Odense Pilot River Basin: implementation of the EU Water Framework Directive in a shallow eutrophic estuary (Odense Fjord, Denmark) and its upstream catchment

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Abstract Implementation of the EU Water Framework Directive (WFD) is a huge environmental management challenge for Europe, demanding an integrated sustainable approach to water management and a common objective of obtaining 'good status' for all water bodies before 2015. The main task is the preparation of a river basin management plan for each of the 96 European river basin districts before the end of 2009. In Odense River Basin (island of Fyn, Denmark), one of 14 appointed European Pilot River Basins, the implementation of the WFD has been developed and tested in practice. Reference conditions and ecological status classification for Odense Fjord, based on eelgrass (*Zostera marina*) depth limit and nutrient concentrations, have been drawn up through a combination of historical data and modelling tools. A subsequent quantitative linking of pressures and impact, *in casu* between land-based nitrogen (N) loading of the fjord and resulting nutrient concentrations and eelgrass appearance, provided an estimate of the needed nitrogen load reduction of the fjord. This amounted to approx. 1,200 tonnes N per year (an annual load reduction of ca. 11 kg N ha^{-1} of catchment area or ca. 19.5 g N m^{-2} of fjord surface)—a load reduction of ca. 60% from the present level—to obtain at least 'good' ecological status sensu WFD. It is presently not possible to quantify a target load for phosphorus (P) in relation to marine environmental objectives. An economically feasible programme of measures to obtain 'good' status in all surface water and groundwater bodies in Odense River Basin, using an integrated cost-effectiveness analysis, showed that re-establishment of wetlands, catchcrops, and reduced fertilisation norms are the most effective measures if large reductions in N loads to the aquatic environment are to be achieved. The total socio-economic cost of implementing the WFD in the river basin amounts to about 13 million €/year, which will

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increase the expense for water services by only 0.5–0.6% of the total income and production value in the basin (15,650 million €/year). Investments to obtain the needed nitrogen load reductions from agriculture are thus economically feasible. Further, it is not an impossible task, either economically or technically, to reach the objectives of the WFD while still retaining the possibility of keeping a high agricultural production in the catchment (maintaining livestock production but decreasing crop production in the case of Odense River Basin). The future conditions in Odense Fjord will not only depend on the success in reducing the load from the river basin area, but will also be affected by the trend in the nutrient loss from the whole Baltic catchment area. The high growth rates in the new EU Member States thus pose an important challenge to water managers, and decoupling of economic growth from pressure on water bodies will be necessary. Finally, a number of challenges facing water managers around the Baltic and within the EU, namely preconditions required to successfully implement the WFD, are presented.

Keywords Coastal eutrophication · River basin district · Ecological classification · Programme of measures · Environmental economics

Introduction

Implementation of the Water Framework Directive (WFD; Anon., 2000) is one of the greatest environmental management challenges facing Europe in recent times. The WFD introduces a holistic and fully integrated sustainable approach to water management by considering groundwater, surface waters and wetlands together and by introducing the overall long-term objective of 'good status' for all water bodies before 2015, unless special circumstances are documented. Implementation of the WFD is closely coupled to implementation of related environmental directives such as the Habitats Directive, the Nitrates Directive, the EC Urban Waste Water Treatment Directive and the recent Marine Strategy Framework Directive to save Europe's seas and oceans (Anon., 1991a, b, 2008). The programmes of measures needed to fulfil the objectives of these directives will often be similar and/or related to the same pressures or sectors. An integrated management approach is therefore a necessary precondition for ensuring 'value for money'.

The hydrological cycle links water bodies independently of national and regional borders. Sustainable water management is almost always a matter of international water management, involving authorities nationally as well as regionally. Out of the 96 nominated river basin districts in Europe, 27 are transboundary districts and cover 65% of the total area. Thus, the need to reduce waterborne and airborne pollution necessitates international cooperation, and the measures to attain the reductions should take into account fulfilment of the environmental objectives for water bodies in neighbouring countries. However, once Denmark, like other EU Member States, has solved its environmental problems in relation to its own natural habitats, watercourses, lakes and coastal waters, i.e. has ensured favourable conservation status and good surface water status, it will largely have made its contribution towards solving the environmental problems in the international water bodies such as the Baltic Sea and Kattegat.

For each river basin district, a management plan must be drawn up before the end of 2009 according to the time schedule for implementation of the directive. Preparation of the 96 such plans thus poses an enormous task for the European water managers. In order to facilitate the implementation of the WFD, a Common Implementation Strategy (CIS) was launched by the EU for the period 2002–2006. Odense River Basin was appointed as one of the 14 European Pilot River Basins, where the implementation of the WFD was to be developed and tested in practice ahead of the WFD schedule (Fyn County, 2003; Environment Centre Odense, 2007).

In this Opinion Paper, we discuss our experience in developing selected, crucial tools and process steps for implementation of the Water Framework Directive, such as characterisation of water quality, the quantitative linkage between pressures and impacts, and assessment and risk analysis in the small Danish estuary, Odense Fjord and its catchment. Based on this analysis, an economically feasible programme of measures is suggested that takes into account not only fulfilment of WFD objectives for Odense Fjord, but also for lakes, watercourses and groundwater in the Odense River Basin area; this is done through an integrated cost-effectiveness analysis.

Odense River Basin

The catchment to Odense River Basin encompasses ca. 1,095 km^2 (Riisgaard et al., 2008)-almost a third of the area of Fyn, an island in the Belt Sea in the heart of Denmark (Fig. 1). The population size in the catchment is ca. 250,000 (of which ca. 180,000 live in city of Odense). The most common landscape feature is moraine plains covered by moraine clay in this lowland catchment, with almost no points above 100 m; clayey soils dominate slightly (51%) over sandy soil types. Annual mean precipitation for the catchment amounts to 768 mm (1961–1990) (Riisgaard et al., 2008).

Characteristics and environmental status of Odense Fjord

Odense Fjord is a eutrophied estuary located at the northern coastline of Fyn (Fig. 1). Odense Fjord belongs to ecoregion 4, the North Sea, according to the Water Framework Directive. Odense Fjord is shallow with a mean depth of 2.25 m. The water-covered area is ca. 62 km^2. The 16-km^2 inner, mesohaline part of the estuary (Seden Strand) has a mean depth of 0.8 m, while the 46-km^2 outer, polyhaline part has a mean depth of 2.7 m. The catchment-area-to-estuary-volume ratio for Odense Fjord is relatively high, which implies a substantial

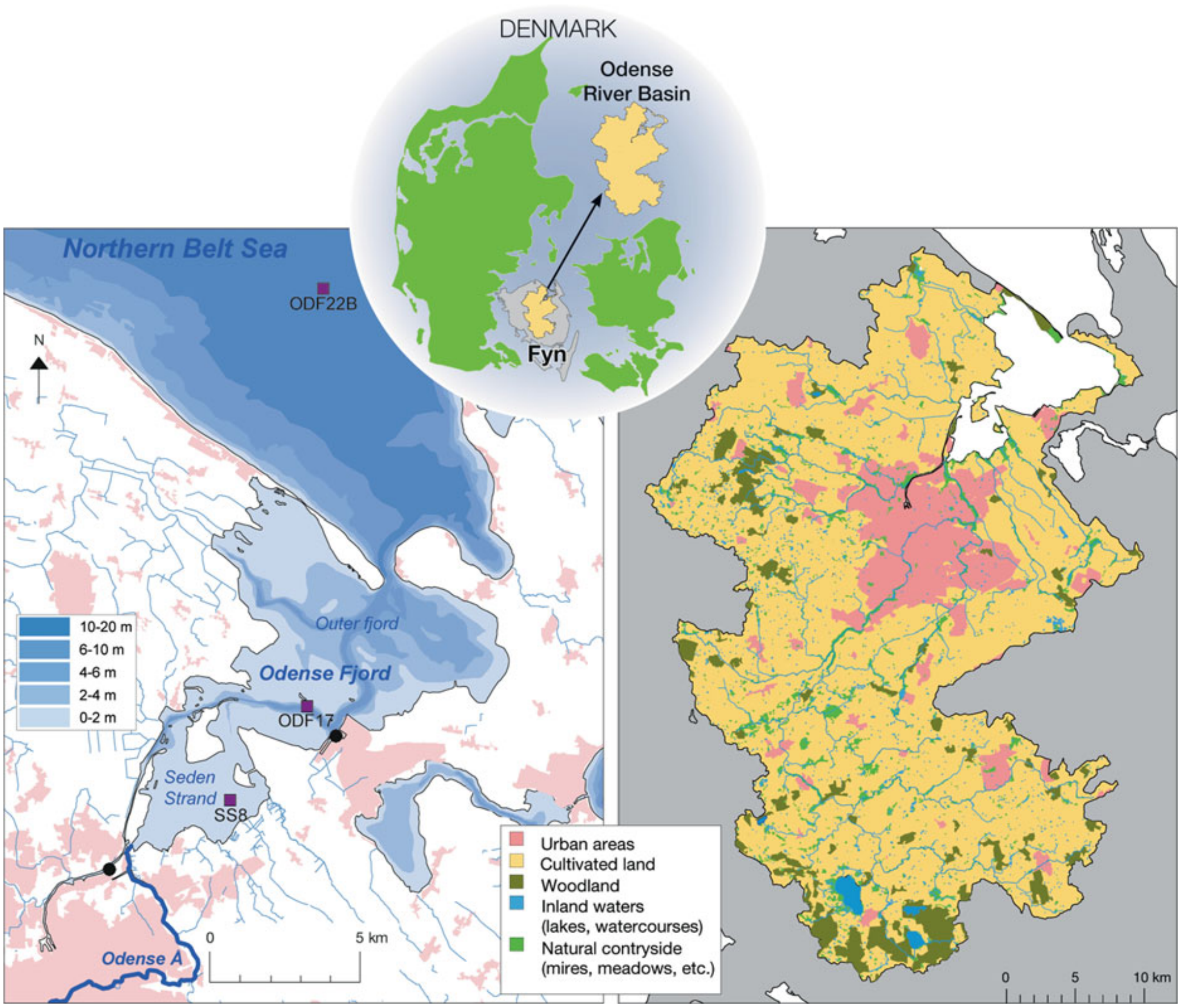

Fig. 1 Odense Fjord indicating the boundaries of the inner fjord (Seden Strand) and outer fjord (*left*). The monitoring stations SS8 in Seden Strand, ODF17 in the outer part of the estuary and ODF22B in the border area outside the fjord are indicated. In the map on the right, the catchment area is shown with associated land use

impact of the catchment on the estuary (Conley et al., 2000).

The largest source of freshwater input to the estuary is from the river Odense Å. Water exchange between the estuary and the adjacent open coastal area (northern Belt Sea) takes place via the narrow mouth, Gabet. Odense Fjord is microtidal with an amplitude of about 25 cm under normal circumstances, and taking the prevailing density-driven currents also into consideration, Odense Fjord is dynamic in terms of water exchange. Estimates of the residence time are 17 days (for river Odense Å water; Riisgaard et al., 2008) and about 1 month (for the whole volume of Odense Fjord water; Rasmussen & Josefson, 2002).

The large freshwater input, with an annual range of ca. 100–500 million m^3 (1981–2005), and nutrient loads to the inner part of the estuary create significant estuarine salinity and nutrient gradients (Fig. 2). The nutrient levels in the estuary are thus high (especially in the inner part receiving most of the load). Due to the strong gradients, the salinity–nutrient relation through the estuary is fairly linear on an annual scale (data not shown). The N concentration is at a seasonal maximum during winter when freshwater discharge is high and consumption by primary producers is low; the seasonal maximum for P is found during summer and is primarily due to a high phosphorus release from the sediment (Riisgaard et al., 2008).

Nutrient mass balance calculations using a hydrodynamic/ecological dynamic 3D MIKE model (Edelvang et al., 2005; Vanderborght et al., 2007) show that 26% of the N load on average is retained in the estuary by denitrification and burial in the sediment (Table 1). For phosphorus, however, the export exceeds the input by 35% on average (but ranging from a small retention to a very high net export). An explanation for this is that the estuary is still in disequilibrium with respect to phosphorus, because the sediment pools that accumulated during the very high P loads up until the late 1980s have not yet been 'washed' out (hence the high sediment–water P-effluxes during summer).

The changing nutrient regime through the estuary is also reflected in the phytoplankton biomass, decreasing towards the sea (indicated by chlorophyll *a* concentrations; Fig. 2). Phytoplankton growth is potentially limited by nitrogen during summer, but by phosphorus during spring (perhaps with occasional co-limitation by silicate) (Riisgaard et al., 2008). In addition to the land-based nutrient load, internal loads, i.e. sediment–water fluxes of N and P, are of importance for the phytoplankton production. In the shallow Odense Fjord, microphytobenthic algae are also important primary producers accounting for about one-third of the annual phytoplankton production.

The chlorophyll *a* levels in Odense Fjord are relatively low given the generally high nutrient loads

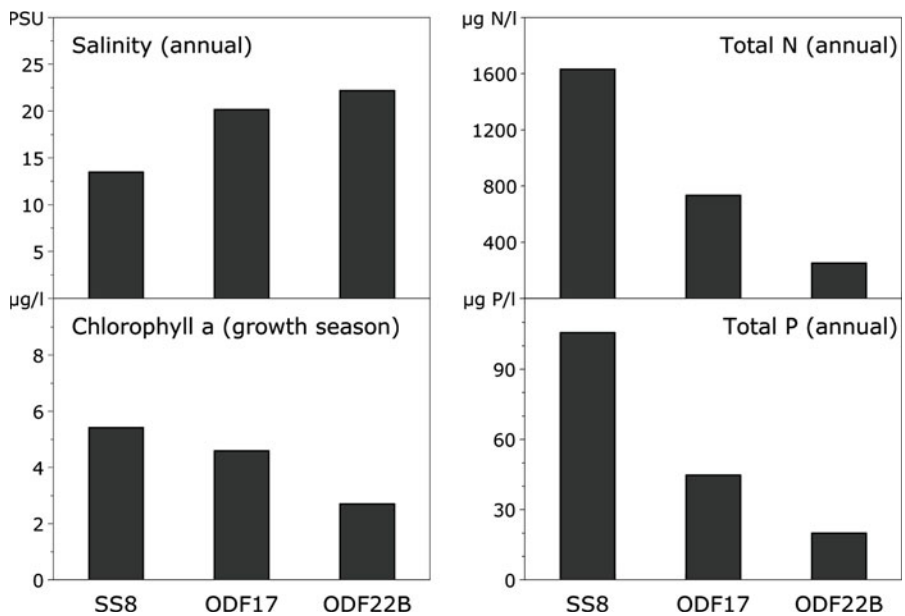

Fig. 2 Estuarine gradients of salinity, total nitrogen (TN), total phosphorus (TP), all annual means and chlorophyll *a* (mean March–October) for 2002–2004 through and outside Odense Fjord (see Fig. 1 for station locations). Data are surface water means except for salinity, which is the mean for the entire water column

Table 1 Annual mass balances of N and P (tonnes) for Odense Fjord, 1997–2004 (minimum, maximum, and average)

	Min-max (1997–2004)		Average (1997–2004)	
	N	P	N	P
Runoff	1,009–3,408	25–82	2,233	55
Export	493–2,906	45–109	1,735	69
Retention	373–620	−37 to 2	495	−14
Retention (%)	15–51	−123 to 4	26	−35

Retention is also shown in % of runoff

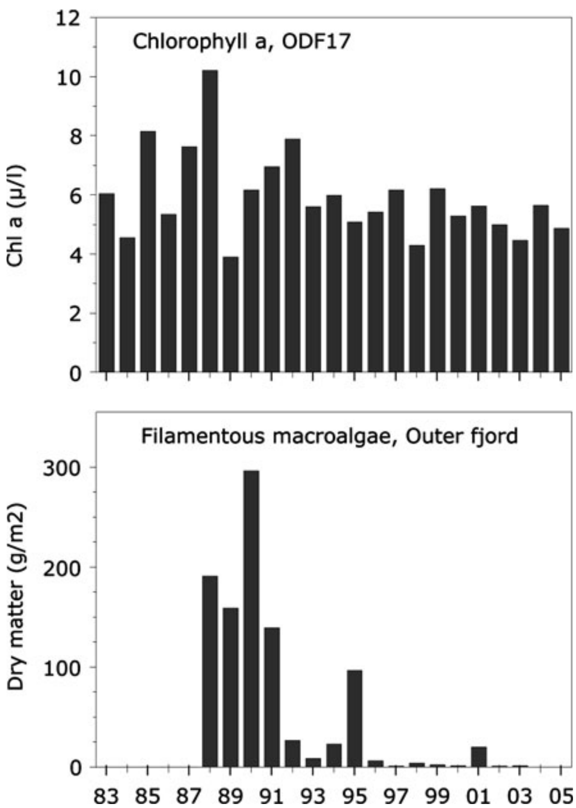

Fig. 3 Concentrations of chlorophyll *a* (May–October mean; stn. ODF17) (*upper*) and biomass of filamentous macroalgae during summer in the outer part of Odense Fjord (*lower*), 1983–2005

(Conley et al., 2000), and chlorophyll *a* has shown only minor changes despite reductions in N and P loads during the last 15–20 years (Fig. 3, upper). A high biomass of filtrating fauna, which are potentially able to filter the entire water volume of the estuary more than once per day, is a major factor responsible for this (Riisgaard et al., 2004, 2007).

The high nutrient input to the estuary also favours the growth of rapidly growing, ephemeral macroalgae, for example sea lettuce (*Ulva lactuca*). During the period of very high nutrient loading in the 1980s, the biomass of sea lettuce was exceedingly large in Seden Strand (Fig. 4, upper left)—in extreme cases more than 1 kg dry matter m^{-2} (cf. Riisgaard et al., 2008)—while ephemeral filamentous macroalgae were prominent in the outer estuary. The biomass of these macroalgae has subsequently decreased (Fig. 3, lower and Fig. 4, lower left) along with a nutrient load reduction that began around 1990. Simultaneously, rooted macrophytes such as widgeon grass (*Ruppia maritima*) in Seden Strand have again increased in abundance (Fig. 4, right), whereas the improvements apparently have favoured an increase in slower-growing perennial macroalgae, primarily *Fucus vesiculosus*, rather than eelgrass (*Zostera marina*) in the outer estuary. Despite the overall improvements, the biomass of sea lettuce in the inner estuary is still high and the coverage and depth distribution of eelgrass in the outer estuary are low. Considering these temporal variations of macro- and microalgae, Riisgaard et al. (2004) suggested that changing nutrient loads may be reflected in changes in the biomass of ephemeral macroalgae rather than phytoplankton, if grazing forces (filterers) prevail in Odense Fjord in the control of phytoplankton biomass.

Pressures and impact analysis

Odense Fjord is clearly impacted by pressures from nutrients—mainly nitrogen and phosphorus loads from land—and hazardous substances as well as by physical pressures (Fyn County, 2003; Environment Centre Odense, 2007; Riisgaard et al., 2008). The impact of the nutrient load is the main subject treated here.

Land use in Odense River Basin is dominated by agriculture. Farmland accounts for ca. 68% of the land use, whereas urban areas (16%), woodland (10%) and the sum of inland freshwaters and natural countryside (6%) account for the rest (Fig. 1). Almost all (90%) of the sewage produced by the inhabitants is discharged to municipal treatment plants.

The dominant crops grown in the catchment are winter and spring cereals, accounting for about two-thirds of the total crop production area. Livestock is

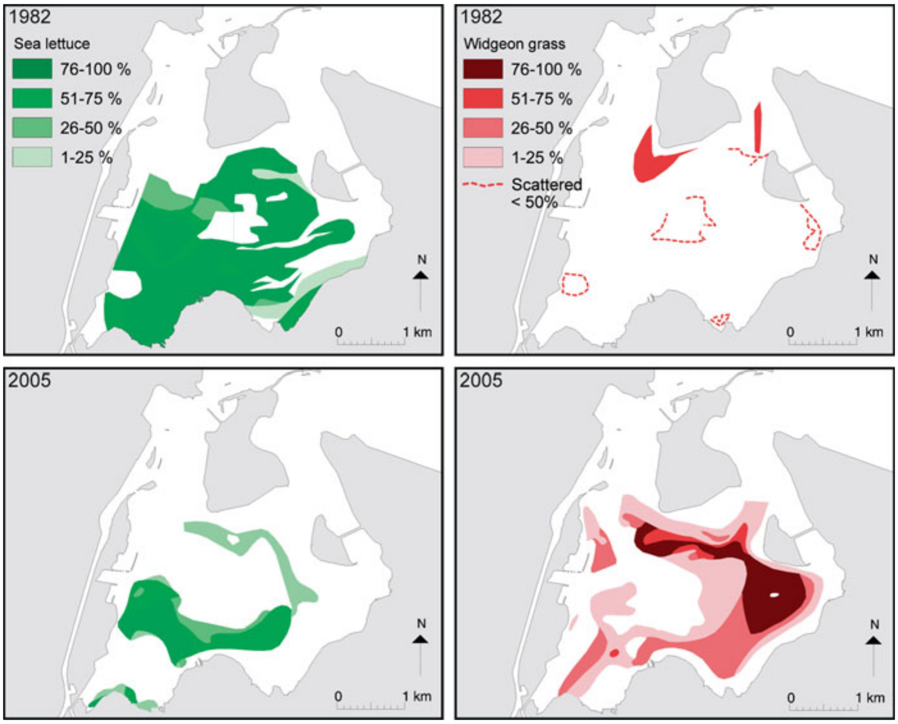

Fig. 4 Coverage of sea lettuce (*Ulva lactuca*; left) and widgeon grass (*Ruppia maritime*; right) in Seden Strand in 1982 (*upper*) and 2005 (*lower*)

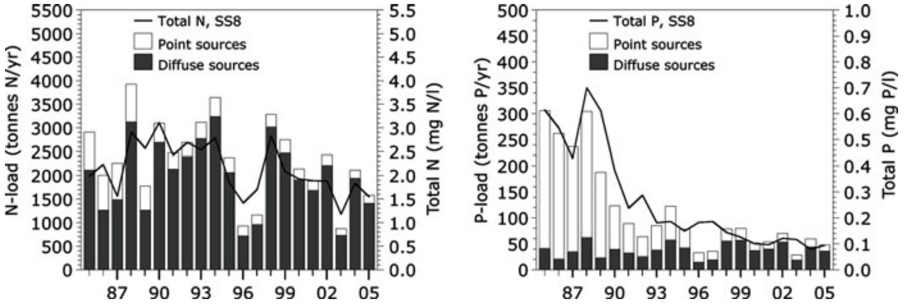

Fig. 5 Annual land-based nitrogen load (*left*) and phosphorus load (*right*) to Odense Fjord and annual mean total nitrogen and phosphorus concentrations at stn. SS8 in Seden Strand, 1985–2005; nutrient loads are partitioned between diffuse loads and wastewater (point source) loads

dominated by pigs, accounting for more than 60% of the total in terms of livestock density per area of farmed land. The total livestock production (including meat, milk and eggs) has increased by ca. 40% between 1985 and 2003. The application of nitrogen and phosphorus to the farmed land (69,000 ha) amounts to 165 kg N ha^{-1} and 28 kg P ha^{-1}. Manure accounts for ca. 40% and 73% of the applied nitrogen and phosphorus, respectively.

The dominance of farmland in the Odense River Basin area is reflected in the nutrient loading to Odense Fjord (Fig. 5). For nitrogen, the diffuse loading is clearly dominated by sources derived from agricultural activities, while point sources, i.e. the combined load from sewage outlets, account for a higher fraction of the phosphorus loading. This implies that variations in the freshwater runoff determine the nitrogen loading to a larger degree than the phosphorus loading (Rask et al., 1999).

Irrespective of this difference, the impact of nutrient loading on the nutrient concentrations in the estuary on an annual scale is apparent from the very high degree of co-variation for both nitrogen and phosphorus (Fig. 5). Owing to both national legislative initiatives—various action plans on the aquatic environment (Ærtebjerg et al., 2003)—as well as regional action, nutrient loads have decreased since around 1990; they are presently (average 1999–2004) at an annual level of about 2,100 tonnes N (or ca. 34 g N m^{-2} of estuary surface) and about 55 tonnes P (or ca. 0.9 g P m^{-2}). As a result of the load reductions, annual concentration levels of phosphorus in the inner estuary have decreased by a factor of 5 to 6, but only by ca. 1.5-fold for nitrogen (Fig. 5). Improvements have primarily resulted from action against point sources, i.e. by greatly improved sewage treatment. Consequently, a further reduction in phosphorus and especially nitrogen loading must primarily come from measures relating to agricultural activities.

Odense Fjord—reference conditions, classification and assessment

In the following sections, we describe the steps necessary to produce a management plan for Odense Fjord in accordance with the principles outlined in the Water Framework Directive (Anon., 2000, 2003). Briefly, the initial step is to define reference conditions, which are crucial as an 'anchor' for a classification system that measures deviations from the reference condition using an ecological quality ratio, EQR, i.e. the ratio between the current status and the reference condition. The ecological status can be defined by five pre-defined quality classes—'high', 'good', 'moderate', 'poor' and 'bad'—along the EQR scale. The 'good–moderate' boundary value is the important management target, as it separates water bodies attaining an acceptable, at least 'good', ecological status from those that do not.

Reference conditions

Ideally, reference conditions should be determined from undisturbed, pristine areas with no (or only very minor) human impact. This is a difficult task (if not impossible, considering Danish coastal areas), however, because human disturbances are found practically everywhere. Thus, reference conditions will typically be established from historical data, by predictive (numerical or statistical) modelling, and by expert judgement, or by a combination of these approaches; expert judgement can be used when other sources fail. However, an element of expert judgement will almost always be necessary in the establishment of reference conditions, for example for historical data, it is often necessary to use expert judgement to correct for different methodologies compared to current data.

Odense Fjord is subdivided into two types and thus two separate water bodies—the mesohaline inner fjord (Seden Strand) and the polyhaline outer part of the fjord, for which the assignment of reference conditions and the subsequent steps in the WFD process should be carried out. Reference conditions in Odense Fjord for biological and physico-chemical indicators are established from a combination of historical data, and numerical and statistical modelling, with necessary assistance of expert judgement (Fyn County, 2003).

Historical data

Generally, the effects of nutrient enrichment of coastal ecosystems, i.e. eutrophication, are well documented (see for example Richardson & Jørgensen, 1996). There are several reasons why eelgrass, and especially its depth distribution, is a useful indicator in terms of environmental status, as reviewed by Krause-Jensen et al. (2005): it is a key organism; it is widely distributed in the northern temperate zone; the depth distribution is (relatively) easy to measure; there is extensive historical material with regard to the distribution; and it is sensitive to human impact (eutrophication). Eelgrass seems to be particularly sensitive to the extent of nitrogen loading through its strong regulation by light conditions (Hauxwell et al., 2003). For example, increased nitrogen availability leads to increased primary production and biomass of phytoplankton and rapidly growing ephemeral algae (Borum, 1996; Pedersen & Borum, 1997); this results in, among other things, less light being available for rooted macrophytes such as eelgrass and therefore a smaller depth limit.

Around 1900, a large amount of data were collected concerning the distribution and depth limit of eelgrass (*Zostera marina*) in Danish coastal waters, including Odense Fjord (Ostenfeld, 1908). At that time, anthropogenic nutrient loading was probably very low (Conley, 1999), although the inner part of Odense Fjord probably was affected by raw sewage

Table 2 Reference conditions and threshold values for eelgrass depth, Secchi depth and nutrient concentrations separating 'high' from 'good' and 'good' from 'moderate' ecological status in the two water bodies in Odense Fjord calculated as 15% and 25% deviations, respectively, from the reference condition

	Eelgrass depth, m		Secchi depth, m	Total N, µg N l^{-1}		Total P, µg N l^{-1}	
	Seden strand	Outer fjord	ODF17	SS8	ODF17	SS8	ODF17
Reference condition	4	6	7.2	666	374	29	22
'High' ecological status (15% deviation)	3.4	5.1	6	826	464	33	25
'Good' ecological status (25% deviation)	3	4.5	5.3	976	548	36	28

Nutrient concentrations are annual means; Secchi depths are March–October means

from the city of Odense. Taking historical data from other Danish estuaries also into consideration (Ostenfeld, 1908; Nielsen et al., 2003), eelgrass depth limits of 4 m for the inner fjord, Seden Strand, and 6 m for the outer fjord are suggested as reference conditions (Table 2), wherever depth as well as bottom substrate allow for growth.

Empirical modelling

Using the large amount of data from Danish fjords and near-coastal areas collected by the 'Danish National Aquatic Monitoring and Assessment Programme' and by regional monitoring programmes of the (former) Danish counties, Nielsen et al. (2002) determined the following empirical relationships:

$$\ln(Z) = 0.755 \times \ln(\mathrm{TN}) + 6,039 \quad r^2 = 0.547,\ n = 128 \tag{1}$$

$$Z = 0.787 \times \mathrm{SD} + 0.339 \quad r^2 = 0.606,\ n = 101, \tag{2}$$

where Z is eelgrass depth limit (m), TN is total nitrogen (µg N l^{-1}) and SD is Secchi depth (m); all data are March–October means. Total nitrogen and Secchi depth refer to the concentration and the depth, respectively, at stations either at or near the location of the eelgrass beds. Obviously, nitrogen concentrations and Secchi depths are linked through the biomass of phytoplankton.

Using the reference eelgrass depth limits and Eqs. 1 and 2 with appropriate conversion to annual TN values gives reference TN concentrations of 666 and 374 µg N l^{-1} for the inner (stn. SS8) and outer (stn. ODF17) parts of Odense Fjord, respectively, and a Secchi depth of 7.2 m for the outer part (Secchi depth is not relevant in the shallow inner fjord) (Table 2).

Numerical modelling

Although nitrogen is the primary limiting nutrient for phytoplankton production and most likely also for ephemeral macroalgae (e.g. sea lettuce, *Ulva lactuca*) in the inner part of Odense Fjord, there is evidence that phosphorus also has an impact. This is exhibited as limiting concentrations or availability for phytoplankton during spring and for sea lettuce in the early summer period (Krause-Jensen et al., 2002; Fyn County, 2003), and necessitates the inclusion of phosphorus in this process. There are no empirical relationships available with respect to phosphorus similar to those for nitrogen, but an available and useful tool is numerical modelling.

The model applied is the previously mentioned 3D MIKE model. The performance of the model, exemplified by seasonal patterns of total nitrogen and phosphorus in a model run of 2004 (simulation 2004), is shown in Fig. 6. The correspondence between model simulation and measurements is generally good.

In addition to model runs using the actual forcing variables prevailing in a given year, scenario modelling has been carried out for selected years. In this type of modelling exercise, specific forcing variables may be changed (e.g. nutrient runoff), whereas others remain unchanged (e.g. freshwater discharge, climate variables) for a given year, and impacts of, in this case, an altered load regime can be evaluated. For specific interest here, a so-called 'Natural state' scenario—comparable to a reference 'year 1900 situation' with respect to nutrient loading (Conley, 1999)—has been used as an alternative means of developing reference conditions for Odense Fjord ('Natural state 2004' scenario run; Fig. 6). For this, the nutrient loading has been markedly reduced as

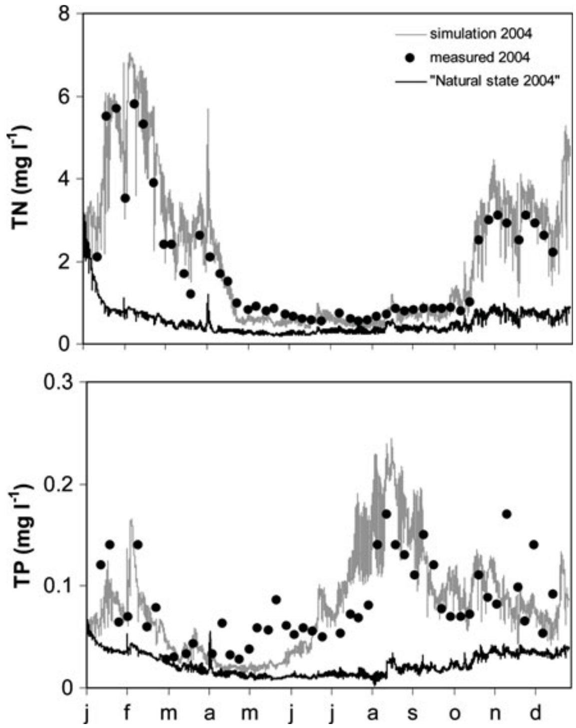

Fig. 6 Seasonal variation of total nitrogen and total phosphorus in MIKE3 model runs for 2004 (simulation 2004) shown together with measured values (measured 2004). Model output from the 'Natural state 2004' scenario run with reference nutrient loading (see text) is also shown

concentrations have been set at 1.0 mg N l^{-1} and 0.05 mg P l^{-1} in the watercourses running into Odense Fjord (based on expert judgement considering conditions in certain Danish and Baltic 'reference streams'), but the same freshwater discharge and meteorology have been used as for the actual year 2004, i.e. the 'simulation 2004' run (the freshwater discharge in 2004, ca. 350 million m^3, is relatively close to the long-term 1975–2004 mean). In order to meet expected conditions in a 'Natural state', various process and pool specifications and other forcing variables, in addition to the altered nutrient loading, have been modified. In the absence of other, more direct approaches, the 'Natural state 2004' scenario thus provides the reference conditions for phosphorus in Odense Fjord: 29 µg P l^{-1} at stn. SS8 in the inner fjord and 22 µg P l^{-1} at stn. ODF17 in the outer fjord, respectively (annual means; Table 2).

The annual means for total nitrogen in the 'Natural state 2004' scenario were 540 and 320 µg N l^{-1} at stn. SS8 and ODF17, respectively. This corresponds well with the above-mentioned 666 and 374 µg N l^{-1} (where a combination of historical eelgrass data and empirical modelling was used), bearing in mind that the estimates are based on independent and very different approaches. We used the latter nitrogen data as reference conditions despite the uncertainties, as they are found on historical observations which must rank above numerical scenario modelling.

Classification and assessment

Defining an acceptable deviation from the reference condition, i.e. setting the boundary between 'good' and 'moderate' ecological status, is difficult (unless it is a strictly political decision), as it involves translating the normative definitions of the WFD into numeric class boundaries. We used 25% and 15% deviation in this analysis as the boundary between 'good' and 'moderate' and between 'high' and 'good' ecological status, respectively (Table 2), in line with others (Krause-Jensen et al., 2005; Andersen et al., 2006). These class boundary values are shown in Fig. 7 in relation to past and present monitoring results (1984–2005). Phosphorus and, to a lesser extent, nitrogen concentrations have decreased in the fjord since the 1980s (Fig. 5), as discussed above. The decrease is highly significant for both nutrients (Kendalls-τ; $P < 0.0005$ and $P < 0.005$, respectively). The concentrations have approached the decisive 'good/moderate' boundary during the period shown in Fig. 7, but at present are a factor of about 2–3 higher; nutrient concentrations in the outer fjord exhibit a similar pattern (data not shown). Thus, for neither of the nutrients has 'good' status been attained.

Corresponding to the nutrient decrease, the Secchi depth has increased significantly (Kendalls-τ; $P < 0.005$) during this period, whereas the eelgrass depth limit did not increase and is currently 2.5–3 m (Fig. 7). Irrespective of the temporal patterns, neither the Secchi depth nor the eelgrass depth limit are close to the values needed for attaining 'good' ecological status. Moreover, the current eelgrass depth limits are not directly comparable to the class boundary values because the latter are based on the historical reference depths (Ostenfeld, 1908). Historical depths were determined by samples taken by rake from a boat and are thus considered to represent the depth of dense eelgrass stands, whereas current eelgrass

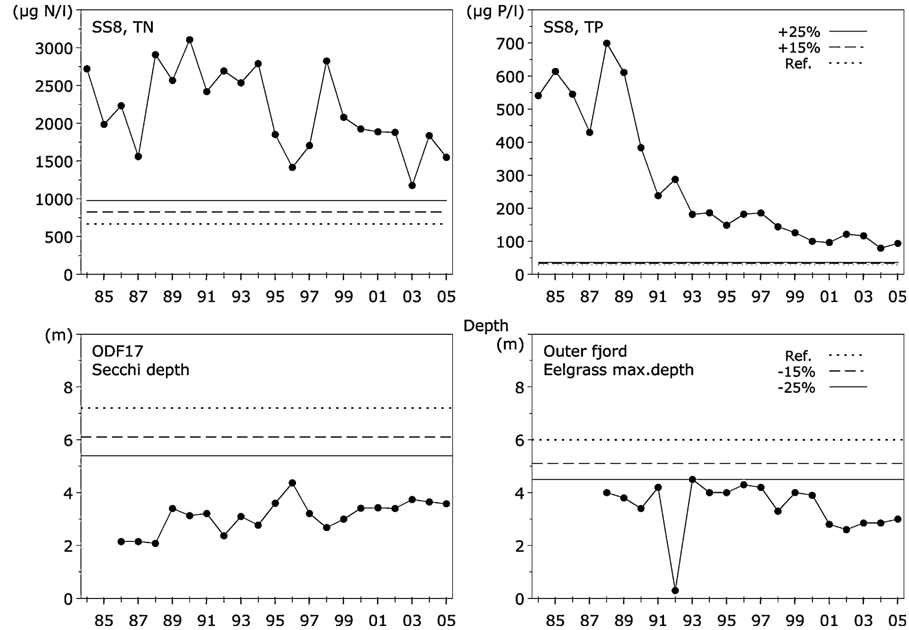

Fig. 7 *Upper panel*: Time-weighted annual means of total nitrogen (*left*) and total phosphorus (*right*) at stn. SS8 in the inner fjord, 1984–2005. *Lower panel*: Time-weighted March–October means of Secchi depth at stn. ODF17, 1986–2005 (*left*) and maximum eelgrass depth limit ('last straw') during July–August in the outer fjord, 1988–2005 (*right*). Actual data are compared to reference conditions and 15% and 25% deviations from reference conditions (Table 2), the latter expressing the boundary between 'good' and 'moderate' ecological status

depths, determined by diver, represent the maximum depth distribution, 'the last straw' (Krause-Jensen et al., 2005). This only increases the difference between the current status and the environmental objective.

Thus, despite the significant changes in nutrient concentrations and Secchi depth, no decisive changes have occurred for the eelgrass depth limit, although this was expected based on the empirical relations of Nielsen et al. (2002). These empirical relations comprise a wide range of depths and concentrations, and they provide general, functional associations between eelgrass and water quality in Danish coastal waters; however, they may not always be able to accurately predict the depth limit in a specific fjord at a given level of light and nutrients. Thus, other factors may be superimposed on the fundamental relation: physical features such as variations in suitable substrate, factors such as oxygen depletion and sulphide exposure, and the presence of impacting hazardous substances, as well as recruitment problems leading to time-lags due to slow recolonisation (Jensen et al., 2004; Greve & Krause-Jensen, 2005a, b). Furthermore, threshold effects such as hystereses, points of no return, etc., possibly leading to structural shifts, might also come into force (e.g. Scheffer et al., 2001). The important thing to keep in mind, however, is that meeting the environmental objectives in terms of light and nutrients is a prerequisite for a marked improvement in the eelgrass depth distribution (and its subsequent maintenance, for example by self-protection against erosion), whereas other possible factors working against an improvement must be dealt with separately.

Another point to be made is that eelgrass depth limits and coverage can show a different temporal evolution. Even though Fig. 7 shows a relatively constant maximum depth in recent years, the coverage has declined and the maximum coverage has been <10% in recent years (data not shown). This calls for investigations as to the quantitative use of macrophyte coverage as an important biological indicator in addition to depth distribution.

We have not made a 'proper' assessment sensu WFD, i.e. using EQR values for several indicators within all quality element groups, the 'one-out-all-out' principle, and other WFD assessment features. Such an exercise has been conducted for Odense Fjord by HELCOM (2009) using more indicators than in this work; this assessment corroborated completely with the findings presented above. It is also completely in line with the past ca. 25 years of continuous monitoring and assessment of Odense Fjord. Hence, despite the improvements outlined above, the environmental conditions in Odense Fjord are not in compliance with 'good' ecological status.

Odense Fjord—risk analysis

Following the assessment, a risk analysis is intended to evaluate the likelihood of whether the designated water bodies in Odense Fjord will be able to meet at least ‘good’ ecological status by 2015, taking the currently adopted measures into consideration. It is thus necessary to establish cause–effect relations, i.e. quantitative links between impacting factors and environmental indicators, to quantify the target for attaining at least ‘good’ ecological status if the water body is at risk of not fulfilling the objective. Finally, the programme of measures specifies the management activities and strategies that should be taken to reach the target in an optimal, cost-effective way (Anon., 2000). This will be discussed in this and the next section.

When considering the risk of not fulfilling the environmental objectives by 2015, the expected future impact of nutrients is the main issue. The assessment above showed that nitrogen is a major problem. In coastal areas with the characteristics of Odense Fjord, a functional relation between nitrogen loading and nitrogen concentrations can often be found. We have used a number of MIKE 3 scenario modelling runs for the year 2004 with variable nitrogen loading to establish a relation between annual nitrogen load and total nitrogen concentration (annual mean) in Odense Fjord. This relationship is linear for the two stations in the inner and outer fjord (Fig. 8). By combining this relationship with the relationship between eelgrass depth and nitrogen concentrations in Eq. 1, the environmental objectives

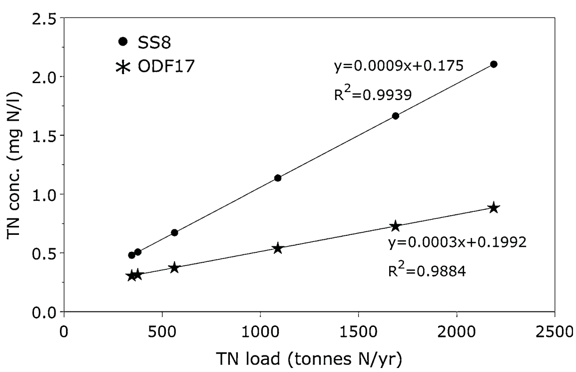

Fig. 8 Annual surface water concentration of total nitrogen in the inner (stn. SS8) and outer (stn. ODF17) parts of Odense Fjord as a function of the annual N load of Odense Fjord using various scenario runs of the MIKE 3 model (model year: 2004)

Table 3 Calculated allowable annual nitrogen load to Odense Fjord (including atmospheric deposition) in a reference condition and at ‘good’ and ‘high’ ecological status

	N load, tonnes N y^{-1}	
	Seden strand	Outer fjord
Reference condition	543	586
‘High’ ecological status (15% deviation)	721	886
‘Good’ ecological status (25% deviation)	888	1,166

of eelgrass depth are linked to an impacting factor, the nitrogen loading (Table 3). It appears that the maximum nitrogen loading allowed to attain at least the desired ‘good’ ecological status is different for the inner and the outer fjord. It is obviously necessary to ensure this status in the whole fjord, hence the lower value, 888 tonnes N, is the annual target load for Odense Fjord. Given the various uncertainties, the annual target load is thus 900 tonnes N (which also corresponds to a ‘high’ ecological status in the outer fjord; Table 3).

It is necessary to establish a current N loading to calculate the necessary annual load reduction. The nitrogen load in 2004 and the annual mean for the 1999–2004 period, both approx. 2,100 tonnes N (including an atmospheric deposition of about 100 tonnes), are a fair representation of the current annual nitrogen load at a (long-term) mean freshwater discharge. Thus, an annual N load reduction of 1,200 tonnes (2,100–900 tonnes N) is needed for fulfilment of the objectives.

What remains in the risk analysis is to evaluate the effect of the adopted/planned activities for reducing the nitrogen load—the so-called ‘baseline 2015’. These activities are measures implemented through existing regulatory planning and control processes, some of which will first become detectable in the water courses during the coming decade. These measures are calculated to provide a total load reduction of ca. 350 tonnes N y^{-1}, of which increased efficiency in agriculture and set-aside areas each contributes more than 40%, while reduction in point source loading only contributes about 5% (see details below). Thus, supplementary measures that provide reduction of a further ca. 850 tonnes N per year (1,200–350 tonnes N) are needed to fulfil the objective of ‘good’ ecological status in Odense Fjord.

Phosphorus loading has been mentioned to have an impact on Odense Fjord, as phosphorus temporarily limits phytoplankton and ephemeral macroalgae growth. This necessitates a reduction in P load along with the N load reduction. There is a general lack of well-working functional relationships linking phosphorus availability (or load) to environmental indicators for coastal systems (unlike for freshwater systems). Further uncertainties are related to the magnitude and dynamics of the sediment pools of phosphorus, especially in relation to numerical scenario modelling. Thus, at present, it is not possible to quantify a target load for phosphorus in relation to the marine environmental objectives.

Odense River Basin—programme of measures

The programme of measures (POM) is the heart of each river basin management plan, specifying the management activities and strategies needed to fulfil the objectives of the Water Framework Directive for identified water bodies. According to the WFD schedule, the river basin management plan should be issued in 2009, the POM made operational in 2012, and the environmental objectives fulfilled by 2015 (unless special circumstances are involved) (Anon., 2000).

In parallel with the risk analysis of Odense Fjord in the previous section, similar analyses show that 'good' ecological status will not be reached before 2015, with basic (baseline) measures, for the majority of the other surface water and groundwater bodies in Odense River Basin (Table 4). Therefore, nutrient target loads have been calculated for lakes and groundwater bodies within the catchment of Odense Fjord, whereas good status for watercourses is mainly related to the alleviation of physical and hydromorphological pressures.

The priority measures to attain the objectives of surface and ground waters in Odense Fjord River Basin include measures regarding sewage outlets from households and industry and measures to reduce diffuse loads of polluting substances from agriculture, including waterborne as well as airborne pollutants. Measures to minimise impacts from physical pressures include re-meandering of regulated watercourses, regaining free passage for migrating fish in watercourses, and regaining retention capacity (nutrients, etc.) in river beds (reconstruction of wetlands).

Economic analyses have been undertaken on a sub-catchment scale, including eleven lakes and the residual estuary catchment area, and five groundwater reservoirs. The reduction effect and unit costs of various measures regarding nitrogen emissions, as the main or sole effect, have been quantified. This makes it possible to rank and implement measures according to their cost-effectiveness in reducing nitrogen inputs to the aquatic environment.

Integrated analyses of measures are important to ensure a coherent POM, and subsequently an integrated river basin management plan that ensures the fulfilment of objectives in all water bodies within the river basin at the lowest cost to society. It has been

Table 4 Risk analysis of all designated water bodies in Odense Fjord River Basin

	Water bodies at risk, %	Main reasons for not fulfilling objectives	Operational targets in excess of 'Baseline 2015' measures
Watercourses/rivers	96	Physical and hydro-morphological conditions, regulation of rivers and river valleys due to land reclamation, waste-water outlets, storm water, scattered settlements	Discontinued maintenance (regular weed cutting and sediment removal) and rewinding of watercourses
Lakes	86	Nutrient loads from agriculture	Total reduced N and P load of $\sim$50 and $\sim$1 tonnes per year, respectively (11 largest lakes)
Coastal waters (Odense Fjord)	100	Nutrient loads from agriculture, hazardous substances	Reduced N load of $\sim$850 tonnes per year, reduced P load
Groundwater tables	50	Pesticides, other hazardous substances and nitrate load, high abstraction levels	N leaching from root zone in nitrate-sensitive areas <25 mg l^{-1} [a]

[a] Reduction of N leaching necessary in 1/3 of nitrate-sensitive areas; measures at play will also reduce pesticide loads

taken into account that a specific measure can affect the quality of more than one water type; for example, some measures addressing the protection of lakes, groundwater and watercourses will also enhance the protection of Odense Fjord.

Furthermore in this connection, special attention has been paid to integrate interactions between sub-catchments/water bodies in the river basin. Measures to meet objectives in most upstream sub-basins are thus identified first, followed by an evaluation of the impact that these measures will have on downstream sub-catchments, etc. Obviously, this entails working with simplified assumptions regarding hydrology and ecological synergistic effects. With this approach, however, it is possible to work with individual nutrient retention factors in as many catchment areas as selected.

Although the integrated effect of measures is of considerable magnitude, it is, nonetheless, far from possible to obtain the nutrient reductions needed to meet 'good' ecological status in Odense Fjord by only addressing groundwater and surface freshwater bodies (lakes and water courses) in the river basin. A specific targeting of additional measures related solely to Odense Fjord has thus been necessary.

Cost-effectiveness analysis

A spreadsheet model prepared specifically for the purpose of the analysis, including data on potentials, effects and unit costs of measures, has been used to analyse the economic and environmental consequences of alternative scenarios of supplementary measures (i.e. in addition to reduced loadings from baseline measures) to fulfil the WFD objectives for water bodies within the Odense River Basin. Measures are ranked and implemented or included according to their cost-effectiveness in reducing nitrogen inputs to the aquatic environment. Measures necessary to achieve 'good' status of water bodies in relation to parameters other than nitrogen, such as physical and hydro-morphological pressures, phosphorus loads, etc., are also included.

It should be noted that costs associated with the baseline are for already planned measures that have not yet been fully implemented. In this way, the baseline does not include costs of fully implemented measures within and prior to the last 20 years.

Scenarios

The two most cost-effective scenarios fulfil the objective of making the necessary reduction in nitrogen loading to Odense Fjord of 900 tonnes per year (Table 3) in excess of reduced loadings from baseline measures. The two scenarios aim at target fulfilment according to the WFD in all surface and groundwater bodies. On the basis of the two WFD scenarios, a third scenario has been analysed to evaluate the economic effect of including simultaneous consideration of target fulfilment for terrestrial natural habitats, for example in NATURA 2000 designated areas.

It should be emphasised that for all scenarios, measures to reduce point-source pollution are implemented to fulfil objectives regarding pollution of watercourses, lakes and marine waters with oxygen-consuming organic substances, phosphorus, bacteria, etc., rather than to reduce nitrogen loads to the aquatic environment per se. Measures to reduce point-source pollution take up almost all the baseline costs (Table 5). Overall, costs associated with the baseline scenario (16.9 million €/year) are higher than the costs of supplementary measures necessary to fulfil requirements of the WFD for all scenarios (12.6–15.8 million €/year).

In scenario 1 ('Mixed scenario'; Table 5), importance is attached to increased environmental efficiency in agricultural production. The combination of measures is aimed partly at increased environmental efficiency and partly at set-aside cultivated land. The most cost-effective measures are '*increased utilisation of animal manure*' (1.82–4.95 €/kg N), '*catch crops*' (1.43–3.77 €/kg N) and '*reduced N-norm application in river valleys*' (3.77 €/kg N). Scenario 1 results in a change in agricultural practice on approximately 19% of the cultivated land area, of which 8% is converted to wetlands, 9% to permanent grassland, and 2% to forest.

In scenario 2 ('Wetland scenario'; Table 5), importance is attached to set-aside cultivated areas that implies a conversion in agricultural practice on approximately 23% of the cultivated area within the river basin. In general, it is more cost-effective to set aside agricultural land in lowland areas (e.g. river valleys) than on higher ground owing to the lower nitrogen retention capacity in the lower areas. Scenario 2 is, to a considerable extent, based on set aside for

Table 5 Overall results of the cost-effectiveness analysis—baseline and scenarios 1 and 2

	Baseline	Scenario 1 Mixed scenario	Scenario 2 Wetland scenario
Socio-economic annual costs (1000 €/yr)			
Increased efficiency in agriculture	402	867	1
Set-aside[a]	671	3,009	4,700
Set-aside[b]		1,387	1,387
Improvement of groundwater quality		1,990	1,990
Reduction of point-source pollution	15,829	5,353	5,353
Total	16,902	12,604	13,430
N load reduction in 12 recipients (tonnes N y^{-1})			
Increased efficiency in agriculture	167	297.6	0.2
Set-aside[b]	145	356	653
Set-aside (%)[c]		204	204
Improvement of groundwater quality		44	44
Reduction of point-source pollution	18	8	8
Indirect effects from other lake catchments	12	29	28
Total	342	937	937
Average cost-effectiveness (€/kg N)	51.11	13.82	14.76
Set-aside			
Set-aside (ha)	1,279	12,479	15,452
Set-aside (%)[c]	2	19	23

[a] Conversion of agricultural practice to forest, wetlands or permanent grassland

[b] Conversion of agricultural practice to wetlands or permanent grassland in connection with physical and hydro-morphological improvements in watercourse areas

[c] % of total agricultural area

wetlands in river valleys (converting 9% of the cultivated area), which is among the most cost-effective measures (5.46 €/kg N), with permanent grassland (converting 8% of the cultivated area at a less efficient 15.73 €/kg N) as a supplementary measure to fulfil WFD targets. Afforestation makes up 3%. An additional 3% of the cultivated area is needed for re-establishment of wetlands to improve physical and hydro-morphological conditions in watercourses.

The analysis shows that re-establishment of wetlands and reduced fertilisation norms are the most effective measures, if large reductions in N loads to the aquatic environment are to be achieved (Scenarios 1 and 2). The potential for re-establishing wetlands in the Odense Fjord catchment is quite high, as about 72% of former wetlands in the area have disappeared since 1890 owing to land reclamation (Fig. 9). Some of these areas could potentially be restored. A major fraction of these areas once formed part of Odense Fjord, and their restoration would, in addition to nutrient reduction, enhance the natural value of the Natura 2000-designated parts of the estuary.

It is important, however, to bear in mind that whereas wetlands are an efficient tool for combating eutrophication when nitrogen has leached to the surface waters, wetland restoration does not change the amount of nitrogen applied to the fields. Thus, the application rate per hectare remains unaltered. Wetlands are consequently not the solution for protecting groundwater resources from nitrate or for protecting vulnerable natural habitats from airborne nitrogen loading.

The total socio-economic costs of the two scenarios of 12.6 and 13.4 million €/year, respectively, can be compared to the cost of currently implemented measures on sewage treatment within the catchment, which is in the order 40 million €/year, and the costs of currently implemented measures to reduce nutrient loads from agriculture, of approximately 1 million €/year. It has been estimated that the total income and production value from households, industry and agriculture is 15,650 million €/year in the Odense River Basin. Implementing the WFD will increase the total expenses for water services from 82 to 95 million €/year, which is 0.6% of the total income and production value in the basin. From this, it is evident that the investments to obtain the needed nitrogen load reductions by agricultural measures are economically feasible, and far below investments already undertaken to reduce point-source pressures.

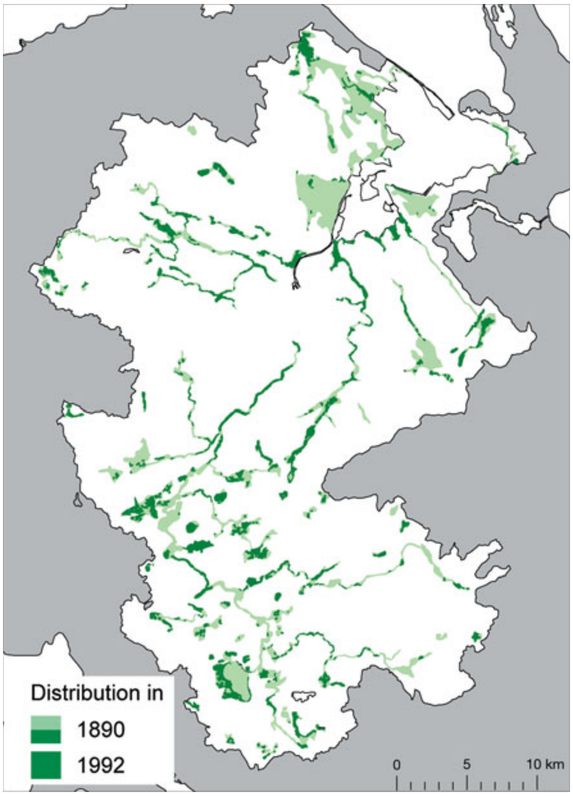

Fig. 9 Distribution of wetlands (meadows and bogs) in Odense River Basin, 1890 (*sum of light and dark green*) and 1992 (*dark green only*)

The financial costs of measures, however, can be affected by a change in future activities. If, for example, livestock production is allowed to increase, thereby 'commandeering' part of the cheapest means of reducing the environmental impact of the expanded agricultural production, this will indirectly increase the costs of reducing the environmental pressure from existing agricultural production. If supplementary environmental measures are not implemented in connection with expansion of livestock production, nutrient loss (waterborne and airborne) to the surrounding areas will increase.

The analysis shows that it is possible to implement environmental measures within agriculture that will reduce nitrogen loading of Odense Fjord by approx. 1,200 tonnes per year—including basic (baseline) measures—as required. This is done without reducing livestock production in the river basin. However, a reduction in cultivated area of 19–23% will necessarily result in reduced crop production.

In Scenario 3 ('Nature scenario'; Table 6), target fulfilment for wet and dry terrestrial natural habitats according to the regional planning system, Natura 2000, and the Rio declaration (on the preservation of biodiversity) is considered in addition to the objectives of the WFD. In areas covered by other directives, such as those designated as Natura 2000 areas, further measures will, in many cases, be necessary to ensure fulfilment of specific requirements, for example 'good conservational status' in the Habitats Directive. In some water bodies, this may correspond to achieving 'high' ecological status according to the WFD. The nitrogen load will then have to be reduced even further (cf. Table 3). It is estimated that a doubling of the natural habitat area, a 50% reduction in gaseous emissions of ammonia, and preservation and improved hydrological conditions in existing natural habitats will be necessary to meet objectives for the terrestrial natural habitats.

Instead of considering measures to meet objectives for natural terrestrial habitats as separate from WFD objectives for water bodies, integration of the two allows for synergistic effects and hence overall cost minimisation. It is estimated that the costs of meeting the objectives for natural terrestrial habitats will be approximately 15.8 million €/year (Table 6); of this, costs to reduce gaseous emissions of ammonia account for about 12 million €/year. However, including considerations about the placement of the cost-effective quantity of set-aside areas for wetlands and permanent grassland in scenarios 1 and 2, and coordinating this with existing soil conditions and occurrence of existing natural habitats, make it possible to achieve the necessary doubling of natural habitat area by implementing the WFD in the river basin. Hence, additional expenses of approximately 2.8 million €/year can be avoided. Moreover, the implementation of either scenario 1 or 2 has been estimated to result in a reduced emission of ammonia, decreasing the need for additional measures costing approximately 1 million €/year. In total, approximately 3.8 million €/year in expenses additional to those of scenario 1 and 2 can potentially be avoided by integrating the planning of measures needed to fulfil targets according to the various Directives to protect natural habitats and the aquatic environment. However, there will still be expenses of approximately 12 million €/year additional to those of scenarios 1 and 2 to meet objectives for natural terrestrial habitats.

Table 6 Objectives and costs of scenario 3 ('Nature scenario') for target fulfilment in the Odense River Basin for terrestrial natural habitats according to the regional planning system, Natura-2000 and the Rio agreement concerning preservation of biodiversity

Measures	Objectives	Annual costs, 1000 €/yr
Increased natural habitat areas	+100% (ha)	
Salt marsh	450	255
Marsh/meadow	2,400	2,281
Commons	600	255
Reduced gaseous emissions of ammonia from animal husbandry (>35 LU*) by the addition of sulphuric acid to slurry (best available practice)	50% reduction	12,060
Nature preservation		
Grazing, etc., of existing natural habitats	2,347 ha	335
Clearing	360 ha	174
Improved hydrological conditions		
Deactivation of drains	300 km	429
Total		15,789

*Livestock units

A Baltic perspective

Although the future conditions in Odense Fjord will mainly be determined by the success of the efforts in reducing local impacts outlined above, the development in the whole Baltic catchment area may play an important role.

The Baltic Sea itself is perceived as being in an unacceptably polluted state (e.g. Turner et al., 1999). Excessive nutrient loading leads to unwanted algal growth and oxygen deficit, which destroy habitat conditions for a large proportion of the naturally occurring species and often render the water unsuitable for bathing (Bonsdorff et al., 1997; Wulff et al., 2001; HELCOM, 2009).

It can be estimated that if agricultural production and nitrogen loss from all farmland in the Baltic Sea catchment increased to the prevailing high level in Denmark, then the total nitrogen loss to the Baltic Sea would probably increase by more than 50%. In contrast, if agricultural production and nitrogen loss from all farmland in the Baltic Sea catchment were reduced to the present low level in Poland, then the total nitrogen loss to the Baltic Sea would probably decrease by 10–25%.

Continued growth in livestock production in the EU, especially in the new EU countries including Poland (which is the EU country with the greatest area of agricultural land) is expected and further, mineral fertiliser use is expected to soar (EAE, 2005). All things being equal, such growth will entail enhanced pressure on the environment unless special environmental measures are implemented concomitantly.

The ecological effects on the Baltic Sea subbasins, the Kattegat, and the Belt Sea area of a 14% increase in nitrogen loading of the Baltic Sea due to increased losses from Polish agriculture have been modelled by Hansen et al. (2003). The results show that increased agricultural production in Poland can potentially have a considerable impact on the ecological conditions in the Kattegat and the Belt Sea area, by e.g. increasing the area of oxygen deficit by 25% to above 50% owing to the increase in primary production. Obviously, the future conditions of Odense Fjord may thus be affected, depending on our success in reducing the nutrient load from the Baltic catchment area.

Conclusions and recommendations

Odense Pilot River Basin

During implementation of the WFD, it is necessary to develop tools for defining reference conditions to which a classification system for an assessment of designated water bodies is anchored; the definition of an acceptable deviation from reference conditions is particularly important as it defines the management target of good ecological status. Subsequently, it is necessary to establish quantitative cause–effect relations, i.e. links between impacting factors and environmental indicators. We have presented an example of an indicator, eelgrass (*Zostera marina*), in the 62-km^2 Danish estuary Odense Fjord, where eelgrass depth distribution is linked to the anthropogenic nitrogen loading from land via nitrogen concentrations in the estuary by using a combination of historical data, empirical and numerical modelling, and a necessary touch of expert judgement. This example indicated that an almost 60% reduction of the current N load was necessary to meet the environmental objectives for the eelgrass depth distribution (defined as 25% deviation from the reference condition).

Risk analyses of all surface water and groundwater bodies in the Odense River Basin have shown that the majority of the water bodies in the catchment are at risk of not fulfilling the WFD environmental objective of 'good' ecological status before 2015 with the measures already planned. This is mainly due to high nutrient loads of especially nitrogen. It is thus very important to address the pressures in an integrated way (considering all types of surface waters and groundwater) to minimise the economic costs, and to obtain the optimal synergistic effects of the proposed measures. Furthermore, the costs of meeting objectives for natural terrestrial habitats should be taken into account, where it is relevant and possible.

Development of an integrated programme of measures, based on quantitatively defined target loads of nitrogen for all water bodies in the Odense River Basin, showed that reduction of the nutrient load from the main source, agriculture, is possible and economically feasible. The costs associated with the proposed measures are of a considerably smaller magnitude than the investments already undertaken to reduce primarily point-source pressures in the national and regional action plans. It is thus demonstrated in Odense River Basin that it is not an impossible task, either economically or technically, to meet the objectives of the WFD while still retaining the possibility of keeping a high agricultural production in the catchment.

Water managers' challenge

The high growth rates in the new EU Member States pose an important challenge to water managers. Decoupling of economic growth from pressure on water bodies will be necessary to avoid a situation in which economic growth causes an increase in pressure instead of the decrease that is needed. Integrated management strategies urgently need to be strengthened to enable the characterisation of all important pressures on the aquatic environment and the development of efficient and coherent strategies to deal with these pressures in a cost-effective manner. Successful and cost-effective implementation of the WFD requires several important preconditions to be met, which are inadequately met at present. These preconditions include:

Legislation

The legislative possibilities to individually regulate pressures from all sectors of society—i.e. from the individual farm, industry or household—must be available at an early stage in the planning process. Present national legislation usually only provides limited possibilities to individually regulate pressure on local water bodies from agriculture and forestry. This is a major obstacle in relation to the preparation and implementation of river basin management plans aimed at ensuring attainment of the environmental objectives for the individual water bodies.

Resources

Adequate resources, both administrative and financial, must be allocated to water management. The financial principles governing implementation of the programmes of measures must be defined and the necessary resources allocated at an early stage of the process to set the framework for the planning and implementation process. When allocating resources for river basin management, a good proportion of the resources should be earmarked for ensuring public participation from the beginning of the planning process. Public information and stakeholder involvement are very important aspects of the process and vital for ensuring successful implementation of the river basin management plans as they maximise 'ownership' of the water management plans.

Monitoring

Comprehensive monitoring is vital for ensuring that the programmes of measures are cost-effective and for characterizing threats to water bodies of 'good' or 'high' status in due time to hinder deterioration of their status. Many water bodies are at present judged to be at risk owing to lack of data. Knowledge of the ecological status and relevant pressures on these water bodies will be a future demand. Monitoring is also vital to control the efficacy of implemented measures.

New technologies

Rapid societal growth necessitates research on environmental technologies to avoid the increased pressures on water bodies resulting from increased levels of activity, including new technologies to minimise pressures from intensified agricultural production.

Climate change

It is necessary for water managers to incorporate 'safety margins' when designing programmes of measures to fulfil WFD objectives. In the same way that engineers routinely incorporate safety margins when constructing for example bridges, it is vital that they are included in programmes of measures to ensure the fulfilment of the stated aims, or to allow for strongly increased pressure during extreme or rare climate conditions. For example, modelling the effects of expected climate change scenarios indicates a deterioration of oxygen conditions in Danish coastal waters as well as in other areas (Justic et al., 2001; Pejrup et al., 2006). Measures based solely on pressure levels expected during normal climate conditions should therefore be considered inadequate for ensuring fulfilment of WFD objectives. The balances of greenhouse gases should be incorporated in the alternative scenarios of programmes of measures.

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References

Andersen, J. H., L. Schlüter & G. Ærtebjerg, 2006. Coastal eutrophication: recent developments in definitions and implications for monitoring strategies. Journal of Plankton Research 28: 621–628.

Anon., 1991a. Council Directive of 21 May 1991 concerning urban waste water treatment (91/271/EEC). Official Journal L 135.

Anon., 1991b. Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. Official Journal L 375.

Anon., 2000. Directive 200/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal L 327/1.

Anon., 2003. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance on typology, reference conditions and classification systems for transitional and coastal waters. Produced by Working Group 2.4 – Coast.

Anon., 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).

Bonsdorff, E., E. M. Blomqvist, J. Mattila & A. Norkko, 1997. Long term changes and coastal eutrophication. Examples from the Åland islands and the Archipelago Sea, northern Baltic Sea. Oceanologica Acta 20: 319–329.

Borum, J., 1996. Shallow water and land/sea boundaries. In Jørgensen, B. B. & K. Richardson (eds), Eutrophication in Coastal Marine Ecosystems. Coastal and Estuarine Studies 52. American Geophysical Union, Washington DC: 243–267.

Conley, D. J., 1999. Biogeochemical nutrient cycles and nutrient management strategies. Hydrobiologia 410: 87–96.

Conley, D. J., H. Kaas, F. Møhlenberg, B. Rasmussen & J. Windolf, 2000. Characteristics of Danish estuaries. Estuaries 23: 820–837.

Edelvang, K., H. Kaas, A. C. Erichsen, D. Alvarez-Berastegui, K. Bundgaard & P. V. Jørgensen, 2005. Numerical modelling of phytoplankton biomass in coastal waters. Journal of Marine Systems 57: 13–29.

EEA, 2005. European Environment Outlook. EEA report No 4/2005. European Environment Agency, Copenhagen, Denmark.

Environment Centre Odense, 2007. Odense Pilot River Basin. Pilot project for river basin management planning. Water Framework Directive Article 13. Danish Ministry of the Environment – Environment Centre Odense.

Fyn County, 2003. Odense Pilot River Basin. Provisional Article 5 report pursuant to the Water Framework Directive. Fyn County, Denmark.

Greve, T. M. & D. Krause-Jensen, 2005a. Stability of eelgrass (*Zostera marina*) depth limits: influence of habitat types. Marine Biology 146: 803–812.

Greve, T. M. & D. Krause-Jensen, 2005b. Predictive modelling of eelgrass (*Zostera marina*) depth limits. Marine Biology 146: 849–858.

Hansen, I. S., T. Urenholdt & K. I. Dahl-Madsen, 2003. Miljøeffektvurdering for havmiljøet. Del 2: 3D procesbaseret modellering af miljøtilstanden i de åbne indre farvande. Institut for Miljøvurdering, Copenhagen (in Danish).

Hauxwell, J., J. Cebrián & I. Valiela, 2003. Eelgrass *Zostera marina* loss in temperate estuaries: relationship to land-derived nitrogen loads and effect of light limitation imposed by algae. Marine Ecology Progress Series 247: 59–73.

HELCOM, 2009. Eutrophication in the Baltic Sea – an integrated thematic assessment of the effects of nutrient enrichment and eutrophication in the Baltic Sea region. In Andersen, J. H. & M. Laamanen (eds), Baltic Sea Environmental Proceedings 115B. 148 pp.

Jensen, H. F., M. Holmer & I. Dahllöf, 2004. Effects of tributyltin (TBT) on the seagrass *Ruppia maritima*. Marine Pollution Bulletin 49: 564–573.

Justic, D., N. N. Rabalais, & R. E. Turner, 2001. Future perspectives for hypoxia in the northern Gulf of Mexico. In Rabelais, N. N. & R. E. Turner (eds), Coastal Hypoxia: Consequences for Living Resources and Ecosystems. American Geophysical Union, Washington DC: Coastal and Estuarine Studies 58: 435–450.

Krause-Jensen, D., C. Jensen, K. Nielsen, M. F. Petersen, D. F. Hansen, M. Laursen, E. M. Platz, P. B. Madsen, G. Bruntse, N. Rask, S. Larsen & E. Hvas, 2002. Næringssaltbegrænsning af makroalger i danske kystområder. Et samarbejdsprojekt mellem Ringkøbing Amt, Nordjyllands Amt, Viborg Amt, Århus Amt, Ribe Amt, Sønderjyllands Amt, Fyns Amt, Roskilde Universitetscenter og Danmarks Miljøundersøgelser. NERI Technical Report No. 293. National Environmental Research Institute, Roskilde, Denmark (in Danish).

Krause-Jensen, D., T. M. Greve & K. Nielsen, 2005. Eelgrass as a bioindicator under the European water framework directive. Water Resources Management 19: 63–75.

Nielsen, S. L., K. Sand-Jensen, J. Borum & O. Geertz-Hansen, 2002. Depth colonization of eelgrass (Zostera marina) and macroalgae as determined by water transparency in Danish coastal waters. Estuaries 25: 1025–1032.

Nielsen, K., B. Sømod, C. Ellegaard & D. Krause-Jensen, 2003. Assessing reference conditions according to the European Water Framework Directive using modelling and analysis of historical data: an example from Randers Fjord. Ambio 32: 287–294.

Ostenfeld, C. H., 1908. Ålegræssets (*Zostera marina*) vækstforhold og udbredelse i vore farvande. Beretning fra den danske biologiske station XVI. Centraltrykkeriet, Kjøbenhavn (in Danish).

Pedersen, M. F. & J. Borum, 1997. Nutrient control of estuarine macroalgae. Growth strategy and the balance between nitrogen requirements and uptake. Marine Ecology Progress Series 161: 155–163.

Pejrup, M., T. J. Andersen, K. Edelvang, A. Erichsen, K. Garde, K. Gustavsen, U. Lumborg, F. Møhlenberg, P. Rasch & K. Richardson, 2006. Klimaforandringers betydning for havet og de kystnære områder. In Søndergaard, M., B. Kronvang, M. Pejrup & K. Sand-Jensen (eds), Vand og vejr om 100 år. Klimaforandringer og det danske vandmiljø. Hovedland, Denmark: 90–106.

Rask, N., S. E. Petersen & M. H. Jensen, 1999. Response to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. Hydrobiologia 393: 69–81.

Rasmussen, B. & A. Josefson, 2002. Consistent estimates for the residence time of micro-tidal estuaries. Estuarine, Coastal and Shelf Science 54: 65–73.

Richardson, K. & B. B. Jørgensen, 1996. Eutrophication: definition, history and effects. In Jørgensen, B. B. & K. Richardson (eds), Eutrophication in Coastal Marine Systems. Coastal and Estuarine Studies 52. American Geophysical Union, Washington DC: 1–19.

Riisgaard, H. U., D. F. Seerup, M. H. Jensen, E. Glob & P. S. Larsen, 2004. Grazing impact of filter-feeding zoobenthos in a Danish fjord. Journal of Experimental Marine Biology and Ecology 307: 171–261.

Riisgaard, H. U., J. Lassen, M. Kortegaard, L. F. Møller, M. Friedrichs, M. H. Jensen & P. S. Larsen, 2007. Interplay between filter-feeding zoobenthos and hydrodynamics in the shallow Odense Fjord (Denmark) – Earlier and recent studies, perspectives and modelling. Estuarine, Coastal and Shelf Science 75: 281–295.

Riisgaard, H. U., M. H. Jensen & N. Rask, 2008. Odense Fjord and Kerteminde Fjord/Kertinge Nor. In Schiewer, U. (ed.), Ecology of Baltic Waters. Ecological Studies 197. Springer-Verlag, Berlin: 361–394.

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke & B. Walker, 2001. Catastrophic shifts in ecosystems. Nature 413: 591–596.

Turner, R. K., S. Georgiou, I.-M. Gren, F. Wulff, S. Barrett, T. Söderqvist, I. J. Bateman, C. Folke, S. Langaas, T. Zylicz, K.-G. Mäler & A. Markowska, 1999. Managing nutrient fluxes and pollution in the Baltic: an interdisciplinary simulation study. Ecological Economics 30: 333–352.

Vanderborght, J.-P., I. M. Folmer, D. R. Aguilera, T. Uhrenholdt & P. Regnier, 2007. Reactive-transport modelling of C, N, and O_2 in a river-estuarine-coastal system: application to the Scheldt estuary. Marine Chemistry 106: 92–110.

Wulff, F., E. Bonsdorff, I.-M. Gren, S. Johansson & A. Stigebrandt, 2001. Giving advice on cost effective measures for a cleaner Baltic Sea: a challenge for science. Ambio 30: 254–259.

Ærtebjerg, G., J. H. Andersen & O. S. Hansen, 2003. Nutrients and Eutrophication in Danish Marine Waters. A Challenge for Science and Management. National Environmental Research Institute: 126 pp.