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THRESHOLDS
Thresholds of Environmental Sustainability
INTEGRATED PROJECT

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THRESHOLDS – a mid term synthesis

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THRESHOLDS – a mid term synthesis

Contents

1 Thresholds – an emerging research theme and an Integrated Project

Thresholds of environmental sustainability refer to the pressures that can be imposed on a given resource while maintaining acceptable levels of environmental quality. Whenever these thresholds are exceeded, resources, services or functions of the affected ecosystem may suddenly shift into states from which they cannot easily be brought back. These "points of no return" pose major challenges for the management of ecosystems and their resources.

Most policies for environmental protection consider, often implicitly, that the responses of ecosystems to natural and human pressures (e.g. nutrient loads, chemicals, fishing, climate change) lead to gradual changes that can be reverted through a corresponding decrease in pressure levels. Yet, ecosystems , as systems composed of multiple interacting components, are complex and prone to non-linear behaviour. Such system do not necessarily recover after a decline in pressures. Time lags and partial irreversibility are caused by a reorganisation in the system functioning, leading to what is known as hysteresis. Existing buffers are lost and new mechanisms and system components, along with new buffer mechanisms that resist further change, develop.

Despite overwhelming evidence that ecosystems respond to pressures through non-linear behaviour and regime shifts, the available scientific tools, current research approaches and existing policies are not yet fully able to deal with the phenomena. Explicit consideration of non-linear changes will improve our understanding of the response of ecosystems to pressures, and will also provide a scientific basis for policies that can avoid driving ecosystems to their breaking points.

The key objective of the THRESHOLDS project is to develop thresholds science in the context of coastal ecosystems, which are frequently characterised by threshold behaviour. The science required to identify thresholds must be quantitative and predictive, and be able to cope with the complexity and non-linear behaviour of environmental systems.

Policies for sustainable development must maintain systems within the limits imposed by the thresholds and identify indicators for this purpose. Effective policies require target setting processes that integrate scientific knowledge of thresholds with indicators of change, an ability to link socio-economic activities to these indicators, and an awareness of the externalities that are associated with these activities.

The THRESHOLDS project faces complex behaviour of ecosystems, such as regime shifts between alternative stable states, and complexity in valuation of the sectors affecting environmental quality. Non-linear cost-pressure relationships and multisectoral interactions are analysed to develop procedures that accommodate the complexity of the socio-economic and environmental systems. In this way procedures can be developed that avoid limitations and oversimplifications and deliver predictive knowledge that is required to formulate effective policies for Sustainable Development.

This report provides an overview of highlights of the research conducted within the first two years of the THRESHOLDS project. It is aimed at policy-makers and managers who can gain understanding of the links between thresholds and policy formulation and implementation, and to scientists, who can utilize this interdisciplinary overview to consider the future phases of thresholds studies.

2 The significance of thresholds

2.1 Thresholds and eutrophication

Eutrophication caused by nutrient loading frequently leads to a number of adverse consequences in aquatic systems. Depending on local and regional conditions the main pressure can be due to phosphorous or nitrogen loading, or a combination of both. The changes in recipient systems are seldom smooth but display various forms of non-linear behaviour.

2.1.1 More Harmful Algal Blooms with Increasing Nutrients

Within THRESHOLDS several studies focus on the nature of harmful algal events. Data on harmful algal events and blooms can indicate shifts in coastal and marine waters. The following examples show that the causes and underlying mechanisms differ between systems.

For a number of areas in Danish marine waters winter phosphate concentrations were correlated with the average number of days when mussel fishing is banned due to Diarrhetic Shellfish Poisoning (DSP) caused by algae (*Dinophysis* spp.). Although not statistically significant, the average number of mussel closure days increases with increasing winter phosphate concentrations. There are indications of a non-linear relationship (Figure 1).

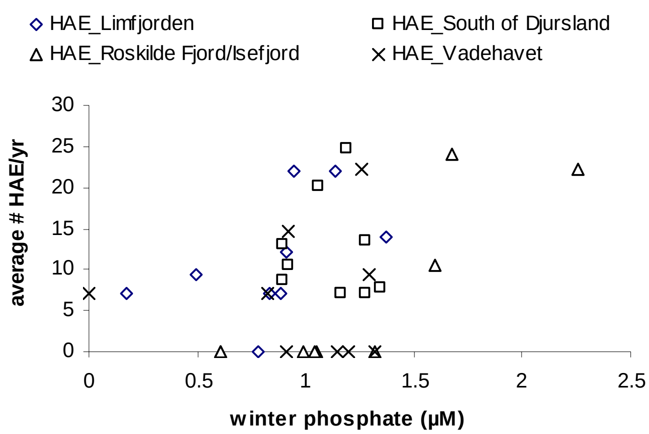


Figure 1. Average number of mussel closure days per area and the average surface concentration of phosphate in winter.

On the North Sea Coast *Phaeocystis* colonies form recurrent high-biomass harmful algal blooms (HAB) in the Eastern Channel and Southern Bight of the North Sea. These blooms develop in spring between the early-spring and summer diatom blooms. The effect of climate variability (NAO index) and human activities (Scheldt nutrient loads) on the magnitude of diatom and *Phaeocystis* blooms in the Belgian Coastal Zone (BCZ), was investigated by conducting a serial statistical analysis based on a comprehensive suite of nutrient loads, hydrometeorology, and phytoplankton data collected at St 330 (located in central BCZ; Fig. 2 left) between 1988 and 2001 (Breton et al., 2006). This analysis concluded that the long-term diatom biomass and the spring dominance of *Phaeocystis* colonies over diatoms are determined by the combined effect of the North Atlantic Oscillation (NAO) and freshwater and continental nitrate carried by the Scheldt (Breton et al., 2006). A nonlinear but monotonic relationship is found between Phaeocystis colony bloom magnitude and winter NO_3 enrichment at St 330 (Figure 2). Such a link was not found for winter PO4 enrichment pointing the key role of NO_3 in determining the height of Phaeocystis colony blooms. This result is consistent with previous studies that demonstrate a positive link between Phaeocystis cell density and NO_3 excess after the early spring diatom bloom controlled by PO_4 and $Si(OH)_4$ (Lancelot et al.,1998; Rousseau, 2000)

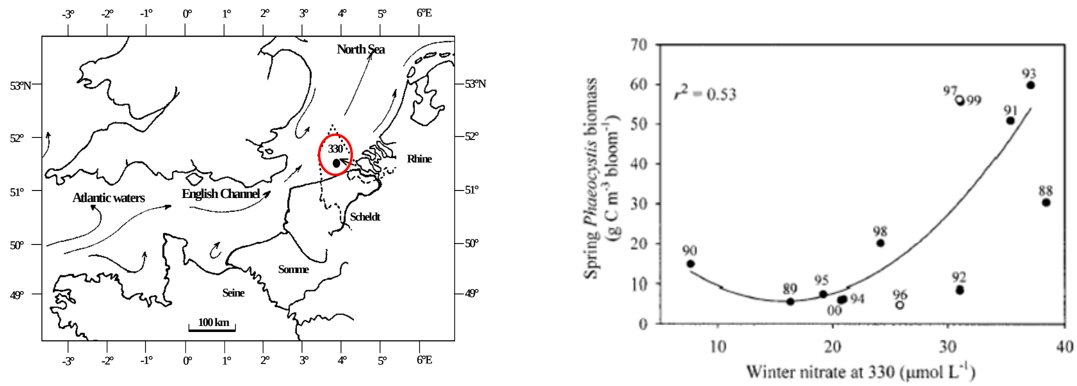


Figure 2. Left: Map of the Southern Bight of the North sea with location of St 330 in the central Belgian coastal zone. Right:. Relationship between spring *Phaeocystis* colony bloom (g C m^{-3} $bloom^{-1}$) at St 330 and winter NO_3. Solid line represents quadratic model fit.

2.1.2 Regime Shifts in Benthic Communities with Hypoxia

A linear regression model describing bottom water oxygen concentrations in Danish coastal waters showed significant links between total nitrogen input from land, the advective transport of water and Skagerrak surface water temperature and oxygen concentrations. The differences between the predicted and modeled oxygen concentrations suggest a regime shift (*Figure 3*). The residuals fall into two distinct groups. Prior to 1985 there was always more dissolved oxygen in bottom waters than predicted, whereas after 1985 there was always less dissolved oxygen than predicted by the model. This change in the residuals indicates a regime shift occurred around 1985 in the benthic community from an ecosystem with less frequent oxygen depletion, to an ecosystem that is more susceptible to nutrient-driven eutrophication and hypoxia (Conley et al. 2006).

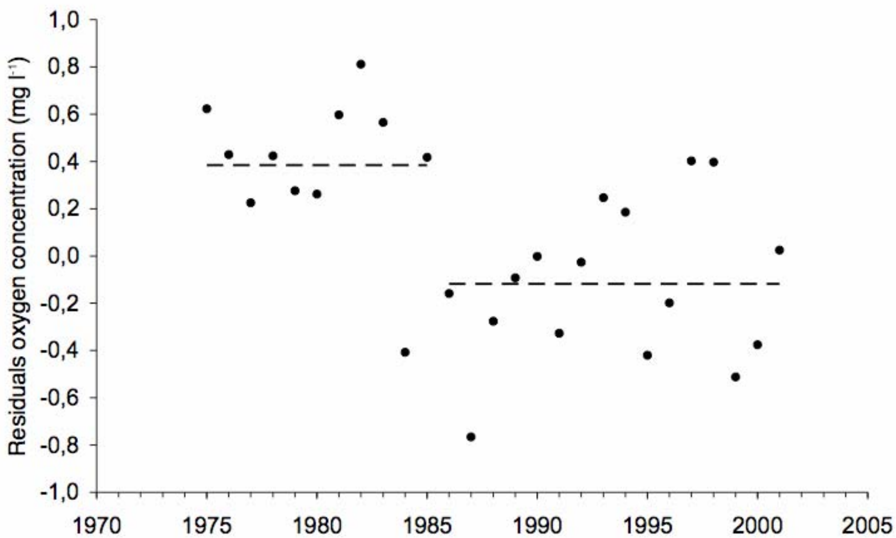


Figure 3. Residuals of oxygen concentration from the multiple regression including total nitrogen (TN) input, advective transport and temperature in the Skagerrak surface water. Dashed lines show the average of the residuals for 1975-1985 and 1986-2001 (Conley et al. 2006).

2.1.3 Loss of Seagrass Communities

Sedimentary organic and nutrient inputs have been identified as important drivers of seagrass decline. Segmented regression equations suggest thresholds of sedimentary inputs above which seagrass decline accelerates (*Figure 4*). The value of the threshold extracted from the analysis is 0.016 gP m^{-2} d^{-1} and is often exceeded near fish farms.

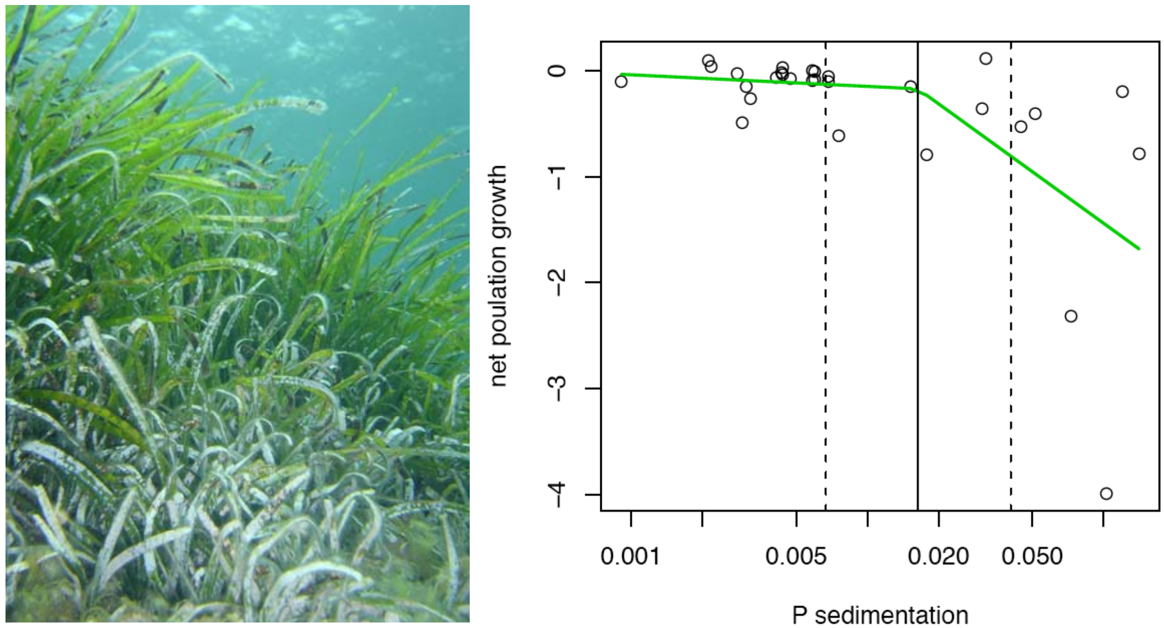


Figure 4. Relationship between phosphorous input rates (P sedimentation, g m^{-2} d^{-1}) and Posidonia oceanica net population growth rate (net population growth, yr^{-1}). The green line represents the fitted segmented equation. Black solid and dashed lines show the break point of the equation (i.e. threshold) and its 95 % confidence limits.

2.1.4 Loss of macroalgal cover

Eutrophication is a major threat to plant communities growing on the sea floor. Increased nutrient concentrations stimulate the growth of phytoplankton and epiphytic algae which in turn shade seagrasses and macroalgae. However, there have been few studies of the quantitative relationships between nutrient concentration and macroalgal cover in coastal waters.

An analysis of a large data set from Danish coastal waters demonstrates that the cover of macroalgal communities in deeper water decreases markedly along an eutrophication gradient. Total algal cover generally decreases 2.2% per 1 µM increase in TN (Figure 5).

The analysis indicates that algal abundance initially responded slowly to increasing eutrophication but showed a more marked response at nitrogen concentrations around 35-40 µM. The existence of possible threshold nutrient levels demands further analyses using the threshold tools developed in the project.

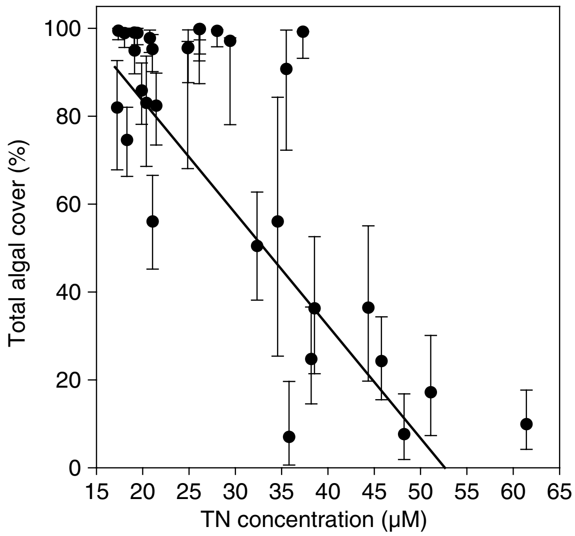


Figure 5. Total algal cover versus total nitrogen (TN) concentration. Filled circles represent area-specific means from various coastal areas. Regression line: Total cover=1.347-0.0226TN (R^2=0.73, p<0.0001).

2.2 Combined effects of nutrient loads and overfishing on a coastal pelagic food web

The effects of human activities on coastal ecosystems were investigated by analysing the pelagic food web structure of temperate coastal water ecosystems. The generic structure was established based on literature review and studied by qualitative structural network analysis making use of two indicators: the bottom-up keystone index and the mixed trophic impacts (Vasas et al., submitted). Two main questions were addressed: (1) the role of species forming harmful algal blooms (HAB) and red tides (*Noctiluca* sp) and jellyfish in eutrophied systems and (2) the contribution of bottom-up (increased nutrient loading) and top-down (overfishing) human influences on food webs.

Results from modeling (Vasas et al., submitted) suggest that HAB-forming species and *Noctiluca* stimulate the microbial network and inhibit higher trophic levels like commercially important fish. Jellyfish acts as a buffer in eutrophied and overfished systems as it retains nutrients from the water column, but its blooms lead to a massive accumulation of large phytoplankton. Human-induced nutrient enrichment favours undesirable species (Figure 6) because of their crucial position in the food web which may explain their far-reaching effects. Finally, while the earlier overfishing of piscivorous fish appears to have inhibited blooms of harmful species and supported blooms of diatoms and other large algae, the present-day loss of planktivorous fish supports HAB-species, *Noctiluca* and jellyfish, acting synergistically with nutrient enrichment.

The results, supported by biological time series in the North Sea and Black Sea ecosystems, highlight fundamental constraints that are inherent in the generic structure of pelagic food webs, thus help to understand the general mechanisms behind eutrophication and overfishing. Finally, our results underline the importance of the food web structure in the functioning of the ecosystem and demonstrate that the very basic « who eats whom » information can tentatively explain observed long-term changes in time series.

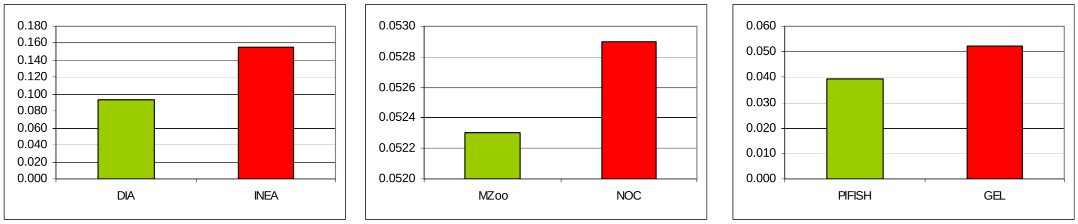


Figure 6. Mixed trophic impacts of nutrient enrichment on harmful groups Inedible algae (INEA), Noctiluca (NOC) and jellyfish (GEL) in comparison with their respective 'desired' competitors diatoms (DA), mesozooplankton (MZOO) and planktivorous fish (PlFish) .

2.3 Toxic effects

Several organic contaminants and metals are known for their environmental persistency, ubiquity and their toxic and/or carcinogenic effects. Furthermore, they may bioaccumulate in organisms and ecosystems, including humans. The thresholds of effects are of considerable importance. Toxicology was one of the first fields to apply the concept of threshold in ecological systems using dose-response curves. This is because the main objective in both human an environmental toxicology is the estimation of a threshold concentration above which toxic effects occur (Figure 7).

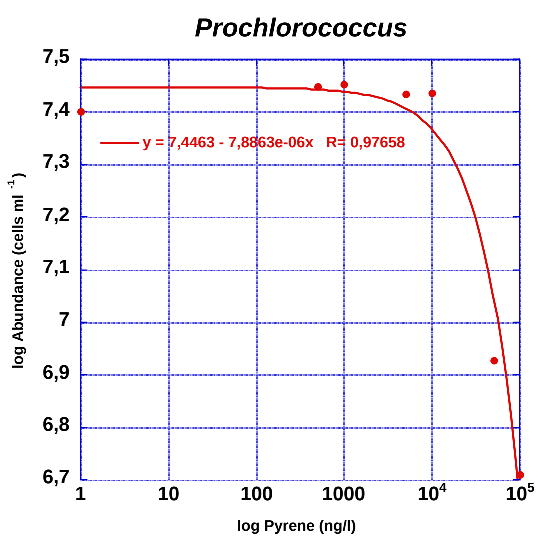


Figure 7. Variability of *Prochlorococcus marina* cell abundance (cells ml $^{-1}$) growing under a gradient of Pyrene dissolved in DMSO. The lethal Pyrene dose required to decimate the population of *Prochlorococcus marina* to the half, was 80,000 ng/l.

Adverse effects can develop at the organism level affecting for example reproduction or swimming behaviour. Molecular level effects may develop because environmental stresses such as chemical pollutants, temperature change and low nutrient levels, may induce gene expression changes, which are translated in the structure and function of cells and then to the organism.

2.4 Economic consequences

Ecological threshold effects have economic consequences by affecting ecosystem services. The consequences of harmful algal blooms, foodweb disruption and jellyfish abundance, anoxia and *Posidonia oceanica* decline have been examined in an economic framework. It is essential to consider not only the direct losses associated with a change, but also time lags, uncertainty and points of no return.

In Belgium a survey has been completed and preliminary results indicate a willingness to pay of €43.57 per year to reach a level of low algal bloom, while beach users request a payment of €144.23 to accept the present status quo showing that the present situation causes constant losses of welfare.

3 Identifying thresholds, regime shifts and their consequences

3.1 Methodological aspects of detecting thresholds and regime shifts

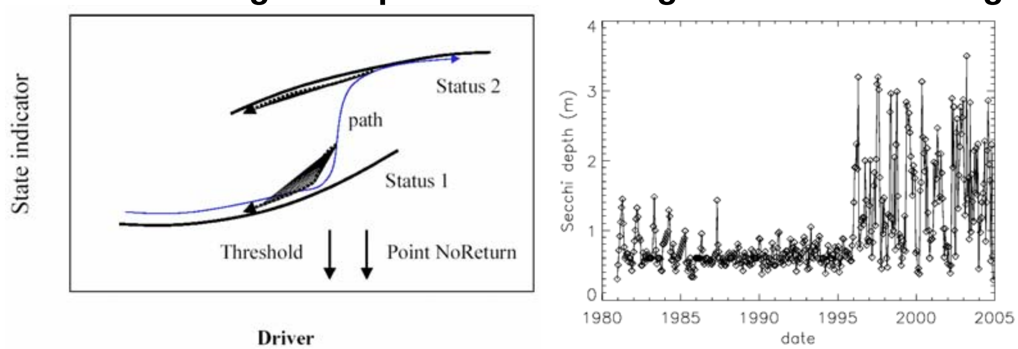


Figure 8. Left: Conceptual illustration of a Threshold in a driver-response curve. Right: Time series from Ringkøbing Fjord showing a change of regime.

A classical figure illustrating the concept of an ecological threshold is depicted in the left panel of Figure 8. A real example of a threshold phenomenon is presented in the right one, which displays a time series of Secchi depths, as a measure of water transparency, from Ringkøbing Fjord, Denmark (one of the case-study areas in the project). The change in regime illustrated in this diagram is quite different from the one in the right panel. In addition, time series are more directly observed than driver-response relationships. This example highlights that threshold phenomena are diverse and can be represented in several ways. This also means that a number of different techniques can and should be used in attempting to identify thresholds in historical data (Table 1).

The mathematical theory of bifurcations, a classification of threshold phenomena in terms of the qualitative features that would appear in the corresponding time series, and their links to driver-response curves, help to understand the nature of thresholds. Examples of theoretical ecosystem models leading to the different types of thresholds have been compiled. Figure 9 shows two of these examples. The left panel gives a time series of phosphorus data from a eutrophication model with increasing nutrient input and an abrupt change of regime. The right panel in Figure 9 illustrates a type of change of regime more similar to the Ringkøbing Fjord case illustrated in Figure 8. In this case, the change occurs in the abundance of a prey species in a model ecosystem when decreasing predation below a threshold.

Table 1: Threshold mechanisms expected in coastal ecosystems with an appropriate statistical method for identification and significance testing.

Threshold indicator	Statistical method for identification
Change in gradient	Non-linear estimation methods for non-linear functional relationships including piecewise linear relationships
Abrupt change in mean	Smooth threshold (non-linear regression) or change-point detection method (t-test type statistics or likelihood test statistics or non-parametric)
Change in seasonal pattern	Change-point detection method (F-test type statistics or likelihood test statistic) for seasonal model (e.g. seasonal means or harmonic function)
Change in variance	Change-point detection method (F-test type statistics or likelihood test statistic)
Change in functional relationship	Change-point detection method (F-test type statistics or likelihood test statistic)
Change in correlation structure	Singular Spectrum Analysis (SSA)

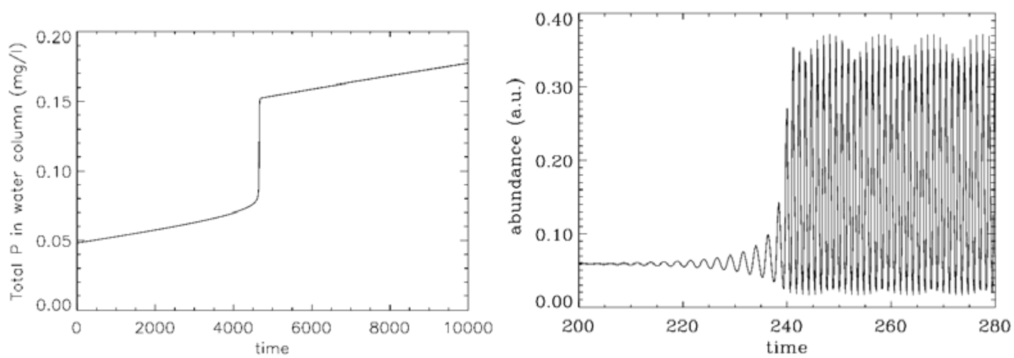


Figure 9. Left: Phosphorous changes using a model of lake eutrophication. Right: Change of regime in the abundance of a prey species in a model ecosystem (Holling-Tanner) with changing predation.

Systems crossing an ecological threshold will exhibit several characteristics, such as changes in mean values and variances of individual system components, as well as changes in the mass flows and functional relationships between them. Most of the reported structural changes in marine systems have been inferred just from apparent step changes in the mean values of time series of observations. While a change in the mean value of a single component can be a simple and intuitive indicator of structural change, the proper statistical testing of the existence of such a change-point is a non-trivial task which requires quantitative tools for separating random fluctuation from actual change.

The example below uses a technique where a model with constant mean value (i.e., no change) is compared to a model where the mean value has a step change in time. The improvement in the model with a change-point can be measured as the increase in goodness of fit, which is equivalent to a decrease in the sum of squared deviations from the mean. This decrease can be expressed as an F-ratio between the residual mean squares of the two models. If we compute this F-ratio for all candidate change-points (i.e., all observation times) the most likely change-point will reveal itself as the one with the highest F-value. We will conclude that the change-point is statistically significant if this F-value is above a certain critical level. The actual change-point estimates for the example below are March 1995 for salinity, October 1995 for chlorophyll, and November 1995 for Total Phosphorous (TP).

Statistical tools that can detect thresholds include a breakpoint analysis (*Figure 10*). The upper panels (A-C) are time series of salinity, total phosphorus (TP) and chlorophyll showing dramatic changes in TP and chlorophyll following the salinity change causes by increasing the water exchange rate with the sea. The breakpoint analysis (lower panel D – F) identifies the most likely time for a structural change in the mean of the time series. The actual change-point estimates are March 1995 for salinity, October 1995 for chlorophyll, and November 1995 for TP. A closer look at the chlorophyll to phosphorus relationship (*Figure 11*) reveals a significant structural change, but delayed 8 months to July 1996. The figure to the right shows the two regression lines for log-transformed chlorophyll versus total P (red symbols: before, black symbols: after). The change in regression slopes probably reflects both a change in the degree of light limitation and in the turnover rate of phytoplankton, following increased grazing by clams after the salinity change.

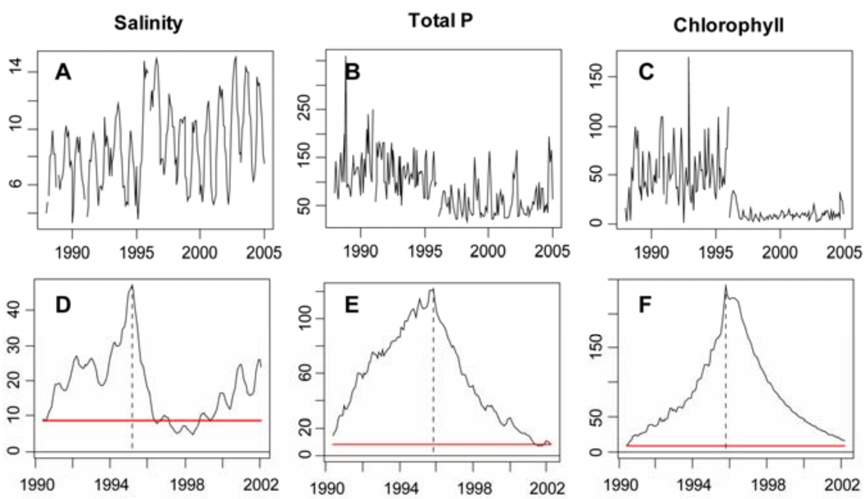


Figure 10. A Breakpoint analysis for threshold identification using monthly data from Ringkøbing Fjord 1988 – 2005.

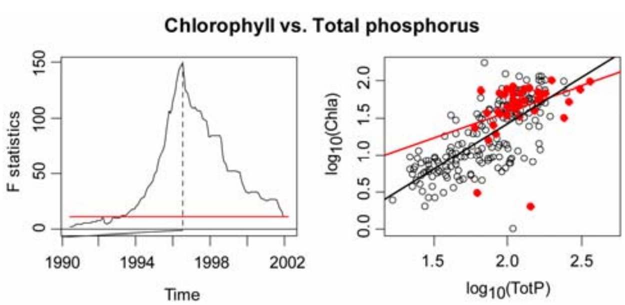


Figure 11. Chlorophyll versus total phosphorus in Ringkøbing Fjord.

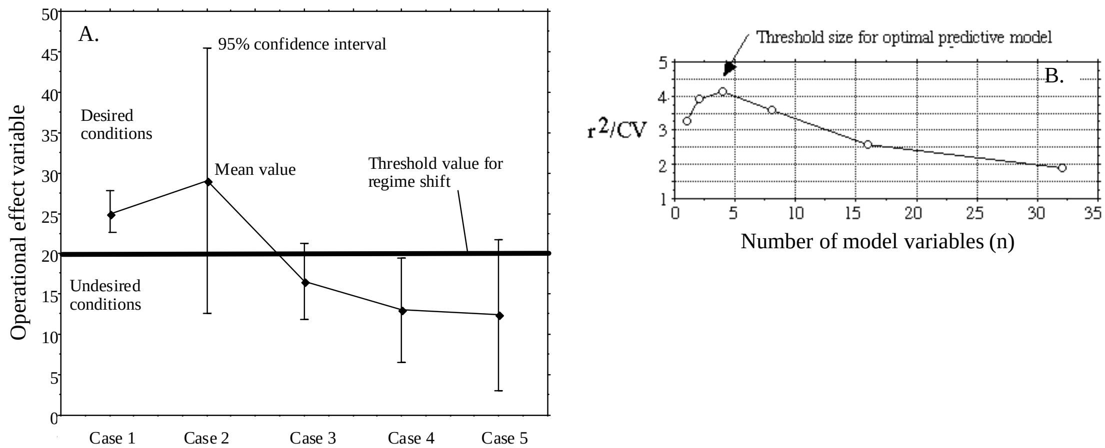


Figure 12. Uncertainties in data and statistical, confidence levels and predictive power of management models.

How do inherent variations and uncertainties in empirical data constrain approaches to predictions and possibilities to identify critical thresholds and points of no return? We address the following

- Problems related to the balance between the changes in predictive power and the accumulated uncertainty as model grow in size (see Figure 12B);
- How inherent uncertainties in empirical data depend on sampling effort and useful strategies to reduce inherent uncertainties (for example expressed by the coefficients of variation, CV = SD/MV; SD = standard deviation and MV = mean value) in empirical data, regressions and approaches to maximize predictive power of regression models (by transformations and by creating mean values for different time periods); and
- Systems in variations in standard water variables within and among coastal systems.

We have introduced the concept "Optimal Model Size" (OMS), and an algorithm to calculate OMS, which accounts for key factors related to the predictive power at different time scales (daily to yearly prediction) and uncertainties in predictions in relation to access to empirical data and the work (sampling effort) needed to achieve predictive power at different time scales. Figure 12A gives mean effects and 95% confidence intervals in 5 cases in relation to a given threshold value.

Case 1 shows a condition where the observed mean effects is statistically significant and above the ecological threshold; case 2 gives a situation in which the effect appears to be above the threshold, but it is not statistically significant. In cases 3 and 5 the effects below the threshold but they are not statistically significant. In case 4 the effect is below the threshold and statistically significant.

Figure 12B shows another key principle related to the usefulness of models. For any target variable in water management, there is an optimal size of the predictive model, as given by the ratio between the r^2-value when modelled values are compared to empirical data, and the accumulated uncertainty in the model (CV). The basic aim is that "everything should be simple as possible, but not simpler".

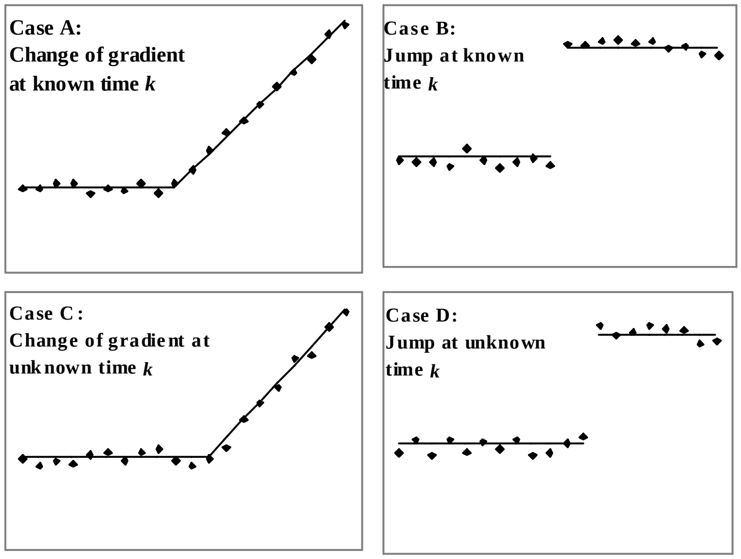


Figure 13. Illustration of basic thresholds mechanisms. The key issue is whether the timing of the change is known or not.

Models of thresholds are inherently non-linear. Methods for detecting four general types of threshold mechanisms have been synthesized and described (Figure 13). In cases A and B, the time of change or threshold value is known and the significance of the thresholds can be investigated within the framework of generalized linear models. Such models (A and B) may be relevant when the hypothesized threshold is related to an action taken (e.g., nutrient reductions or change of sluice practice) or an event occurring (e.g. hurricane or accidental release of harmful substances).

However, in many cases the timing of the threshold can also be unknown (cases C and D). The continuous relationship in case C can be analyzed as a non-linear model with 4 parameters, whereas the jump function in case D has to be analyzed within the framework of change-point detection methods. Bloom intensity in relation to hypoxic conditions prior to the bloom is a typical example of case C (Figure 14).

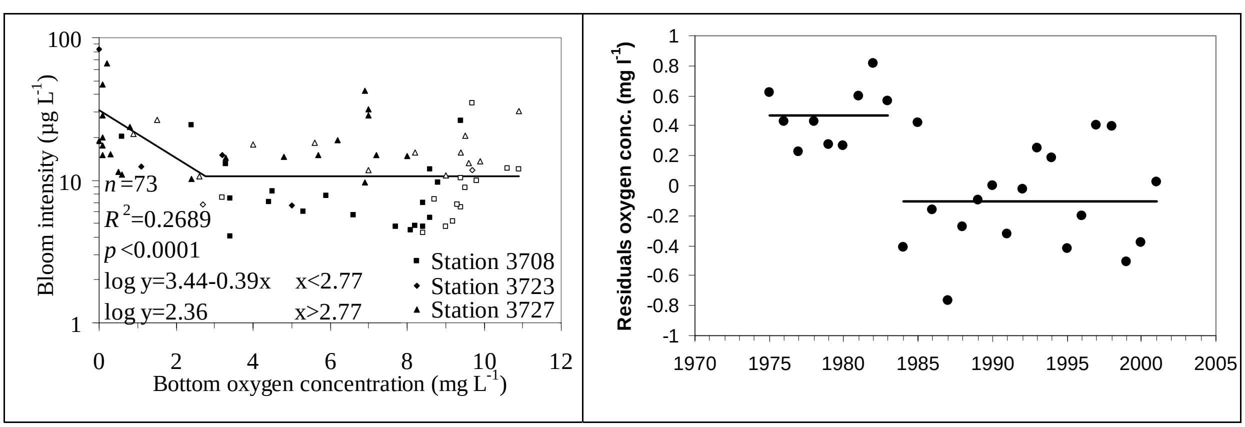


Figure 14. Intensity of algal blooms in relation to oxygen concentration.

3.2 Detecting regime shifts in planktonic ecosystems related to nutrient thresholds

The spatio-temporal resolution of North Sea and Black Sea biological time series that has been collected in the project was found insufficient for applying sophisticated statistical tools for detecting regime shift. As an alternative, we used long term model simulations and trophic criteria to identify nutrient thresholds of planktonic ecosystem changes. As a first step, the methodology was tested in the eutrophied Belgian coastal waters [BCZ, Southern Bight of the North Sea submitted to direct (Scheldt) and indirect (Somme, Seine) river nutrient loads] dominated by Phaeocystis colony blooms. For this purpose the coupled RIVERSTRAHLER-MIRO model (Lancelot et al., 2007) was run over the 1950-2000 period and included two additional scenarios mimicking pristine time and 2015 (i.e. after the implementation of the WFD by Belgium and France). Two ecological criteria were used: one based on maximum observed biomass reached by grazable Phaeocystis colonies and the other based on trophic efficiency estimated from annual model simulation of primary and secondary production. Results obtained show that, except for the pristine scenario, the simulated Phaeocystis colony biomass is always greatly above the identified threshold of 150 mgC m-3 (Figure 15).

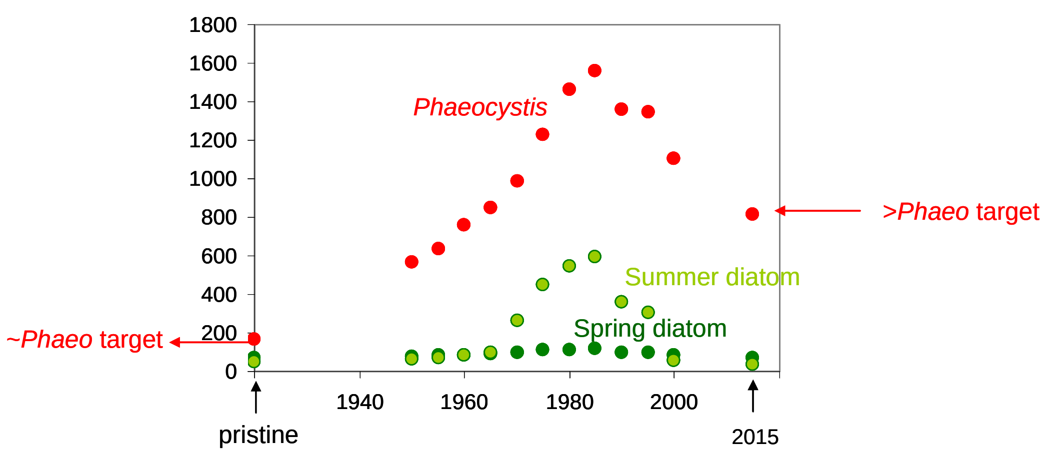


Figure 15. RIVERSTRAHLER-MIRO simulations of maximum diatom and *Phaeocystis* colony biomass (mgC m^{-3}) reached between 1950 and 2000 and for pristine and 2015 scenario.

Long-term analysis of the simulated trophic efficiency, expressed as ratios between diatom production and total primary production (Figure 16, left) or copepod production and total primary production (not shown), in relationship with the concomitant variations of simulated winter NO3 and PO4 enrichment points the N:P winter ratio as an indicator of trophic shift (Figure 16, left). This critical shift in N:P is explained by the strong decrease of PO4 loads in recent years while elevated NO3 loads were maintained. The comparison between changes in winter N:P-ratios and NO3 loads to BCZ estimated from budget calculation of model simulations suggests that total NO3 inputs higher than 60 kT N year^{-1} are critical for the BCZ ecosystem.

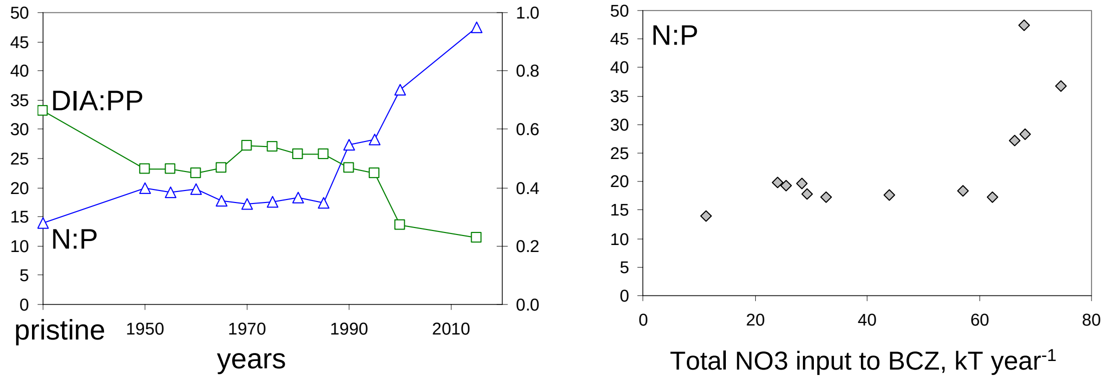


Figure 16. Left: Comparison between simulated winter N:P ratios and diatom: total primary production (DIA:PP). Right: Relationship between simulated N:P winter ratios and the simulated total NO3 inputs to BCZ.

3.3 Contaminants and thresholds

Several families of organic contaminants and two metals demonstrate the methodological approaches that have been used and developed in the study of thresholds related to contaminants. The contaminants include: polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers PBDEs). The two selected metals are cadmium (Cd) and mercury (Hg).

3.3.1 Analysing the effects of contaminants

The effects of contaminants can be analysed at several levels. The classical way to investigate (eco)toxicological effects of environmental pollutants is to look for acutely toxic effects. THRESHOLDS explores methodologies that allow effects to be determined at several levels by focusing on coastal organisms, using individual species as well as mesocosms experiments with species mixtures.

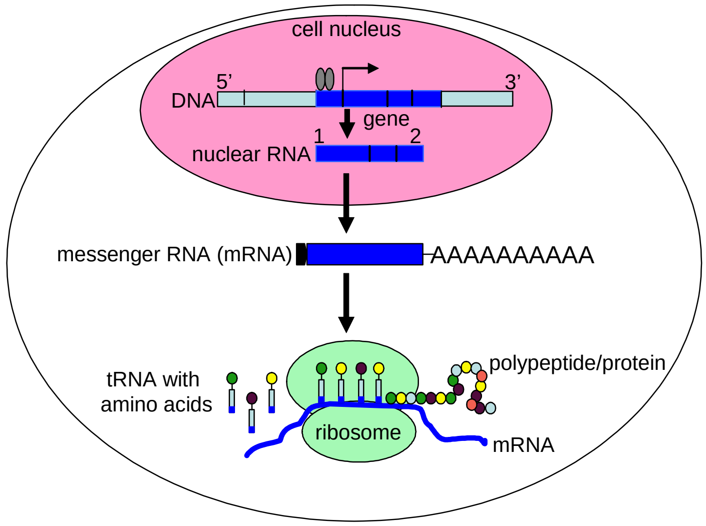


Figure 17. Principle of genetic transcription and translation (in eukaryotic cells. Contaminants may affect at different steps in the whole process.

Molecular level effects

The expression of genetic information is the first step in a series of cellular processes that are translated into structure and function of cells and then to the organism (

Figure 17). Environmental perturbations and stresses (e.g. chemical pollutants, temperature change, low nutrients etc.), may induce gene expression changes. These changes could thus be used as early indicators of effects on the organism.

The effects of polycyclic aromatic hydrocarbons (PAHs) in the marine diatom Thalassiosira pseudonana was analysed for gene expression level changes. Our experiments showed that some genes involved in the formation of the silica shell of the diatom as well as genes involved in photosynthesis, were down-regulated by PAH exposure. This could impair the cellular integrity as well as energy production.

Toxic effects at individual/population level

Laboratory experiments were performed with phytoplankton cultures to analyze the lethal concentration of PAH's for populations of these organisms. Solutions of pyrene and phenantrene dissolved in DMS were added. The results indicate that most species were highly resistant to pyrene and phenantrene, except Prochlorococcus and Synechococcus, which showed lethality at high concentrations of pyrene and phenantrene. The threshold concentration of pyrene was 80 μg L-1 for Prochlorococcus, whereas for phenantrene it was between 100 μg L-1, (toxic) and 500 μg L-1, (lethal), see Figure 18.

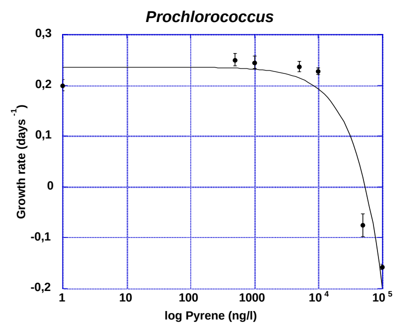


Figure 18. Variability of Prochlorococcus marina growth rate growing under a gradient of Pyrene concentrations dissolved in DMSO.

Experiments on natural communities

Experiments to analyze the lethal threshold of PAH's and metals on natural populations of phytoplankton were performed with coastal Mediterranean plankton. Preliminary results indicated that pyrene and phenantrene appeared to be toxic above 50 μgL-1 and lethal at 100 μg L-1 in both Prochclorococcus sp and Synechococcus sp populations. Metals also caused toxic effects at concentrations ranging from 1.12 ppb for cadmium and 12 ppb for lead. Lethal concentrations were 12 ppb for cadmium and 112 ppb for lead. These levels are well above the concentrations normally observed in waters, which have been 3.4-30.4 ng/l)[1].

The role of zooplankton in contaminant cycling

Even though several studies have dealt with the bioconcentration of polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in phytoplankton, much less is known of processes driving POP accumulation in zooplankton. Indeed, it is not clear whether zooplankton, and other heterotrophic organisms, accumulate POPs from feeding on phytoplankton or passively from surrounding waters. Furthermore, it still remains unclear whether these POPs introduced in zooplankton by dietary uptake biomagnificate, or are eliminated through egestion of fecal pellets or metabolization. As part of the THRESHOLDS project, uptake and depuration constants have been measured for PAHs in zooplankton (Figure 19).

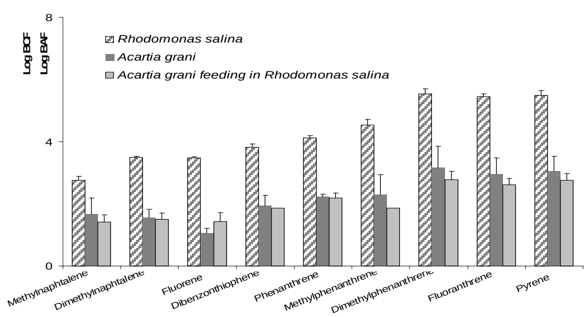


Figure 19. Bioaccumulation factors of PAHs in phytoplankton (Rhodomona Salina), zooplankton due to diffusive uptake and zooplankton feeding on phytoplankton.

[1] (http://www.nature.nps.gov/hazardssafety/toxic/benzoepy.pdf)

Table 1: Overview of thresholds of no effect reported in the present study. AFM: Assessment Factor Method; SEM: Statistical Extrapolation Method; EqPM: Equilibrium

Chemicals	Thresholds of no effect for			Overall threshold of no effect for the marine environment
	Pelagic organisms	benthic organisms	marine top predators	
	ng.L^{-1}	µg.kg^{-1} dw	µg.kg^{-1} ww	ng.L^{-1}
PCB 105	-	-	-	-
PCB 118	-	-	1.1 (low reliability)	-
PBDEs	53	310	333	12.2 (←top predators)
Fluoranthene	100	155	11,500	32 (←sediment)
Benzo[a] pyrene	5	-	-	0.005 (←seawater)
Benzo[k]fluoranthene	3.6	60 (AFM) 38-537 (EqPM)	-	0.4 (←sediment)
Benzo[g,h,i] perylene	0.8	-	-	0.8 (←seawater)
Cadmium	1.6 (AFM) 210 (SEM)	-	-	210 (←seawater)
Mercury	1(not relevant)	-	-	-

Determining the level of no effects

Chemicals effects assessment comprises two steps of the risk assessment procedure: hazard identification and dose-response assessment. The Predicted No Effect Concentration (PNEC) is determined as the thresholds below which no unacceptable effects are detected. The methodology to derive PNECs is based on a European consensus (European Commission, 2003).

The approach has been applied to the selected substances in the project. Unfortunately, there is not sufficient data available in the literature to perform a complete assessment. Table 1 summarizes the main findings that have been obtained so far.

3.3.2 Spatio-temporal variability of pollutants in coastal systems

High temporal variability and spatial patchiness have been observed in the occurrence of pollutants in the marine environment and atmosphere. This variability affects the threshold level and has practical consequences for pollutant monitoring. Furthermore nonlinearities present at several levels affect the pollutant distribution and, hence, the risk of exceeding a threshold.

Phase speciation is also linked to the entry route. Diffuse air-water exchange and wet deposition are important pathways for the input of pollutants to aquatic environments. A combined approach using monitoring and modeling is being developed to gain better understanding of the variability

Monitoring has been carried out to support the development of contaminant thresholds at Thau lagoon (France), Far Ses Salines (Mallorca, Spain), and during a Mediterranean cruise (Figure 20, Figure 21).

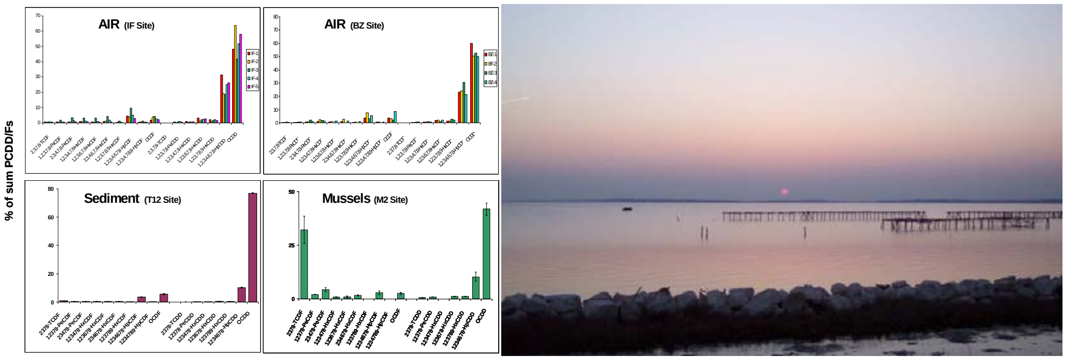


Figure 20. Etang de Thau (France). Air-Sediments-Mussels concentrations of PCBs and PCDD/Fs.

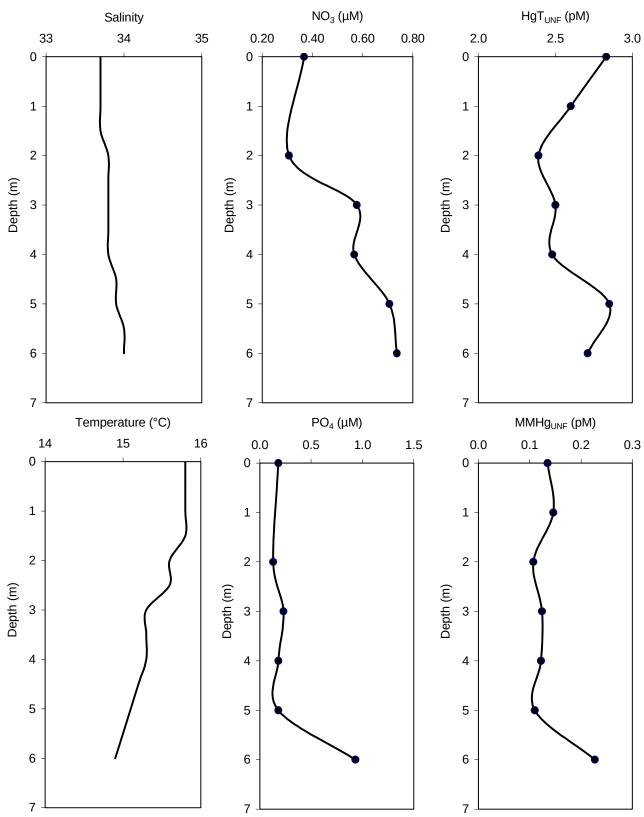


Figure 21.Vertical profiles for salinity, temperature, nitrate, phosphate, total mercury (HgT) and monomethylmercury (MMHg) in unfiltered samples of the water column at station C5 of the Thau Lagoon.

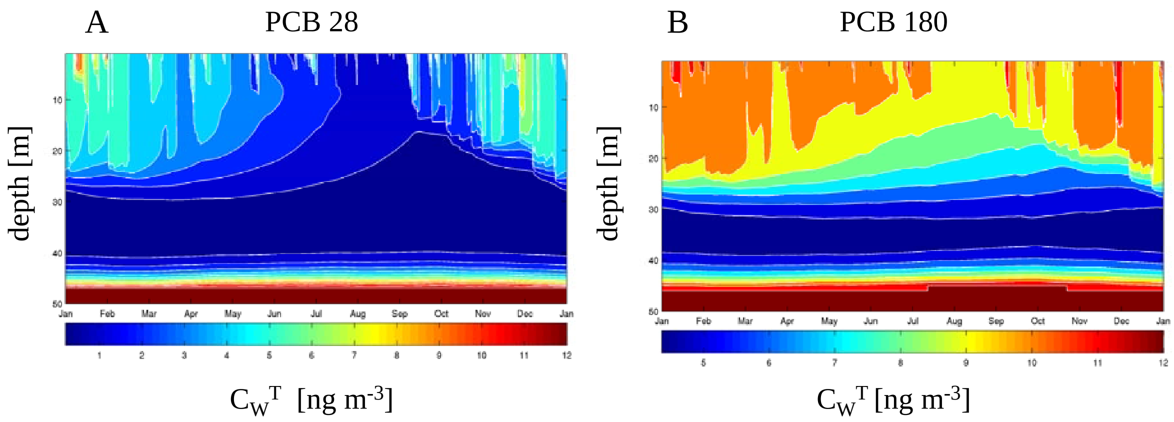


Figure 22. Depth-time distribution of total concentration in the water column for PCB 28 and PCB 180

In order to study the spatio-temporal distribution of contaminants a one-dimensional dynamic coupled hydrodynamic-contaminant model has been developed and applied to a Mediterranean continental shelf environment. In this case, the effect of vertical turbulent mixing on the dynamics of Persistent Organic Pollutants has been assessed. The simulation highlights the role of the turbulent processes in determining the POP distribution and variability in the water column (Figure 22).

A turbulent flux of contaminants strengthens the upward diffusion of contaminants in the sediment and determines the extent to which inputs from the atmosphere mix into the water column. It acts in parallel with degradation and sinking fluxes The combined effect yields a surface enriched – depth depleted – benthic layer enriched region distribution. The model results are very similar to reported experimental measurements. This model is being coupled with an ecological model to analyse the effects at ecosystem level.

3.3.3 Combined effects of pollutants and nutrients

Several mesocosms experiments (12 days) have been performed, to investigate the combined effects of pollutants and nutrients on natural communities. Pyrene had different effects in nutrient enriched and non-enriched communities. The effects were stronger in the enriched community in all the investigated trophic groups, but at different points in time (Figure 23, Figure 24). Despite the almost double amount of chlorophyll-*a* in the enriched community, concentrations of chlorophyll-*a* decreased to the same levels as the non-enriched community during the first day after Pyrene exposure, indicating a more severe effect and quicker rate of decline in the enriched community than in the non-enriched community.

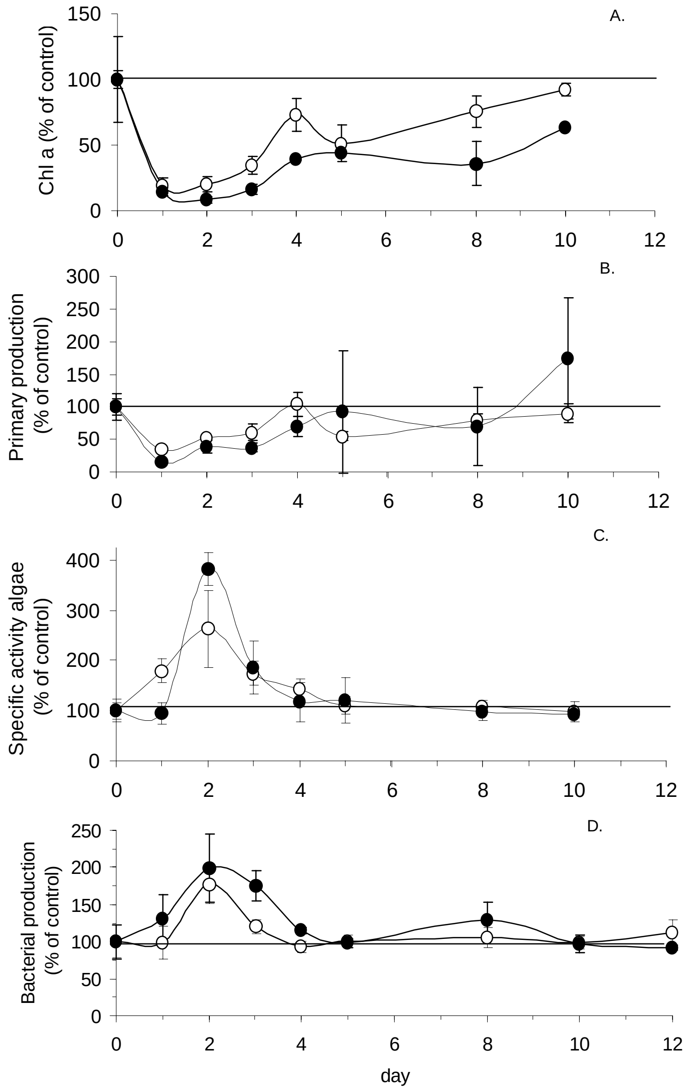


Figure 23. Development of variables in communities exposed to 50 nM (10.1 µg L-1) pyrene as % of control during the mesocosm experiment. Chlorophyll a (A), phytoplankton specific activity (B), total community activity of phytoplankton (C) and bacterial production (D). Open circles are non-enriched communities and closed circles are enriched communities. Data are shown as means ± SD (n=3).

Under the investigated conditions of biomass and nutrient status, there were no dilution effects of Pyrene due to higher initial biomass. Diatoms showed to be the least sensitive algae to Pyrene exposure in both non-enriched and enriched communities. Indirect effects on zooplankton caused a lower abundance of zooplankton and thereby differences in grazing pressure, which in turn resulted in differences in phytoplankton responses to Pyrene exposure between the enriched and non-enriched communities at the end of the experiment. The results show that interactions between trophic levels and their relation to abiotic factors have to be investigated in efforts to understand effects of contaminants.

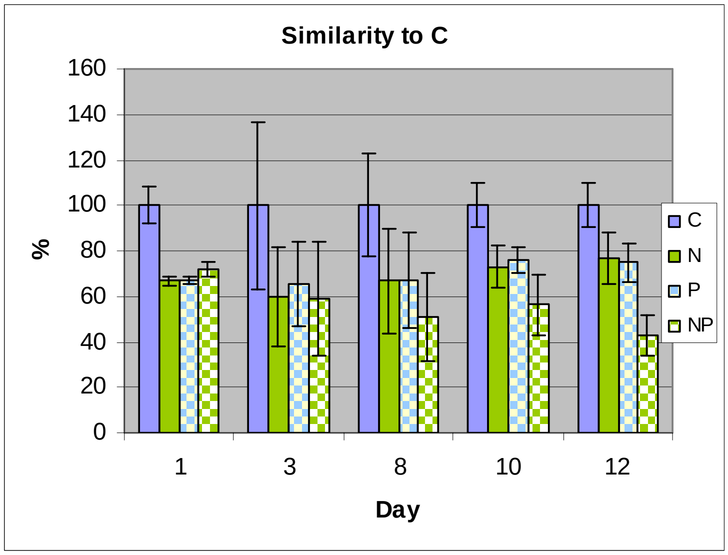


Figure 24. Development of adult zooplankton community in relation to the control community (C) without additions of nutrients (N) and Pyrene (P).

3.4 Thresholds and indicators

In the quest for biological indicators, taxonomical species abundance data are often used, but frequently with limited success in relation to the effort required to produce the data. THRESHOLDS has developed an analysis sequence for phytoplankton species, using a Baltic Sea data set. A combination of multivariate techniques allows an evaluation of the indicator value of individual species or aggregated taxa.

The first goal was to reduce ambiguity and redundancy by reducing the number of categories (taxa). Species were aggregated to genus level and rare genera into a single category (others). This reduced the original 462 counting categories to 129 taxonomic units, of which 118 were at genus level, while 10 represented higher taxonomic levels. The sample-by-taxon table was then merged with environmental variables. The resulting table contained 1304 samples from 60 stations over a period of 31 years (1970 to 2001). Data were averaged by station and year and station-years with 2 samples or less were excluded. This reduced the final data matrix to 147 station-years by 118 genera, and 6 environmental variables (salinity, Secchi depth, pH, phytoplankton biomass, total P, and total N – of which the latter three were log transformed).

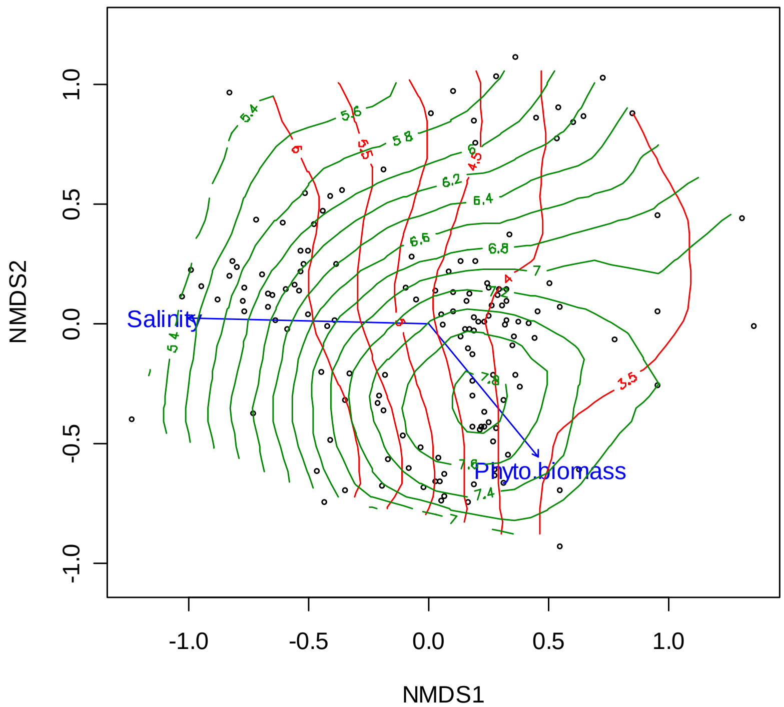


Figure 25. Ordinations and environmental gradients: the figure shows site scores as open circles, and isolines of fitted surfaces for salinity (red) and log(total phytoplankton biomass) (green)

The sequence of community analysis methods of the Vegan package for R was used. Bray-Curtis distances on square-root transformed biomasses were used after double Wisconsin standardization as dissimilarity measure, and non-metric multidimensional scaling (NMDS) with multiple starting points to define ordination axes based on taxon data alone. Environmental variables were then fitted to NMDS axes using a Monte Carlo permutation test for significance. (Figure 25). It is evident that the first NMDS axis is almost exclusively a salinity axis, with salinity decreasing right to left, while the second NMDS axis can be interpreted as an eutrophication axis.

Having identified salinity and total biomass as the most important environmental gradients in this data set, we can also look in more detail on the responses of individual genera along these gradients. For this we use generalized additive models (GAMs) with binomial response to predict presence/absence of a genus along an environmental gradient. GAMs are based on spline smoothers and can in principle have any shape constrained to the interval 0 to 1 (Figure 26). Notice that although the probability of nuisance taxa like *Aphanizomenon* or *Dinophysis* increases with eutrophication, there is still 25-50% probability of occurrence even at the least perturbed sites.

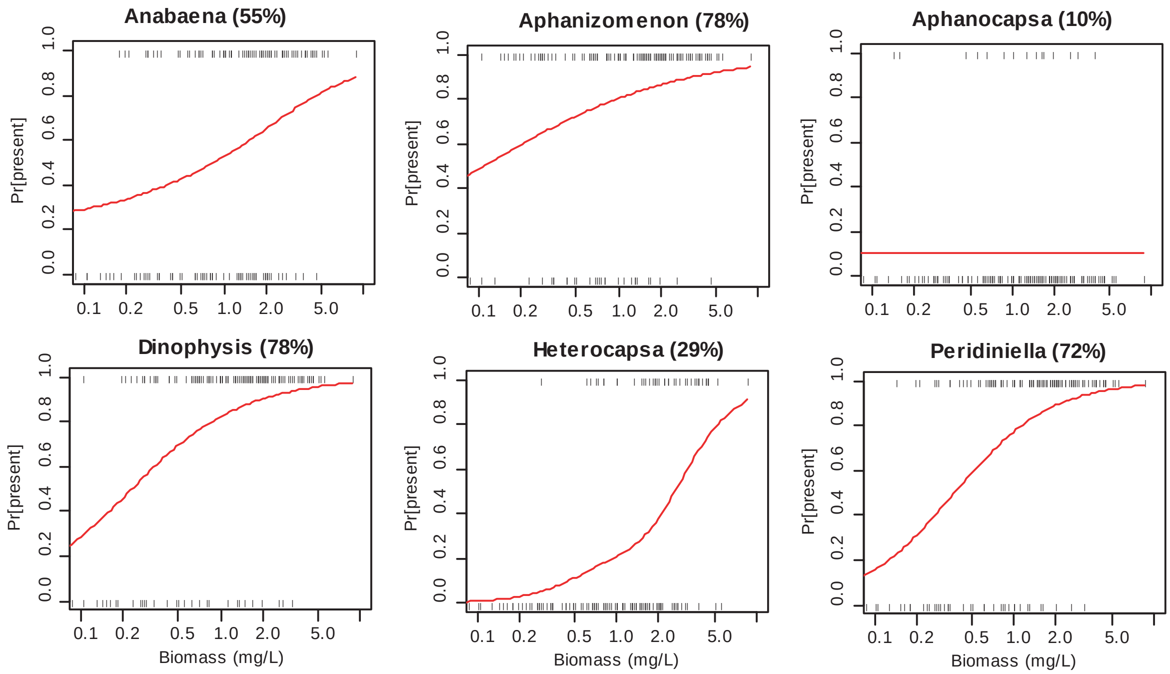


Figure 26. Examples of GAM response curves (probabilities for a genus being present under given environmental conditions) for total phytoplankton biomass from 6 different genera.

3.5 System-level and mechanism-level analyses

3.5.1 Analyses of mesocosms

The ecosystem mass flow networks were analysed for a planktonic mesocosm experiment conducted within an earlier EU project (COMWEB, FP4), where nutrient concentrations were manipulated over a 3-week period. The flow networks were analyzed both with holistic or system-level network analysis techniques, as well as with mechanism-level analysis of single processes and their relationships. The overall target is to develop robust indicators for changes in the basic structure or processes of the ecosystem.

Linear inverse modelling can be used to reconstruct mass flow networks based on non-destructive and indirect methods. Hence the method enables us to estimate process rates that are notoriously difficult to quantify by direct methods. Equally important, the method lets us explore the quantitative relationships among important ecosystem level processes, when they are at work simultaneously in whole and integral ecosystems. The method thus overcomes some of the conceptual problems related to inference from isolated process data, obtained under controlled conditions. It gives indications of how relevant processes work when they are embedded in a functional and complete network, where indirect effects play a role. A correct description of indirect effects is essential for mechanistic predictions beyond the immediate response of a perturbation.

During the 3-week experiment, most masses and flows increased substantially with the fertilization level (Figure 27). However, no large changes in the relationships among the mass flow networks took place, and the relative magnitude of the different flows did not change much. Hence, the system level indices showed only minor changes. However, the mechanism level indices were more sensitive to the nutrient manipulation.

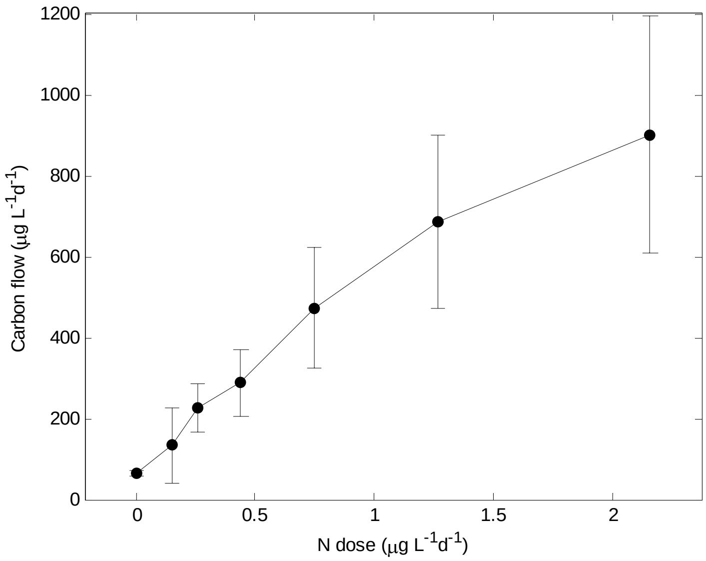


Figure 27. Sum of all internal system flows vs. nutrient loading rate.

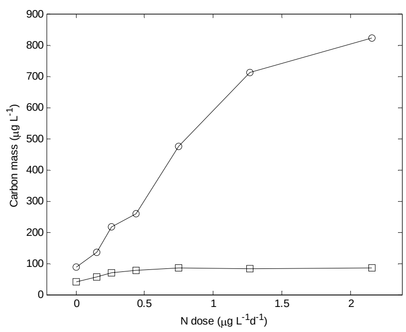
The analyses revealed several notable patterns:

1) The increased algal production induced a linear increase in the grazing flow, but a saturating increase in herbivore biomass (Figure 28).
2) A significant increase in the biomass specific loss to the detritus pool for all living components combined (16\% to 52\%, Figure 30). Hence, an increased portion of the algal production is lost as detritus via the zooplankton.
3) Bacterial production efficiency dropped. This could be ascribed to a lowered quality of the available DOM, or more possibly, a surplus resource usage, maximizing net production instead of production efficiency.
4) Zooplankton assimilation efficiency dropped precipitously as N-loads increased (Figure 31). The explanation could be luxury feeding, or a lowered digestability of the phytoplankton. However, this was not accompanied by any notable change in the total clearance rates of the zooplankton.

In addition, the experiment revealed several anticipated responses. The ratios of community respiration over gross primary production (CR:GPP, Figure 28) and the ratio of gross bacteria production over gross primary productin (GBP:GPP, not shown) both fell exponentially from more than 1.2 or the unfertilized system to below 0.6 for the most fertilized bag. This reflects a change from a net heterotrophic to a net autotrophic system. It also shows the quicker response of the autotrophs compared with the bacteria to N-addition. This is also reflected in a larger increase in grazer flows relative to bacterial production flows (not shown).

The mechanism level approach has sharper focus, as fewer process rates are used in the calculation. In addition these mechanism-based indices have the advantage of being easier to interpret due to the much richer and mature theories at the mechanism level, compared to those at the system level. In addition, the theories on system level properties are of dynamic nature, making predictions on system succession. Since our data and reconstructed mass flow networks covered the time-averaged response for a gradient of fertilization perturbations, they were not ideal to test dynamic system theories.

Although some of the qualitative responses found this way, for instance a lowered zooplankton assimilation efficiency with increased N-loading rate, can be given *ad hoc* explanations with hindsight, the flow network analysis approach allows us to quantify the changes, paving the way for hypothesis development and testing.



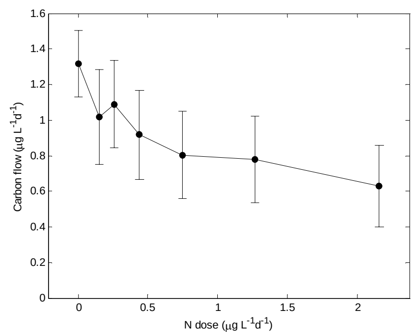
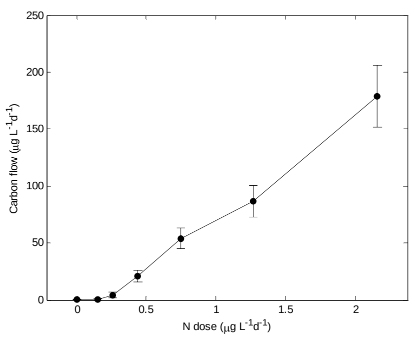


Figure 28 (left) Biomasses of total autotrophs (circles) and zooplankton (squares) vs. nutrient loading rate.
Figure 29 (right). The ratio of community respiration over gross primary production vs. nutrient loading rate.



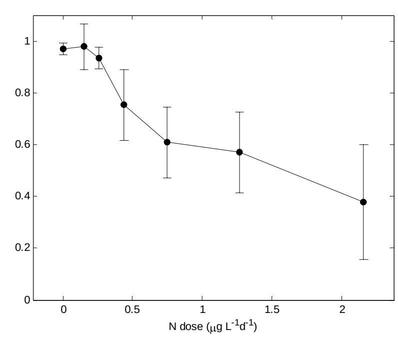


Figure 30 (left). The loss of material as detritus vs. nutrient loading rate.
Figure 31 (right) . Zooplankton assimilation efficiency vs. N-loading rate. The assimilation efficiency is calculated as the ratio of (ingested – egested) / ingested mass flow for all the zooplankton compartments combined.

3.5.2 Empirical relationships between water quality and the distribution and abundance of benthic vegetation

Nutrient load affects the benthic vegetation through a cascade of interactions leading to shading and also, for example, anoxic events. Even though these general eutrophication effects are well-known, quantitative relationships are limited and scattered in the literature. Such quantitative information is needed for management. In THRESHOLDS existing information on empirical relationships between water quality variables and distribution and abundance of seagrasses and macroalgae based on large-scale empirical data.

The compilation demonstrates that seagrasses and macroalgae reflect water quality in a quantitative manner by growing deeper, being more abundant and more widely distributed in clear waters with low nutrient concentration compared with more eutrophic systems. Vegetation in deeper waters shows the strongest response because it is most markedly affected by eutrophication-induced shading. Similar types of response are found in many different systems, indicating a wide robustness and generality of the findings. However, system-specific attributes such as water residence time or presence of filtrating organisms affect the level of response (see Figure 32) across systems.

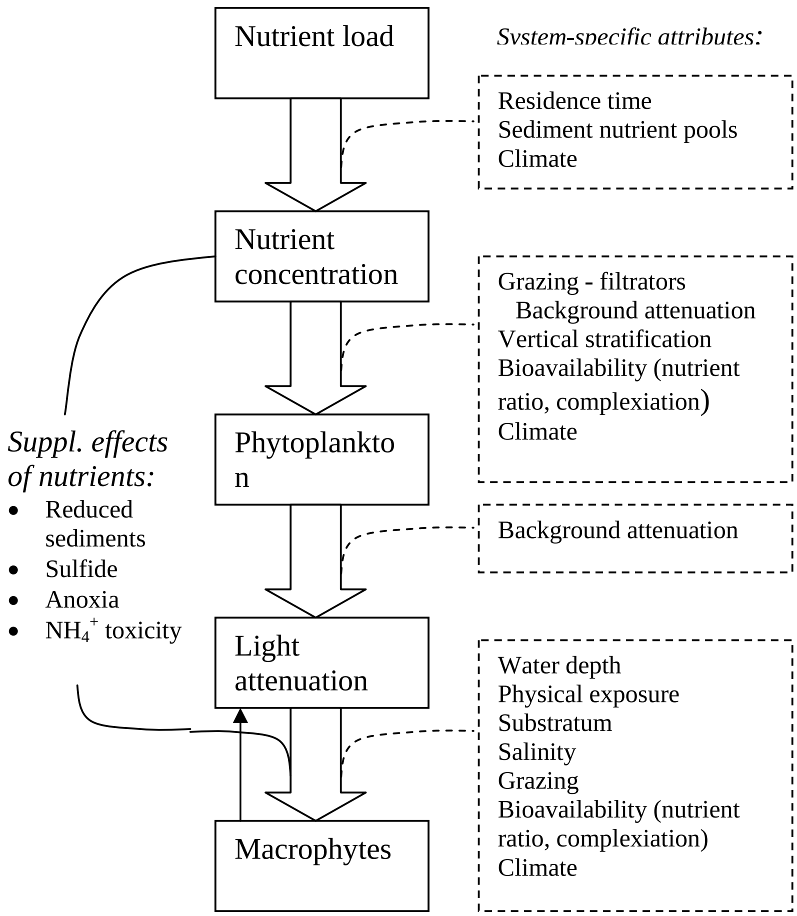


Figure 32. Conceptual model illustrating 1) eutrophication-caused shading operating through stepwise links from nutrient load over nutrient concentration, phytoplankton chlorophyll and light attenuation to vegetation response (large arrows), 2) other indirect and direct effects of eutrophication, which may modify the response of the vegetation (thin lines), and 3) system-specific attributes that may affect each of the stepwise links by constraining or amplifying the response and thereby obscuring the coupling to nutrient load (dotted lines)(Krause-Jensen et al 2007).

3.5.3 Light attenuation and seagrass colonization depth

The possibility to predict the threshold depth to support seagrass growth in coastal waters from light attenuation has been confirmed (Figure 33). However, the analysis conducted showed that the relationship between light attenuation and seagrass colonization depth differs between turbid and clear waters. This threshold can be used to set up water quality targets to meet required levels of ecosystem integrity, in terms of seagrass abundance and cover.

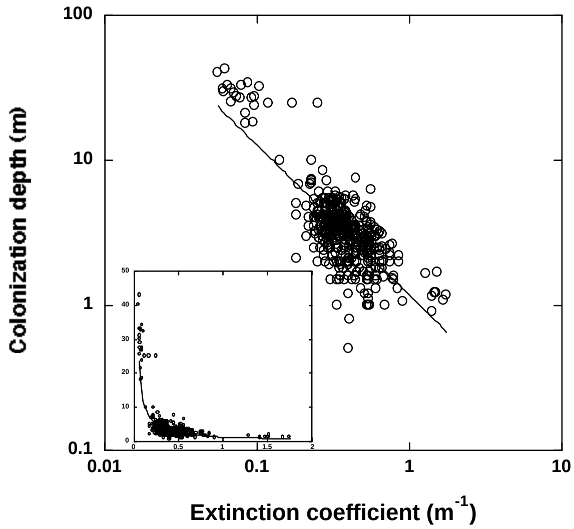


Figure 33. Relationship between attenuation and seagrass colonization depth (Duarte et al. 2006)

3.6 Determining economic consequences of thresholds

Thresholds can be related to discontinuities in a number of areas which affect the way in which changes in the welfare of society develop. Such “threshold effects” can exist in a number of forms:

i. In the pressure-ambient state relationship.
ii. In the activity-environmental pressure relationship
iii. In the ambient state-valuation relationship

This means that there can be multiple discontinuities, which affect welfare when activity levels change modifying pressures (Figure 34). The diagram highlights the need for care in assuming linear relationships. For the ambient state-valuation linkage, valuation “ladders” are frequently observed. When the ambient state changes, activities also change resulting in a jump in the level of damage to society.

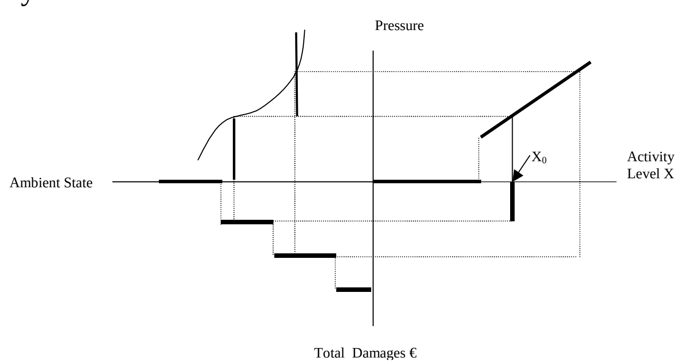


Figure 34. An activity-pressure-state-damage diagram showing how thresholds result in a step-wise increase of costs.

3.7 Costs of algal blooms and beach congestion

State-of-the-art choice experiment techniques can elicit the values placed on algal blooms. Focus groups can provide indicative values for issues such as visibility, the duration of algal bloom and congestion. Choice sets have been developed to explore the economic values that users attach to algal foam in the water and on the beach, the duration of the nuisance and the level of congestion on the beaches (Table 2). Photographs are used to help respondents understand the foam problem and the congestion issue (Figure 35, Figure 36).

Table 2. Choice Set Example. Different combinations of these kind were presented to the interviewees who expressed their willingness to pay a particular sum ("extra charge")

Caractéristique **Characteristics**	**Projet A** **Project A**	**Projet B** **Project B**	**Aucun projet** **No project**
Quantité de mousse Quantity of foam	En surface de l'eau seulement Only on surface of water	Grandes accumulations sur la plage Major accumulations on the beach	Grandes accumulations sur la plage Major accumulations on the beach
Durée de la mousse (entre avril et juin) Duration of algal foam conditions (between April and June)	8 semaines/weeks	4 semaines/weeks	8 semaines/weeks
Nombre de personnes à la plage Number of people on the plage	Elevé/High (moins de 3 m entre vous et la plus proche personne) (less than 3 m between you and the closest person)	Elevé/High (moins de 3 m entre vous et la plus proche personne) (less than 3 m between you and the closest person)	Moyen/Medium (3 à 20 m entre vous et la plus proche personne) (between 3 and 20 m between you and the closest person)
Charge additionnelle Extra charge 2007	9€	36€	0€

Figure 35. Medium level of algal foam in Belgium: some deposits on the beach

Figure 36. High level of congestion in Belgium: nearest person is less than 3 metres away

4 Managing thresholds

4.1 Management issues in coastal and marine waters

Single measures are generally insufficient to improve a system that cannot deliver ecosystem services due to a regime shift. Improvements of the state are likely to require integrated measures, taking into account that some measures may have to be used in concert to achieve the desired outcome. Others are simply additive and the choice can be made based on straightforward cost-effectiveness considerations. If positive synergies between measures can be achieved, the final effectiveness of a combination of measures may even be higher than the sum of the individual measures.

Strategies and action plans play a key role in the implementation of the water framework directive (WFD) and the future Marine Strategy Directive. These strategies and action plans have to address two fundamental questions, in which thresholds may play a key role. The first question is how a “good environmental status” for a coastal/marine area can be specified? Answers to this question has been sought in numerous EU and national research and development projects. The goal has been to specify the limits between classes of status for different types of water bodies throughout the EU. There is ongoing work on intercalibration with the objective "to harmonise the understanding of ‘good ecological status’ in all Member States, and to ensure that this common understanding is consistent with the definitions of the Directive." [2]

The methods developed within THRESHOLDS may help managers to identify how stable the "good ecological status" is and how great the risk is that the good status may be lost. The results compiled within THRESHOLDS have shown that the change may be rather sudden due to regime shifts. This leads to the second question: Which are the relevant mechanisms that determine the change between different classes of status. In particular: how can one avoid slipping from good to moderate status or return from moderate to good?

Answers to these questions support the identification of measures that must be part of strategies and action plans. The measures must be assessed in terms of their costs, feasibility and acceptability. Frameworks for the assessment and evaluation will be developed using the THRESHOLDS case studies and by taking into account specific results and tools:

- Methods for devising quality indicators and identification of thresholds and points of no return;
- The methodological approach to socio-economic assessment of externalities in the presence of thresholds effects;
- Examples of drivers (human activities) and sources of pressures (riverine responses to changes in drivers) that may be associated with threshold effects;
- Examples of avoidance and mitigation measures aimed at influencing drivers in order to reduce pressures on coastal zones to levels where the risk of exceeding thresholds is minimised;
- Examples of chemical monitoring programmes and the assessment of potential risk of chemicals on good ecological status. The results of this work are already helping the Common Implementation Strategy of the Water Framework Directive on the Priority Substances Working Group where the THRESHOLDS partners are actively involved in the Chemical Monitoring Activities.

4.2 Riverine nutrient responses to changes in emissions

From a management point of view it is essential to know how long it can take to detect the response of a river system to changes and implemented measures e.g., in agriculture. Such information is needed to establish realistic goals.

Specific cases studied in THRESHOLDS include the major changes in agricultural practices in Europe. In the late 1980s, the use of commercial fertilisers in most Eastern European countries began to decrease at an unprecedented rate. (Figure 37, left panel)

[2] (http://ec.europa.eu/environment/water/water-framework/objectives.html)[10.12.2006].

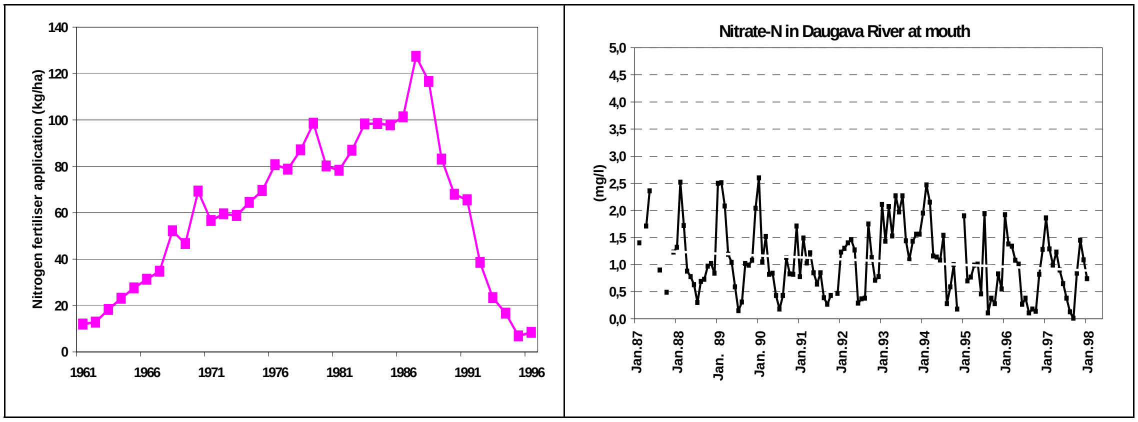


Figure 37.Figure illustrate the large drop in commercial N fertiliser use in Latvia (left panel) vs. the riverine response of nitrate-N at mouth of the Daugava River in Latvia (Stålnacke et al., in prep.

The number of livestock decreased almost by a factor 4 during the same period. Moreover, the import of feedstuff in most Eastern European countries dropped to almost nil, and extensive slaughtering of livestock reduced the amount of manure. Such abrupt and substantial changes in land use have rarely been recorded in the history of modern European agriculture, and they have created unique opportunities to study the impact of agricultural practices on water quality.

Although a number of researchers have considered this subject, scientists have not yet conducted a systematic analysis and comprehensive overview of the situation. In THRESHOLDS the goal was to summarise existing knowledge about how nutrient levels in rivers have responded to the large-scale changes in agriculture in Eastern Europe over the past decade. Published reports are inconclusive and comprise several examples of both decreasing and non-decreasing nutrient concentrations.

The findings indicate that large cuts in nutrient inputs do not necessarily induce an immediate response, particularly in medium-sized and large catchments (Figure 38, right). The reasons and causes for the differences in trends and especially the effects of hydrological processes (e.g., pathways) on nutrient losses are warrant further studies.

In THRESHOLDS various types of functional relationships between nutrient driver and pressure reported in the scientific literature and in reports have been compiled and examined. Special attention was paid to empirical relationships between nutrient loads, fluxes and concentrations vs. diffuse (i.e., non-point source) nutrient emissions/losses from agriculture and forest, at various spatial scales (plot, field, catchment and river basin).

On a global and continental (and to some extent also regional) scale, the level of riverine nutrient loads (pressures) can be derived from simple empirical relationships with driving forces like population, land use, fertiliser use and atmospheric deposition, but the causality in those relationships should be discussed. Further discussion is also warranted since the statistically-derived relationships seem not to hold or are much weaker at river basin and smaller scales. Especially problematic are relationships between nutrient drivers and pressure in agriculture and forest ecosystems. This is explained by the fact that nutrient losses show high short-term and almost unpredictable temporal variability in combination with a large spatial variability within and between catchments, with few strong correlations to the main driving force.

The results suggest that no simple and straightforward relationship between nutrient losses and traditional explanatory factors/environmental indicators can be easily obtained at ’smaller’ scales. This is due to the prevalence of several governing small-scale factors and especially the great retention potential in both agricultural and forested ecosystems that will be explored further in the project.

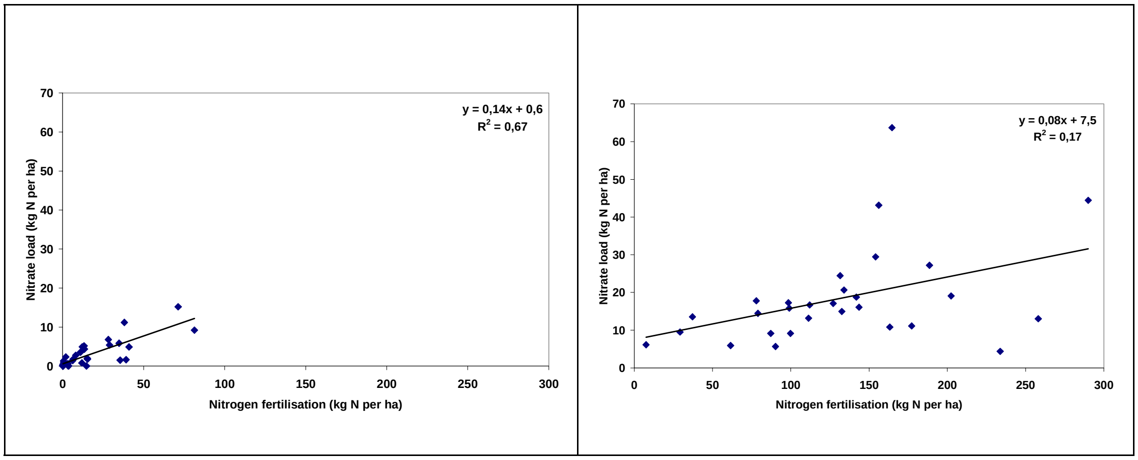


Figure 38. Figure illustrate the ‘spatial nutrient paradox’, i.e. that patterns and relationships at large scale (left panel, data from large rivers) are not necessarily present at smaller scales (right panel-data from small streams in agricultural catchments)

This ‘spatial nutrient paradox’ needs further studies. We also conclude that in a spatial context potential threshold values, the empirical relationships and especially the underlying mechanisms and causal relationships are poorly known and warrant further studies.

4.3 Modelling

An important aspect in understanding sudden changes, thresholds and regime shifts in coastal areas is to identify the transport processes that cause such changes (Figure 39). In threshold-science, it is essential to understand changes in the structure and function of coastal ecosystems and to relate such structural changes to variations in abiotic variables, such as changes in salinity and temperature, but also in chemical variables, such as nutrient concentrations, and in toxic substances.

One basic idea of the models developed and tested in THRESHOLDS is to analyse important factors regulating changes in concentrations of substances. This is a key issue in coastal management, especially in connection with thresholds and remedial actions that aim at avoiding critical thresholds. In the following examples of the modeling work in THRESHOLDS are provided (see also section 3.2).

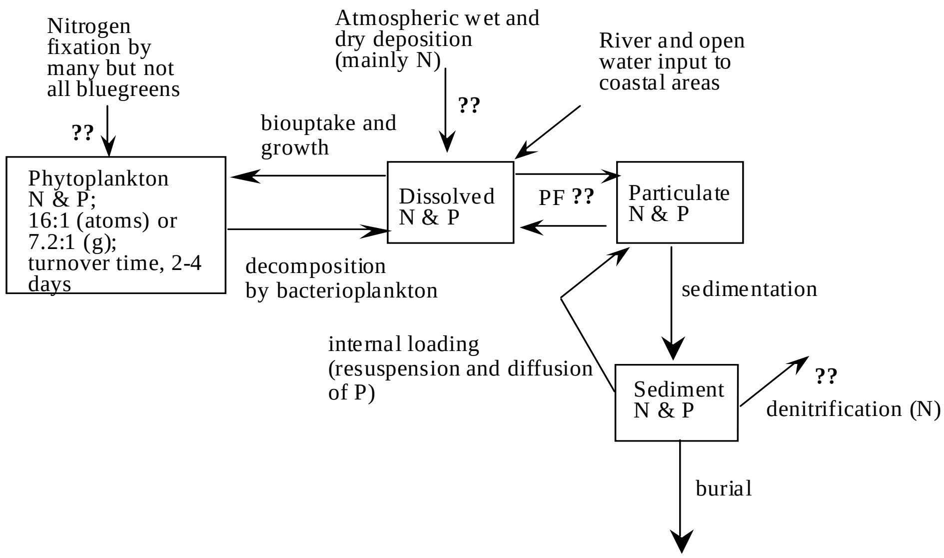


Figure 39. Example of an overview of key transport processes and mechanisms related to the concept of "limiting" nutrient, a key issue in coastal management and science, especially related to thresholds and remedial actions to avoid critical thresholds.

4.3.1 Empirical base

Empirical data on chlorophyll-*a* concentrations, concentrations of bluegreen algae (cyanobacteria), concentrations of nutrients and different forms of nutrients, water temperature and salinity have been analysed. The data concerns more than 500 aquatic systems covering a very wide domain in terms of nutrient concentrations and salinity. The focus has been on identifying the factors regulating and limiting the predictive power of models for variations among systems in median summer values of chlorophyll-*a* concentrations and concentrations of bluegreen algae.

The analysis has shown that there is no simple relationship between the TN/TP-ratio and empirical chlorophyll concentrations or concentrations of bluegreens. If the classical Redfield ratio would be the criteria for nutrient limitation, then practically all systems have TN/TP-ratios higher than 7.2 and there would be very few nitrogen limited systems. A model to predict concentrations of bluegreens from data on TP, the TN/TP-ratio, salinity and temperature has been developed. Using this model and models for chlorophyll and Secchi depth based on similar extensive data sets, we have introduced a general trophic level classification for aquatic systems based on chlorophyll classes (for oligo-, meso-, eu- and hypertrophic systems).

Classification systems like the one above are meant to provide a scientific framework for water management and comparative studies based on empirical data are meant to reveal typical and general patterns in aquatic ecosystem structures and functions so that specific features, thresholds and points of no return may be better understood and managed. Variations in TP rather than TN generally seem to be more important to predict variations among systems in chlorophyll-*a* and bluegreens.

Different "bioavailable" forms of the nutrients (DIN, DIP, phosphate, nitrate, etc.) have been shown to have very high coefficients of variation (CV), which means that many samples are needed to obtain reliable empirical data which are necessary in models aiming for high predictive power and practical usefulness. If too few samples are taken and if the CV-values are high (which is the case for many of the studied water variables), the values requested to detect critical changes and thresholds in the system will be uncertain. This may imply that the objective of the monitoring cannot be achieved.

4.3.2 Eutrophication and contaminant models

Specific models related to phosphorous driven coastal eutrophication and contamination of organic pollutants have been developed within THRESHOLDS. The models are process-based mechanistic models which aim to increase our understanding of how coastal areas respond to the loading of nutrients and toxins, and hence also to changes in pollutant loading. The basic aim is to be able to predict how a given coastal system would respond to often costly remedial measures.

These process-based models have been tested against empirical data and, e.g., the model for coastal eutrophication has been validated for 23 coastal areas and shown to predict phosphorus concentrations, Secchi depth (a measure of water clarity), chlorophyll-*a* concentrations and sedimentation within the error bars of the empirical data for monthly mean values.

Key questions that can be answered using the phosphorous model are: 1) How would a coastal area respond to changing phosphorus load, e.g. 30 % reduction in tributary inflow? 2) How would this reduction influence phosphorus concentrations and important bioindicators or ecosystem effect variables included in the model, such as algae biomass, water clarity or oxygen concentration in the deep-water zone regulating the survival of zoobenthos?

It is important to gain understanding why thresholds or regime shifts appear, and why there may be compensatory effects preventing them from appearing. One important way to explain, understand and predict mechanisms behind desired - or undesired - changes in concentrations of nutrients and toxins in aquatic systems, and drastic changes in key biotic variables (functional species or groups) related to such changes in chemical concentrations (drivers or abiotic variables), is to calculate proper mass-balances for the chemical variables. The model provides such a tool for coastal ecosystems that are controlled by phosphorus loading.

A coupled hydrodynamic and fate model has been developed to simulate the spatio-temporal variability of contaminants in coastal ecosystems. The organic families selected as representative includes: polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers PBDEs). Selected member of these families have been introduced in the 1D version. The model will be validated with experimental data, extended to 3D for specific cases and coupled with the biology to take into account toxic and/or bioaccumulation effects. The model takes into account the role of atmospheric processes as well as sediments (Figure 40).

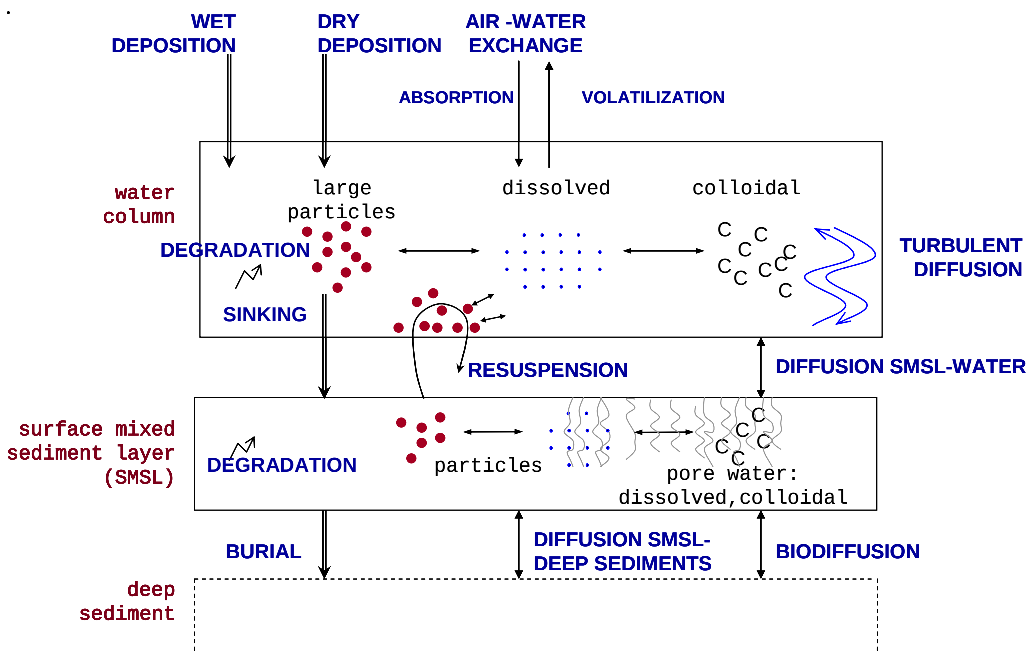


Figure 40. ***Fluxes of POPs accounted in the 1D coupled hydrodynamic-contaminant model.***

4.4 The impact pathway approach

A synthesis of the work in THRESHOLDS will help to develop a methodology that includes comprehensive scenario analysis with impact assessments for coastal zones. The methodology can be characterised as an impact pathway approach that has close links to the driver-pressure-state-impact-response (DPSIR) framework. Specific tools to perform scenario analyses will be produced.

Six case studies have been analysed with regard to content and scope and to what extent they include or should strive for a full impact pathway approach. Additionally a seventh, the ‘generalization case study’ has been defined that should include generic features of the coasts of Europe. Each of the case studies provides different sets of drivers, pressures, state changes and impacts showing how the analysis is to be adapted to the specific contents of the cases. In addition to illustrating the methodology, the application of the approach can guide further data gathering and the compilation of information on possible mitigation measures.

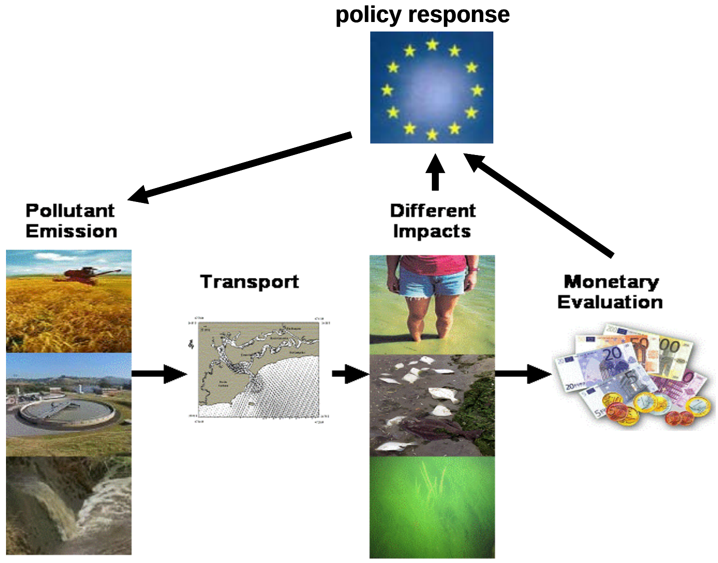


Figure 41.The Impact Pathway Approach (IPA) as exemplified for coastal zone impacts.

The ***North Sea*** case study is the most elaborated case study, with almost all models and data already available. Depending on the effects that can be valued in monetary terms, various policy trends or aspects may be analyzed. Given the already adopted European Water Framework Directive (WFD) with a compulsory tertiary waste water treatment for agglomerations larger than 10.000 inhabitants by 2015, it is planned to perform scenario analysis beyond this year. This implies that the respective models need to be adapted to be able to run until for instance 2020 if the implemented measures in the near future are not sufficient enough to reduce algal bloom occurrences to an acceptable level.

The ***Varna Bay*** case study is similar to the North Sea case in terms of investigating riverine loads to coastal ecosystems in which algal blooms and other species abundance shifts occur. At both sites, contingent valuation surveys are already or will be carried out within other THRESHOLDS. The difference is that no model is already available for Varna Bay, and therefore possibilities to adapt the North Sea model will be explored. A policy question to be analysed is to anticipate changes in water protection policies after Bulgaria joins the EU.

The impact pathway identified to be investigated for the ***Marine fish farms*** (i.e., the effect of deposition of feed and excrements to benthos on sea grass decline) is rather straightforward. The link between sea grass decline and identified and quantifiable impaired services is already available for CO_2 sequestration in the Mediterranean Sea. But efforts will be undertaken within THRESHOLDS to provide relationships for further services. In addition to that, further threshold models that would include benthic fauna decline in the North Sea will be examined. Scenarios will be used to analyse increasing demand for farmed fish and ways of managing expanding fish farming in an ecologically sound manner.

The most obvious and from a scientific point of view ‘beautiful’ threshold effect in the ***Ringkoebing Fjord lagoon*** is related to the salinity which is driven by sluice management. This is a special and unique case, whose results cannot be generalised for coastal areas throughout Europe. There are many data and even two models in place for Ringkoebing Fjord. For this case study, there is an ecological lagoon model developed within THRESHOLDS by University of Uppsala. However, for linking drivers (e.g., agriculture and residential or industrial activities) to pressures in the lagoon, a catchment model is needed. Thus it was decided not to carry out a full impact pathway analysis in this case. Instead it supports the methodological development in the project.

The impact pathways identified to be investigated for the ***Mallorca*** case study are rather straightforward. The missing links are clearly identified and efforts undertaken within THRESHOLDS will provide data. The scenarios will focus on the impact of tourism and its regulation.

The analysis has show that it is not meaningful to have a contaminant scenario study on ***Thau Lagoon*** unless one looks at pesticides. This is due to the emission data availability and the need for a rather large scale model or speciation model if POPs or heavy metals will be included, respectively. It was found that at present concentrations of these pollutants are too low to give relevant threshold levels observable at the ecosystem scale. Also, political thresholds or thresholds in the valuation function are estimated not to exist. The Thau Lagoon will therefore not be included in the scenario analysis.

Based on these considerations a ***Generalization case study has been initiated.*** The aim of this case study is to analyse threshold effects using the impact pathway approach for larger areas or even the whole of Europe based on the findings of the case studies for individual sites. The most critical part is the development of indicators for thresholds effects applicable at large temporal and spatial scales. The concept and the scope of the generalisation have to be developed taking into account the availability of information. This is particularly so as detailed and standardised discharge etc. data are not expected to be (readily) available throughout Europe.

Thus, the case studies identified for scenario analysis are the North Sea case, Varna Bay case, marine fish farms, Mallorca and the Generalization case. The existing models and data show rather different degrees of availability. Particularly, the North Sea case is almost ready for the analysis. For the cases 'Varna Bay' and 'Generalization', there are substantial model and/or data gaps identified. Efforts to fill these gaps are under way.

5 Policy implications of thresholds

5.1 The science-policy dialogue

The 21st century has produced a rich set of holistic review papers focusing on indicator development and selection. Several attempts have been made to develop a common language between natural and social sciences in order to incorporate ecological indices into policy processes and adding humans into the 'equation'. The translation of research into information that can be used in the policy domain and the translation of policy questions into specific research questions remain (Figure 42), however, challenging tasks.

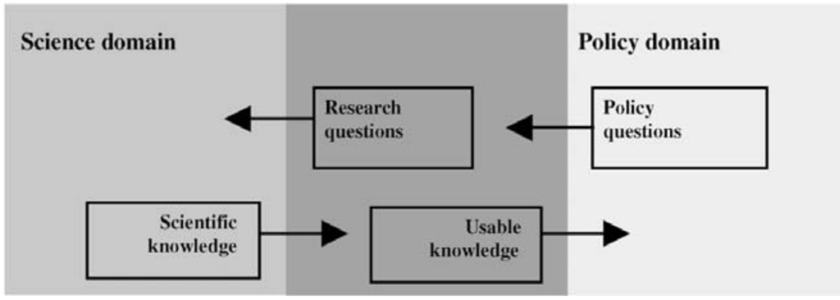


Figure 42. Science-policy interface as an overlapping fuzzy boundary area. Scientific knowledge is translated into usable knowledge and policy questions are translated into research questions (after Turnhout et al 2006).

One of the objectives of THRESHOLDS is to provide an input into ongoing policy dialogues in the EU. It can do so by

1) Identifying new issues and question for the policy agenda, for example by highlighting the EU-level significance of specific aspects of contaminants or eutrophication and their consequences for ecosystems or human wellbeing.
2) Providing advice on how the existence of ecological thresholds should be taken into account so that the legal texts and the measures they imply are consistent with best available knowledge of the nature of thresholds. For example the Marine Strategy Directive should reflect the possibility that systems can change through thresholds. Also documents guiding the implementation of the Water Framework Directive (WFD) may be adjusted to take into account information on thresholds.
3) Providing methods for the implementation of the agreed policy, in particular the implementation of the WFD may benefit from novel methods that can be used by national and local authorities and other actors.
4) Evaluating the consequences of already implemented policies by exploring the intended and unintended effects of these policies from a thresholds perspective. For example, is the reduction of nutrient discharges resulting in expected changes or not in the status of the water bodies and should the focus be changed? What are the costs relative to the effects or benefits of the policies? Can more cost effective solutions be found?
5) Providing novel views of the functioning of coastal and marine ecosystems. Can, for example, the present implicit and explicit assumptions underlying water policies and the corresponding legal documents be questioned and is there a need to base future policies on a different set of assumptions.

5.2 Thresholds and policy learning

European policies that aim at safeguarding coastal and marine waters from degradation use concepts linked to ecological thresholds. Although ecological thresholds have not been used explicitly as a legal concept many policy instruments and interventions are implicitly based on assumed but unknown thresholds.

A reliable estimation of thresholds may help in justifying standards and limit values in policies. If, however, research shows that the exact level of a threshold is subject to considerable uncertainty and context dependent variation, fixing limit values by giving them normative legal status may cause significant unintended economic and social consequences. In this case the policy should be developed to include adaptive elements, that encourage learning processes. These processes should involve a broad range stakeholders, who may gradually learn how to develop sustainable ways of dealing with systems that exhibit threshold values.

For example, even though there has been a considerable research effort in the understanding of the effects of contaminants in aquatic ecosystems there are still an important number of gaps in our knowledge. These gaps are related to the non-linear effects, the existence of combined thresholds and the overall estimation of effects at ecosystem level, meaning that limit values are in many cases rather arbitrary, and should be given normative status only after very serious consideration.

Ecological thresholds can in theses learning processes usefully be used as references and foci for the attention. They should thus always be coupled with processes of appropriate monitoring, ex post evaluation and revision mechanisms. These can in the long run identify and deepen the understanding of relevant non-linear processes, and eventually also lead to better and more reliably specified limit values at appropriate spatial and temporal scales.

6 Dissemination

THRESHOLDS is actively **disseminati**ng results of the research activity. The purpose of the dissemination has until now been to raise awareness of the concept of thresholds among stakeholders and an interested public. By producing basic material such as a web site and a brochure and a policy brief, and by creating an accumulating information package of popular articles on thresholds, a basis has been created for further dissemination.

The objective is to produce and provide material that can be adapted to different settings and contexts for dissemination to stakeholders and the public. It is based on a recognition of the diversity of thresholds in the coastal ecosystems throughout Europe.

Pilot work that will "translate" and distribute results of the scientific studies in thresholds to a wider audience of professionals and interested members of the public have been carried out and will be geared up once scientific articles have been published.

The dissemination work has put thresholds on the agenda in policy level discussions. An example includes a presentation and discussion at a meeting of the Directors of European Environment Agencies in April 2006 in Vienna, and presentations at the Baltic Sea Conference in Helsinki in November 2006.

THRESHOLDS has a web space where all the news concerning the state of advancement of research are available (http://www.thresholds-eu.org). The deliverables produced so far can be downloaded from the web site. A newsletter is regularly issued to increase awareness of the state of advancement of research among partners.

THRESHOLDS partners have been particularly active in producing scientific papers for publications. Over 50 papers have been submitted to the major scientific journals for publications, and some of them have been accepted and already published. THRESHOLDS also intends to promote the diffusion of the results among the wider public through publications in popular scientific journals.

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