperature, rainfall) is considered for the 1950-2000 period while changes in land use and changes in annual urban and industrial wastewater discharges are documented by 10 year periods and by 5 year periods respectively. Details on the coupling between RIVERSTRAHLER and MIRO (R-MIRO) are to be found in Lancelot *et al.* (2007). Simulations are analyzed in terms of historical evolution of nutrient sources to BCZ and nutrient enrichment and phytoplankton maxima in BCZ.

Nutrient inputs to BCZ over 50 years

Figure 6.7 compares the R-MIRO simulated 1950-2000 evolution of nutrient inputs to the BCZ distinguishing between the Seine-enriched Atlantic (S-Atl) inflow and the local Scheldt delivery. A contrasted historical evolution is clearly visible for the three nutrients considered (DSi, DIN and DIP). Between 1950 and 2000, Scheldt and S-Atl are contributing almost equally as DSi sources to the BCZ and both decrease slowly over the period up to 1995 after which a small increase is simulated (Fig. 6.7a). These variations result from the level of DSi consumption simulated by RIVERSTRAHLER in the rivers in response to the increased (or decreased) inputs of N and P from the watershed. On the other hand, DIP simulations show for both sources significant and similar trends: a sustained increase between 1950 and 1985 (factor 3; Fig. 6.7b) explained by the population increase and economical development, followed after 1985 by a progressive decrease until values typical of around 1965 are reached in 2000 (Fig. 6.7b). This decrease reflects the efficiency of P reduction measures implemented in the late eighties (mainly the abolition of polyphosphates in washing powders; Billen *et al.*, 2001; 2005). When comparing the two nutrient sources, it is clear that DIP is brought mostly by the S-Atl (74-94%). As for DIP,

R-MIRO simulates a factor 3 enrichment of DIN fluxes to the BCZ over the 1950-1985 period (Fig. 6.7c). A difference here with respect to DIP is the contribution of the different sources and the simulated evolution of DIN inputs which remain high until 2000 (Fig. 6.7c). Interestingly the contribution of the Scheldt to the DIN pool in the BCZ increases over the simulated period (1950-1970: 25%; 1975-2000: 40%) probably related to the importance of cattle wastes on the Scheldt watershed. Altogether, the historical R-MIRO reconstruction of nutrient inputs to the BCZ indicates that increased human activities on the watershed were modifying not only the quantity but also the quality of nutrient fluxes delivered to the coastal sea, being already Si-depleted in 1950 and in excess N with respect to P (N:P>25) after 1990.
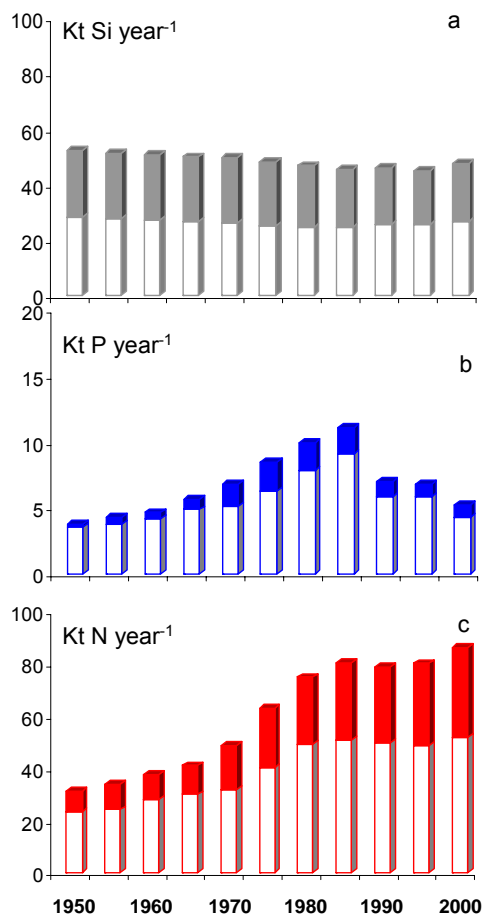

Figure 6.7. R-MIRO simulations of DSi (a), DIP (b) and DIN (c) inputs to the BCZ over the 1950-2000 period discriminating between the S-Atl inflow (open block) and the local Scheldt delivery (filled block).

50 year evolution of nutrient enrichment in BCZ

The quantitative and qualitative changes in DSi, DIN and DIP delivery to the BCZ (for which simulation results are shown in Figure 6.7) affect the nutrient status of the coastal area. This is best seen from the R-MIRO simulated winter stock (Fig. 6.8a) and molar ratios of DSi, DIP and DIN when the later are compared to phytoplankton needs (Fig. 6.8b). The DSi stock maintains a value of 15.7 mmole m^{-3}, *i.e.* close to the reference given for Atlantic waters (Radach *et al.*, 1995), up to 1980 after which it gradually decreases to 13.4 mmole m^{-3} in 2000. The simulated evolution of winter DIN and DIP concentrations parallels that shown for their inputs (Fig. 6.7 & 6.8). For both DIN and DIP, Figure 6.8 shows an exponential increase after 1960 reaching in 1985 a maximum value (DIP: 2.75 mmole m^{-3}; DIN: 47.8 mmole m^{-3}; Fig. 6.8a). The simulated trends of DIN and DIP winter concentrations diverge after 1985 with DIN increasing slowly up to 48.5 mmole m^{-3} in 2000 and DIP decreasing gradually up to 50% of the maximum 1985 value in 2000 (Fig. 6.8a). These contrasted trends have an impact on the qualitative nutrient status of the BCZ which, when compared to the nutrient requirement of coastal diatoms (molar N:Si:P=16:16:1), suggests a Si limitation between 1960 and 1985 but P limitation after this period (Fig. 6.8b). DIN is always in excess, especially after 1990 when N:P simulated values are above 25 (Fig. 6.8b).
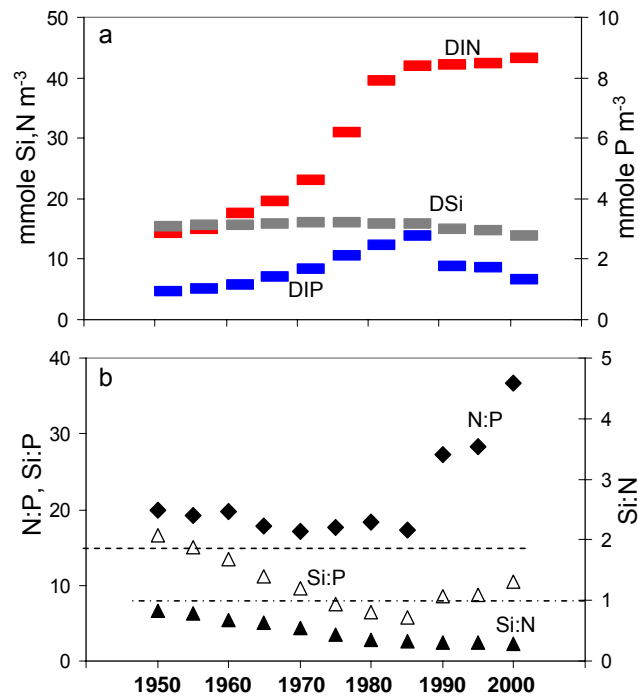

Figure 6.8. R-MIRO simulations of BCZ winter nutrient concentrations (a) and their elemental ratio (b) over the 1950-2000 period.

50 year evolution of phytoplankton blooms in BCZ

Phytoplankton blooms are usually defined as maximum concentration reached and/or as bloom-integrated biomass. Here we choose maximum biomass reached in spring as an indicator of an ecological imbalance between phytoplankton and zooplankton development. The analysis of R-MIRO simulations of phytoplankton bloom dynamics in the BCZ over the 1950-2000 period (not shown) indicates that nutrient enrichment has no effect on the timing of the spring diatom-*Phaeocystis* colonies-summer diatoms succession but does have an effect on their relative abundance. The long term evolution of R-MIRO spring maxima of bulk phytoplankton (Chl *a*; Fig. 6.9a), spring and summer diatom and *Phaeocystis* colony biomass (Fig. 6.9b) are then analyzed in comparison with concomitant changes in nutrient enrichment (Fig. 6.8). Clearly, phytoplankton biomass increases in response to DIN and DIP enrichment (r^2=0.99) up to 1985 when a Chl *a* maxima three times higher than in 1950 is simulated (Fig. 6.9a). After 1985, model results show significant decreases (30% over 15 years) of total biomass which can be related to DIP decrease (Fig. 6.8a, Fig. 6.9a).
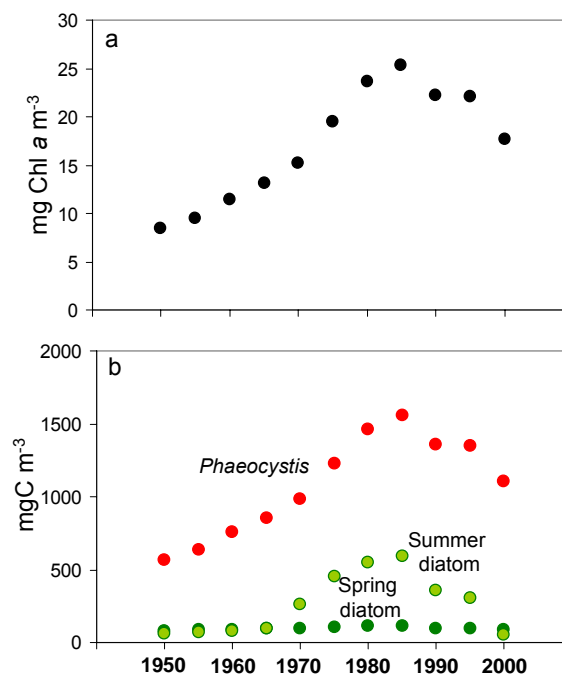

Figure 6.9. R-MIRO simulations in BCZ of maximum spring phytoplankton expressed in Chl *a* (a) and diatoms (spring and summer) and *Phaeocystis* colony biomass (b) over the 1950-2000 period.

Curiously the maximum biomass reached by spring diatoms seems little modified by increased nutrient enrichment which affects both *Phaeocystis* colonies and summer diatoms (Fig. 6.9b). After 1960, the maximum biomasses of summer diatoms and *Phaeocystis* colonies increase in parallel and both reach their maxima (~one order of magnitude higher than in 1950; Fig. 6.9b) in 1985 when DIN and DIP enrichment is maximum (Fig. 6.8a). Then, summer diatom and *Phaeocystis* colony maxima decrease in parallel apparently correlated with the simulated DIP decrease. In 2000 summer diatoms don't bloom and are maintained at biomass less than in 1950. In comparison, the decrease of *Phaeocystis* colonies is of little significance, *i.e.* 30% between 1985 and 2000 (Fig. 6.9b). Altogether this suggests that a well-balanced DIN and DIP enrichment is beneficial to *Phaeocystis* colonies and summer diatoms while spring diatoms remain unaffected; the imbalanced decreases of DIP and DIN in favor of elevated N:P ratios (>25) limit the growth of summer diatoms more than *Phaeocystis* colonies that are maintained at high biomass.

6.4 Support for eutrophication assessment and mitigation in the BCZ

6.4.1 Disturbance indicators due to eutrophication

Any assessment or mitigation of eutrophication requires the definition of reference indicators or thresholds to undesirable perturbation. There exists no unambiguous and universal indicator of disturbance due to marine eutrophication because of the high diversity of coastal ecosystems and their sensitivity to a sustained nutrient pressure. Simple bulk indicators such as winter stock of nutrients and maximum Chl *a* can be defined regionally in comparison with reference conditions. The latter are difficult to set both because unperturbated marine ecosystems are mostly inexistent and monitoring data are generally limited to the past 50 years, *i.e.* often too short to properly define reference conditions. In the scope of the OSPAR Strategy to combat eutrophication (2005), Belgium has provided 'best educated guesses' for bulk indicators of eutrophication problems in the BCZ. These thresholds are defined on basis of the winter concentrations of DIN (15 mmole m^{-3}), DIP (0.8 mmole m^{-3}) and molar DIN:DIP (24) and maximum Chl *a* (15 mg m^{-3}). *Phaeocystis* is cited as an indicator species but there is no clear consensus on the definition of a critical value in the OSPAR document. Most suggestions are based on the presence during 30 consecutive days of a number of cells (10^6 or 10^7 cells L^{-1}) which, from an ecological point of view, is of little significance as only colony forms are undesirable. Recently Lancelot *et al.* (in press) suggest a more ecologically- based criterion considering that most undesirable effects reported to *Phaeocystis* are due to its ability to form colonies which grow in size during their development and rapidly exceed the filtering capability of zooplankton (Lancelot *et al.*, 1994). On this basis, a *Phaeocystis* colony reference value of 150 mg C m^{-3} can be proposed for the BCZ, estimated from field observations of maximum biomass reached by zooplankton grazable *Phaeocystis* colonies (maximum diameter of 400 μm; Weisse *et al.*, 1994).

6.4.2 The geographical limits of BCZ waters affected by eutrophication

The geographical limits of problem (PA) and potential- (PPA) and non-problem (NPA) maritime areas in terms of eutrophication can be determined based on 3D-MIRO&CO simulations in winter and in spring and making use of OSPAR criteria for winter DIN, DIP, DIN:DIP and maximum Chl *a*. Figure 6.10 shows for each of above criteria the geographical frontier between eutrophied and non-eutrophied waters suggested by 3D-MIRO&CO simulations for the 1993-2000 period. Clearly the simulated area affected by eutrophication differs between criteria being more extended when using the winter nutrient criterion than the Chl *a* one (Fig. 6.10). Combining this allows to propose for the first time geographical limits to PA, PPA and NPA for not only the BCZ but also the 3D-MIRO&CO domain in general (Fig. 6.11). Clearly, following the OSPAR criteria, the eutrophication problem in the 3D-MIRO&CO domain is limited to nearshore waters in the vicinity of river mouths (Seine, Scheldt, Rhine/Meuse and Thames). PPA extends the affected area offshore showing a band parallel to the coast and pointing 70% of the BCZ area as PA and PPA with respect to eutrophication (Fig. 6.11a; Fig. 6.12). Interestingly enough the use of the *Phaeocystis* colony threshold of 150 mgC m^{-3} suggests that non-grazable *Phaeocystis* colonies are invading the whole BCZ in spring (Fig. 6.11b).
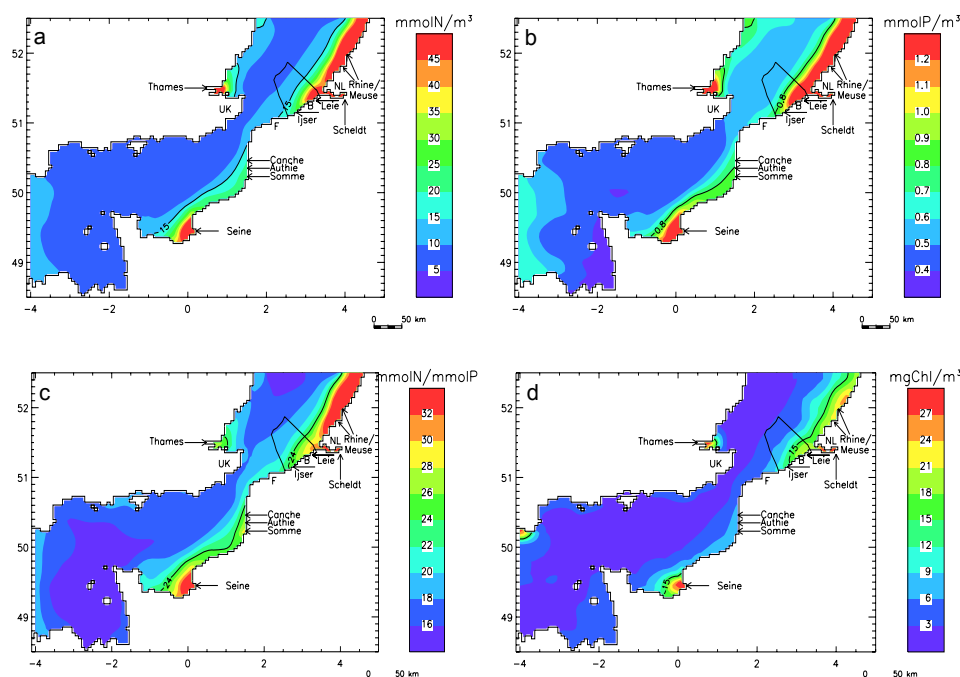

Figure 6.10. Eutrophication assessment based on 3D-MIRO&CO simulations of DIN (a), DIP (b), DIN:DIP (c) and Chl *a* (d) and making use of OSPAR criteria. (black line parallel to the coast)
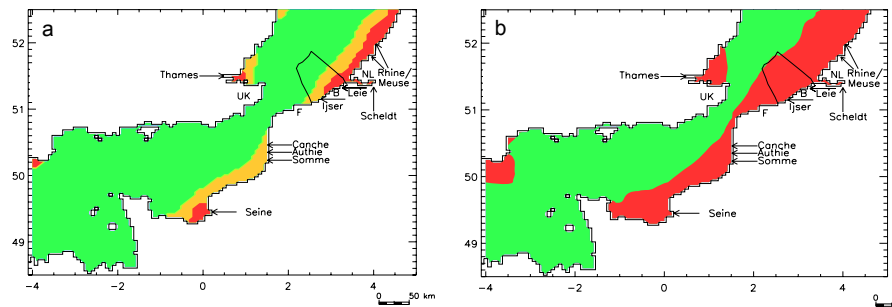
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Figure 6.11. Eutrophication status of the 3D-MIRO&CO domain and the BCZ based on the combination of maps shown on Figure 6.10 with red: PA, orange: PPA, green: NPA (a) and geographical distribution of grazable (green) and non-grazable (red) *Phaeocystis* colonies in spring (b).

6.4.3 Mitigation

Which nutrient reduction needs to be targeted?

The first way to approach eutrophication mitigation with ecological models consists in the identification of the optimal nutrient reduction [which nutrient(s) to reduce and by how much?] needed for decreasing substantially eutrophication. Sensitivity tests can be made on river inputs, comparing results obtained for winter nutrients and Chl *a* maxima with OSPAR targets. The nutrient reduction needed to limit the geographical extension of undesirable eutrophication in BCZ can therefore be estimated through 3D-MIRO&CO model scenarios with changing river nutrient inputs.
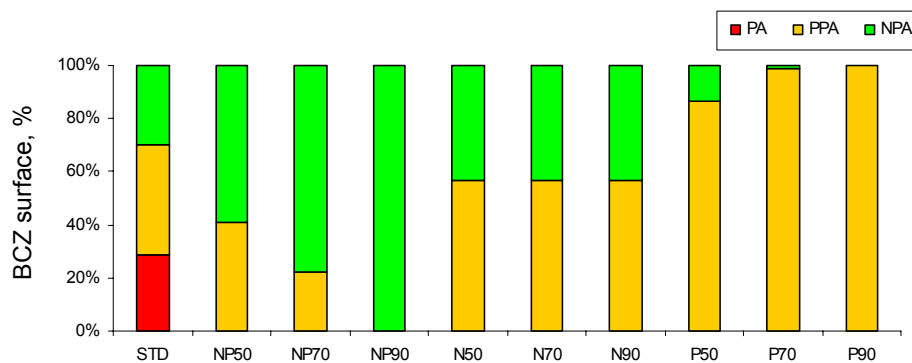

Figure 6.12. Changing BCZ areas (in % of the total BCZ surface) affected (PA), potentially affected (PPA) and non-affected (NPA) by eutrophication after nutrient reduction (50, 70, 90% of N; 50, 70, 90% of P and 50, 70, 90% of NP).

Figure 6.12 shows for each nutrient reduction tested (50, 70 and 90% N, P and N+P) the predicted decrease in the BCZ area affected or potentially affected by eutrophication. Clearly all the 50% nutrient reduction scenarios eliminate PA in the BCZ while a full recovery of the latter is obtained when both N and P are decreased by 90% (Fig. 6.12). As a general trend the best mitigation of eutrophication in BCZ is obtained when both N and P are decreased. The reduction of the only P predicts PPA for almost the whole BCZ surface (Fig. 6.12) explained by the elevated N:P (>24) resulting of P reduction while high N is maintained. The N reduction scenarios predict a similar distribution of NPA and PPA areas in BCZ whatever the N reduction level is, because the limit between PPA and NPA is here determined by DIP distribution and is not affected by N reduction scenarios

What is feasible?

Best nutrient reduction measures suggested by model sensitivity analysis are not necessarily feasible either technically or due to socio-economical constraints. As most nutrient reduction measures are implemented on the watershed and concern waste water treatment, industrial emissions and agricultural practices, the R-MIRO model is an appropriate tool for assessing the efficiency of ongoing or planned nutrient reduction measures. Figure 6.13 shows R-MIRO simulations of nutrient inputs to the BCZ, winter nutrient enrichment and maximum phytoplankton biomass reached in the BCZ obtained for 2015 after careful implementation of measures on waste water treatment and agricultural practices decided by Belgium and France in the scope of the Water Framework Directive of the European Union. These simulations are compared with the current situation illustrated by year 2000 simulations as well as a pristine scenario considering that the whole North Sea watershed is covered by forests. Analysis of results shows the efficiency of planned measures for reducing P and N, the latter however to a less extent. Interestingly DIP inputs simulated in 2015 are close to values obtained for the pristine reference. This is not found for DIN inputs which remain largely above pristine fluxes (Fig. 6.13). The reduction of nutrient inputs to the BCZ reached in 2015 is reflected in the nutrient status of the BCZ (Fig. 6.13). In 2015 DIP enrichment is expected to have been counteracted yielding a concentration in the BCZ of 0.9 mmole P m^{-3} i.e close to the OSPAR target of 0.8 mmole P m^{-3} and the pristine reference of 0.7 mmole P m^{-3}. The N reduction achieved in 2015 is largely insufficient with a winter DIN concentration of 43.6 mmole N m^{-3}, *i.e.* above the OSPAR target of 15 mmole N m^{-3} and 5 times higher than the pristine value (9.8 mmole N m^{-3}). This unbalanced nutrient reduction has dramatic consequences for the N:P ratio which reaches values as high as 46 (not shown) *i.e.* well above the OSPAR threshold of 24. The nutrient reduction reached in 2015 has positive feedbacks on the maxima reached by phytoplankton (Chl *a*), diatom and *Phaeocystis* colony biomass (Fig. 6.13). It is interesting to observe that, while the bulk phytoplankton target of 15 mg Chl *a* m^{-3} recommended by OSPAR is well achieved in 2015, the simulated decrease of *Phaeocystis* colony biomass is very disappointing and well above the ecologically-based criterion of 150 mg C m^{-3} (Fig. 6.13). The DIP reduction mostly affects summer diatoms which reach an annual maximum lower than under pristine conditions (Fig. 6.13). These simulations indicate that a bulk indicator of ecosystem change such as

maximum Chl *a* is not sufficient for appraising the ecosystem health of the BCZ. A *Phaeocystis* indicator and/or an unbalanced N:P winter ratio appear necessary.
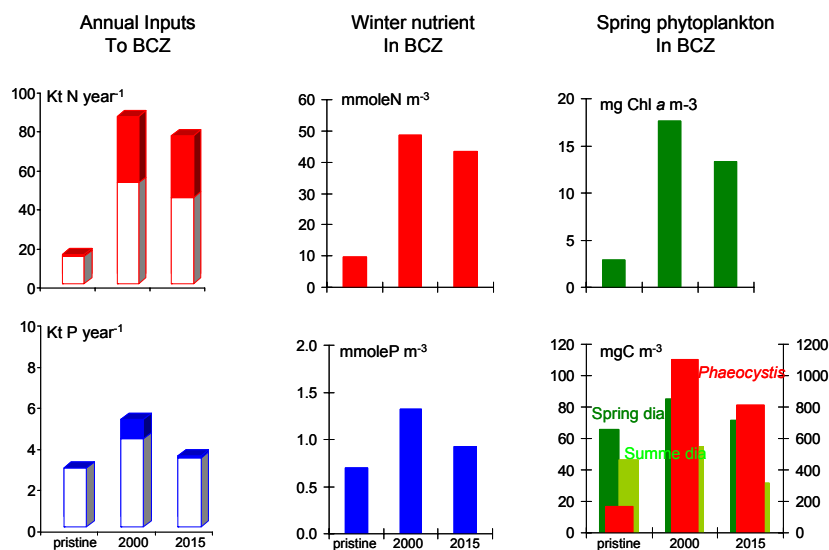

Figure 6.13. R-MIRO simulations of annual N and P inputs to BCZ (left panel; filled block: Scheldt; open block: Atlantic waters+Seine); winter N and P enrichment in BCZ (middle panel) and maximum phytoplankton biomass reached in spring (right panel) obtained for the pristine reference, the current situation (2000) and the 2015 projection.

6.5 Concluding remarks and perspectives

The sustained modeling effort in Belgium has provided mathematical models (0D-MIRO and 3D-MIRO&CO) that can be used for assessing eutrophication problems and addressing issues that cannot be resolved based on field studies. Being based on the current scientific knowledge, these models have their own limitations and need to be continuously developed. As pointed here the obtained simulation of *Phaeocystis* colonies in the Baie de Seine, an area where colonies have never been reported, will not be understood before a full resolving of the *Phaeocystis* life cycle and especially the identification of factors triggering the formation of colonies. In this way 0D-MIRO and 3D-MIRO&CO are complementary to field studies and cannot be substituted to them.

The validation of ecological models is recognized as critical for their reliable use in policy support (Radach & Moll, 2006), yet validation data is extremely sparse compared to the space, time and parameter coverage of these models. For most parameters, including nutrients and phytoplankton species, data is available only from a limited number of seaborne cruises and/or a single weekly

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monitoring location (*e.g.* Rousseau, 2000). For secondary trophic levels such as zooplankton, biomass data are even sparser. Good spatial and reasonable temporal coverage is provided by optical remote sensing but the parameter measured, Chl *a* concentration, is not a model state variable and gives no indication of the species composition of the phytoplankton community. Moreover, there are concerns about satellite chlorophyll data quality in coastal waters with high non-algae particle concentration or coloured dissolved organic matter absorption (Ruddick *et al.,* 2008). In the future, data for model validation should become more extensive. In this respect automated flow cytometer instruments could provide unparalleled temporal resolution of phytoplankton species composition. Satellite remote sensing data quality and quantity will improve considerably giving a reliable and almost complete description of surface Chl *a* concentrations. Combination of these two technologies and/or the possible improvement of remote sensing algorithms and hyperspectral sensors to discriminate between different phytoplankton species would greatly enhance the potential for model validation.

When coupled to the RIVERSTRAHLER model, a model calculating nutrient delivery to the sea as a function of change in land use and economical activity, these models can provide to decision makers a way of testing and evaluating the results of planned measures for nutrient reduction.

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