

J.H. Andersen
D.J. Conley
Editors

Developments in Hydrobiology 207

Eutrophication in Coastal Ecosystems

*Towards better understanding
and management strategies*



Springer

Eutrophication in Coastal Ecosystems
Towards better understanding and management strategies

Developments in Hydrobiology 207

Series editor

K. Martens

Eutrophication in Coastal Ecosystems

Towards better understanding and management strategies

*Selected Papers from the Second International Symposium on
Research and Management of Eutrophication in Coastal
Ecosystems, 20–23 June 2006, Nyborg, Denmark*

Editors

J.H. Andersen¹ & D.J. Conley²

¹*DHI, Agern Allé 5, 2970 Hørsholm, Denmark*

²*GeoBiosphere Science Centre, Lund University, Sölvegatan 12, 223 62 Lund, Sweden*

Previously published in *Hydrobiologia*, Volume 629, 2009

Editors

J.H. Andersen
DHI, Agern Allé 5,
2970 Hørsholm,
Denmark

D.J. Conley
GeoBiosphere Science Centre,
Lund University, Sölvegatan
12, 223 62 Lund, Sweden

Cover illustration: A victim of coastal eutrophication. The location is Aarhus Bay, Denmark and the Common Starfish (*Asterias rubens*) has died as a result of hypoxia. Aarhus Bay is a shallow, stratified bay and receives high loads of nutrients, especially from municipal waste water treatment plants and neighboring coastal waters. Seasonal hypoxia, and sometimes anoxia, has been common in most parts of Aarhus Bay for more than two decades. Programmes of measures to reduce nutrient loadings have been implemented, but have so far not led to a good ecological status. Photo: Peter Bondo Christensen, National Environmental Research Institute, University of Aarhus, Denmark.

All rights reserved.

Library of Congress Control Number: 2009934012

DOI: 10.1007/978-90-481-3385-7

ISBN: 978-90-481-3384-0

e-ISBN: 978-90-481-3385-7

Printed on acid-free paper.

© 2009 Springer Science+Business Media B.V.

No part of this work may be reproduced, stored in a retrieval system, or transmitted in any form or by any means, electronic, mechanical, photocopying, microfilming, recording or otherwise, without written permission from the Publisher, with the exception of any material supplied specifically for the purpose of being entered and executed on a computer system, for exclusive use by the purchaser of the work.

springer.com

Contents

PREFACE

Eutrophication in coastal marine ecosystems: towards better understanding and management strategies
J.H. Andersen · D.J. Conley 1

INTRODUCTORY PAPER

Eutrophication and the macroscope
S.W. Nixon 5

REVIEW PAPER

Ecosystem thresholds with hypoxia
D.J. Conley · J. Carstensen · R. Vaquer-Sunyer · C.M. Duarte 21

OPINON PAPERS

Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management

D.L. Breitburg · J.K. Craig · R.S. Fulford · K.A. Rose · W.R. Boynton · D.C. Brady · B.J. Ciotti · R.J. Diaz · K.D. Friedland · J.D. Hagy III · D.R. Hart · A.H. Hines · E.D. Houde · S.E. Kolesar · S.W. Nixon · J.A. Rice · D.H. Secor · T.E. Targett 31

Assessment of the eutrophication status of transitional, coastal and marine waters within OSPAR

U. Claussen · W. Zevenboom · U. Brockmann · D. Topcu · P. Bot 49

Can the EU agri-environmental aid program be extended into the coastal zone to combat eutrophication?

O. Lindahl · S. Kollberg 59

Eutrophication and agriculture in Denmark: 20 years of experience and prospects for the future

N.P. Nørring · E. Jørgensen 65

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

J.D. Petersen · N. Rask · H.B. Madsen · O.T. Jørgensen · S.E. Petersen · S.V. Koch Nielsen · C.B. Pedersen · M.H. Jensen 71

Relationship between eutrophication reference conditions and boundary settings considering OSPAR recommendations and the Water Framework Directive—examples from the German Bight

D. Topcu · U. Brockmann · U. Claussen 91

PRIMARY RESEARCH PAPERS

Modelling the distribution and growth of ‘problem’ green seaweed in the Medway estuary, UK

J.N. Aldridge · M. Trimmer 107

Short-term effect of oxic to anoxic transition on benthic microbial activity and solute fluxes in organic-rich phytotreatment ponds

M. Bartoli · L. Vezzulli · D. Nizzoli · R. Azzoni · S. Porrello · M. Moreno · M. Fabiano · P. Viaroli 123

Denitrification efficiency for defining critical loads of carbon in shallow coastal ecosystems

B.D. Eyre · A.J.P. Ferguson 137

Responses of benthic–pelagic coupling to climate change in a temperate estuary

R.W. Fulweiler · S.W. Nixon 147

Potential phytoplankton indicator species for monitoring Baltic coastal waters in the summer period

A. Jaanus · K. Toming · S. Hällfors · K. Kaljurand · I. Lips 157

Using bio-optics to investigate the extent of coastal waters: A Swedish case study

S. Kratzer · P. Tett 169

Identification of the coastal zone of the central and eastern Gulf of Finland by numerical modeling, measurements, and remote sensing of chlorophyll *a*

G. Lessin · V. Ossipova · I. Lips · U. Raudsepp 187

Interactions of light and organic matter under contrasting resource simulated environments: the importance of clonal traits in the seagrass *Zostera noltii*

I. Olivé · M.P. García-Sánchez · E.G. Brun · J.J. Vergara · J.L. Pérez-Lloréns 199

Long-term modeling of large-scale nutrient cycles in the entire Baltic Sea

O.P. Savchuk · F. Wulff 209

Response of eutrophication in the eastern Gulf of Finland to nutrient load reduction scenarios

O.P. Savchuk · T.R. Eremina · A.V. Isaev · I.A. Neelov 225

Modeling eutrophication and oligotrophication of shallow-water marine systems: the importance of sediments under stratified and well-mixed conditions

K. Soetaert · J.J. Middelburg 239

SHORT RESEARCH NOTE

Reference conditions for phytoplankton at Danish Water Framework Directive intercalibration sites

P. Henriksen 255

SUMMARY PAPER

Coastal eutrophication research: a new awareness

C.M. Duarte 263

Eutrophication in coastal marine ecosystems: towards better understanding and management strategies

J. H. Andersen · D. J. Conley

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 1–4.
DOI: 10.1007/s10750-009-9758-0 © Springer Science+Business Media B.V. 2009

The Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems took place 20–23 June 2006 in Nyborg, Denmark. The Symposium was attended by more than 200 persons with a specific interest in eutrophication processes as well as a common interest in science-based management and implementation of nutrient reduction strategies. More than 120 oral presentations were made, mostly focussing on both science and management of nutrient enrichment and eutrophication. The papers in this Special Issue of *Hydrobiologia* are all based on presentations made at the Symposium.

Electronic supplementary material The online version of this article (doi:10.1007/s10750-009-9758-0) contains supplementary material, which is available to authorized users.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

J. H. Andersen (✉)
DHI, Agern Allé 5, 2970 Hørsholm, Denmark
e-mail: jha@dhi.dk

D. J. Conley
Department of Geology, GeoBiosphere Science Centre,
Lund University, Sölvegatan 12, 223 62 Lund, Sweden

About the symposium

The symposium focused on the following four topics: (1) new and existing knowledge regarding coastal eutrophication, (2) specific eutrophication issues such as: (a) definition(s) and causes, (b) nutrient cycling and nutrient limitation, (c) reference conditions and (d) linkages to other pressures (climate change and top/down control), (3) summaries of existing knowledge in relation to monitoring and modelling coastal eutrophication and (4) adaptive environmental management strategies in relation to coastal eutrophication.

The symposium was jointly organised by the Danish Environmental Protection Agency (EPA), the Swedish EPA, Fyn County and DHI Water & Environment and received financial support from the organising institutions. In addition, the Symposium has been kindly sponsored by: (1) Baltic Sea 2020, (2) Danish Agriculture, (3) the International Agency for ¹⁴C Determination, (4) MARE—the Swedish Marine Eutrophication Research Programme and (5) the University of Southern Denmark. Further, the symposium received support from the European Commission's Joint Research Centre, Hotel Nyborg Strand and Scandinavian Airlines Systems (SAS).

The planning of the symposium was coordinated by an Organising Committee with the overall responsibility and a Scientific Committee which compiled a broad programme focussing on both science and management. A list of members of the

committees is available as supplementary online material.

Eutrophication research and management—the Danish connection

The symposium was a follow up to the highly successful 1993 Symposium *Nutrient Dynamics in Coastal and Estuarine Environments*, organised by the Danish EPA in collaboration with the European Commission, Directorate-General for Science, Research and Development. The Symposium Proceedings were published in the journal *Ophelia* with several seminal papers, for example, Duarte (1995), Nixon (1995) and Richardson & Heilmann (1995).

There was great regional and international interest for a follow-up symposium with a focus on both science and management. This interest in science and management has been stimulated by legislative settings, particularly the EU Water Framework Directive, in which coastal eutrophication problems are important issues in adaptive management plans (Anon., 2000).

During recent decades, Denmark and Sweden have been at the forefront of research on and management of eutrophication in coastal marine ecosystems (Jørgensen & Richardson, 1996; Christensen et al., 1998; Carstensen et al., 2006; Table 1), partly because the straits between Denmark and Sweden connecting the Baltic Sea to the North Sea are vulnerable to nutrient enrichment. Denmark and neighbouring countries have made substantial efforts to improve the marine environment through nutrient reductions both at the national level and through decades of regional cooperation regarding the Baltic Sea under the Helsinki Convention (www.helcom.fi) and the North Sea through the OSPAR Convention (www.ospar.org).

Both Denmark (Fig. 1) and Sweden have made large reductions in the discharge of nutrients. Billions of Euros have been spent, and they have not been spent in vain. The point source inputs of nutrients to the marine environment are significantly lower than they were 20 years ago. However, these reductions have not been sufficient to reduce the harmful effects of eutrophication and the targets for improved ecological status have not been reached.

Three Danish Action Plans for the Aquatic Environment over the past two decades (Conley et al., 2002) have resulted in significant reductions in the loss of nutrients to the environment (Conley et al., 2002; Carstensen et al., 2006). Point source inputs of phosphorus have decreased by more than 80%. Losses of nitrogen are expected to be reduced by ~50% when changes in agricultural practises that have already been implemented result in reduced loads to the marine environment. Figure 1 shows the temporal trends in total nitrogen loading to the Kattegat and Danish Straits over a 100-year period, with a peak in total nitrogen loading in the 1980s.

Since the late 1970s, loads originating from both point sources and diffuse sources have been declining. However, more than three decades since the first measures were implemented and more than a decade after the First International Danish Symposium on eutrophication, the problems associated with eutrophication are still far from being resolved. There has been a major development in scientific knowledge and in the conceptual understanding of nutrient enrichment and eutrophication in coastal waters. New questions and challenges have emerged, especially in relation to modelling and management of coastal eutrophication. In parallel, new legal and management settings have emerged or will emerge in the near future, for example, the EU Water Framework Directive and the process in relation to the implementation of the European Marine Strategy. Therefore, it was proposed and agreed in 2004 that a follow-up symposium focussing on both science and management of coastal eutrophication should be organised for June 2006.

About this Special Issue

The 21 papers in this Special Issue are a mixture of Research Papers, Opinion Papers and Short Notes, which reflect the broad range of presentations at the June 2006 symposium. Each manuscript was reviewed by at least two independent reviewers and by one of the guest editors. Copy editing was conducted by Janet F. Pawlak and Carolyn Symon. The Special Issue has received direct financial support from the Nordic Council of Ministers via the working group on the Sea and Atmosphere

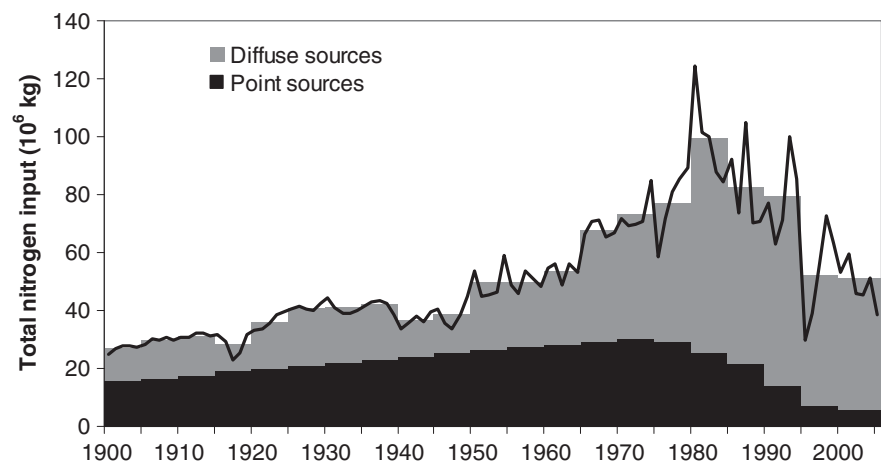
Table 1 Danish nutrient reduction targets sensu the Action Plans for the Aquatic Environment I, II and III. Baseline is 1987; reductions and targets were agreed by the Danish Parliament in 1987 and subsequently adjusted in 1990, 1999 and 2004

| Sector | Total nitrogen loads (tonnes) | | | | | | Total phosphorus loads (tonnes) | | | | | |
|-------------|-------------------------------|---|-----------|----|---|---------|---------------------------------|---|-----------|----|---|--------|
| | 1987 | ÷ | Reduction | % | = | Target | 1987 | ÷ | Reduction | % | = | Target |
| Agriculture | 311,000 | ÷ | 152,400 | 49 | = | 158,600 | 4.400 | ÷ | 4,000 | 91 | = | 400 |
| UWWTPs | 18,000 | ÷ | 11,400 | 63 | = | 6,600 | 4.470 | ÷ | 3,250 | 73 | = | 1,220 |
| Industries | 5,000 | ÷ | 3 | 60 | = | 2,000 | 1.250 | ÷ | 1,050 | 84 | = | 200 |
| Total | 334,000 | ÷ | 166,800 | 50 | = | 167,200 | 10.120 | ÷ | 8,300 | 82 | = | 1,820 |

See Carstensen et al. (2006) for details

UWWTPs: urban wastewater treatment plant effluents

Fig. 1 Trends in estimated total nitrogen inputs (*solid line*) from Denmark to the Danish Straits including the Kattegat since 1900, with 5-year averages of point and diffuse sources. Used with the kind permission of Jacob Carstensen, NERI; based on Conley et al. (2007)



(Hav- og Luftgruppen) as well as indirect financial support from the Danish EPA, Swedish EPA and DHI.

This Special Issue as well as others (Kononen & Bonsdorf, 2001; Rabalais & Nixon, 2002; Bachmann et al., 2006) demonstrate that considerable knowledge has been generated since the First Danish Symposium in 1993. We, as guest editors, are pleased with the Special Issue as compiled and hope that the readers will share this opinion.

Despite the vast knowledge and common understanding of eutrophication, some important gaps still remain, especially with regard to regime shifts, thresholds and multiple stressors. In addition, climate change needs to be taken into account. A fundamental problem that needs to be addressed is the lack of political will to implement adequate nutrient management strategies. A broader acceptance of the need

to use the best scientific information we have (whilst still seeking to improve knowledge = “moving whilst improving”) rather than wait for ‘perfection’ is recommended. Finally, it should be kept in mind that we do not manage eutrophication as such, we manage humans with the aim of reducing the effects of eutrophication.

Acknowledgements Thanks are expressed to the Nordic Council of Ministers and the organisers and sponsors of the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems. The Preface improved as a result of comments from Jacob Carstensen and Scott W. Nixon. Special thanks go to Sif Johansson, Jørgen Dan Petersen, Jens Brøgger Jensen, Torkil Jønch Clausen, Henning Karup, Jørgen Magner and Morten Søndergaard. We are indebted to the reviewers and to Janet F. Pawlak and Carolyn Symon; this Special Issue would not have been possible but for their helping hands.



norden

Nordic Council of Ministers

Open Access This article is distributed under the terms of the Creative Commons Attribution Noncommercial License which permits any noncommercial use, distribution, and reproduction in any medium, provided the original author(s) and source are credited.

References

- Anon., 2000. Directive 2000/60/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.
- Bachmann, R. W., J. E. Cloern, R. E. Hecky & D. W. Schindler (eds), 2006. Eutrophication in freshwater and marine systems. *Limnology & Oceanography* 51: 351–800 (Special Issue).
- Carstensen, J., D. J. Conley, J. H. Andersen & G. Ærtebjerg, 2006. Coastal eutrophication and trend reversal: a Danish case study. *Limnology & Oceanography* 51: 398–408.
- Christensen, P. B., F. Møhlenberg, L. C. Lund-Hansen, J. Borum, C. Christiansen, S. E. Larsen, M. E. Hansen, J. H. Andersen & J. Kirkegaard, 1998. The Danish marine environment: has action improved its state? *Havforskning fra Miljøstyrelsen*, Nr. 62. 115 pp.
- Conley, D. J., S. Markager, J. Andersen, T. Ellermann & L. M. Svendsen, 2002. Coastal eutrophication and the Danish National Aquatic Monitoring and Assessment Program. *Estuaries* 25: 706–719.
- Conley, D. J., J. Carstensen, G. Ærtebjerg, P. B. Christensen, T. Dalsgaard, J. L. S. Hansen & A. Josefson, 2007. Long-term changes and impacts of hypoxia in Danish coastal waters. *Ecological Applications* 17: 165–184.
- Duarte, C. M., 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41: 87–112.
- Jørgensen, B. B. & K. Richardson (eds), 1996. Eutrophication in Coastal Marine Ecosystems. *Coastal and Estuarine Studies*, 52. American Geophysical Union, Washington, DC: 273 pp.
- Kononen, K. & E. Bonsdorf (eds), 2001. Man and the Baltic Sea. *Ambio* 30: 171–326 (Special Issue).
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Rabalais, N. N. & S. W. Nixon (eds), 2002. Nutrient over-enrichment in coastal waters: global patterns of cause and effect. *Estuaries* 25: 639–900 (Dedicated Issue).
- Richardson, K. & J. P. Heilmann, 1995. Primary production in the Kattegat: past and present. *Ophelia* 41: 317–328.

Eutrophication and the macroscope

Scott W. Nixon

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 5–19.
DOI: 10.1007/s10750-009-9759-z © Springer Science+Business Media B.V. 2009

Abstract It is important to view eutrophication as an increase in the supply of organic matter to an ecosystem rather than as a simple problem of nutrient pollution. This emphasizes that eutrophication is a fundamental change in the energetic base that may propagate through the system in various ways and produce a variety of changes. Some of these changes may be desirable (e.g., increased secondary production) and some may not (e.g., hypoxia). Defining eutrophication in terms of changing nutrient concentrations or chlorophyll levels or species composition confuses symptoms with the underlying phenomenon. While nutrient enrichment is the most common cause of eutrophication, it is not the only one. As recent and ongoing nutrient reductions make an impact in the coastal waters of the wealthier nations, we will see an increasing number of systems in which primary production is decreasing. This reduction in the supply of organic matter is here defined as oligotrophication, a phenomenon now well documented

in lakes. So far, there has been little appreciation of this limnological study by coastal marine ecologists or managers, but there is much we can learn from it. The great ecologist H.T. Odum long argued that we need ‘macrosopes’ to help ecologists see the problems they study as they are embedded in the larger scales of nature and society. Marine eutrophication (and oligotrophication) is a perfect example of a problem that must be studied with a view toward the larger scales as well as toward the microscopic details. While much of the hardware (e.g., satellite imagery) for the mythical macroscope has been developed in the last 30 years, many ecologists and managers still look at eutrophication as a local problem linked to local sources of nutrient enrichment. Such a parochial view isolates eutrophication from its long intellectual history—a history that is linked to the development of our understanding of production in coastal waters. It also neglects the intellectual richness and complexity of eutrophication. One example of the importance of the macroscopic view is the emerging importance of climate-induced changes in phenology and the consequences of changing phenology on productivity. These changes may lead to eutrophication or oligotrophication. Climate changes may also exacerbate or alleviate conditions such as hypoxia that are associated with eutrophication. Seeing eutrophication in the macroscopic view is important for understanding and managing the phenomenon.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from
the Second International Symposium on Research and
Management of Eutrophication in Coastal Ecosystems, 20–23
June 2006, Nyborg, Denmark

S. W. Nixon (✉)
Graduate School of Oceanography, University of Rhode
Island, Narragansett, RI 02882, USA
e-mail: swn@gso.uri.edu

Keywords Eutrophication · Oligotrophication · Productivity · Nutrient enrichment · Trend reversal

Eutrophication

At the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments held in Denmark in 1993, I offered a definition for the term 'eutrophication' as a noun meaning "an increase in the rate of supply of organic matter to an ecosystem" (Nixon, 1995). In addition to simplicity and brevity, the main arguments in favor of the definition are that it focuses on eutrophication as a change in the rate of supply of carbon and energy to an ecosystem, and it thus differentiates the phenomenon of eutrophication from its various causes (e.g., nutrient enrichment, reductions in grazing pressure, increases in water residence time) and from its various consequences (e.g., hypoxia, fish kills, turbidity). Since the increase in the supply of organic matter can be due to allochthonous or autochthonous sources, this definition also links the large body of study dealing with the responses of coastal waters and sediments to organic loading from sewage and manufacturing wastes with more recent studies that focus on the impacts of increased primary production within the system. For example, the well-known Pearson–Rosenberg (1978) model of macrobenthic succession in response to organic loading has proven useful in understanding benthic responses to inorganic nutrient enrichment (Heip, 1995).

While the definition I proposed seems to have proven useful (it is recently cited 50–70 times each year according to Science Citation Index), it is not without its critics. Three points raised at the symposium that produced the collection of articles in this issue of *Hydrobiologia* are important and deserve comment. First, K. Sand-Jensen, who acted as respondent to the talk on which this article is based, noted that total system production may not increase with nutrient loading, but the types and relative abundance of the primary producers may change as rooted macrophytes are replaced by macroalgae or phytoplankton (e.g., Sand-Jensen & Borum, 1991). The results of our experimental studies with shallow (1 m deep) coastal lagoon mesocosms containing

eelgrass, macroalgae, epiphytes, and phytoplankton agree with his findings (Nixon et al., 2001), but nutrient addition experiments with the deeper (5 m deep) phytoplankton-based MERL mesocosms show total system production increasing over a very wide range in nutrient loading (e.g., Nixon et al., 1986, 2001). At least for shallow systems, however, Sand-Jensen's point is well taken. Nonetheless, one definition cannot do everything—species substitution may be one of the consequences of nutrient enrichment but it may not be a cause or a direct consequence of eutrophication. In shallow macrophyte-dominated systems, species substitution can be a conspicuous change, while in phytoplankton-dominated systems the shifts in species with nutrient enrichment may be equally important but less easily observed and documented.

Second, in discussion following the talk, R. Elmgren made the good point that the impact of adding 100 g of carbon as cellulose to a bay is quite different from adding an amount of nitrogen (N) or phosphorus (P) that will stimulate the fixation of 100 g of carbon by the phytoplankton. He was right, of course, since the N and P may be recycled many times and stimulate the fixation of much more carbon. However, this seems to me to be a distinction that lies outside the definition. The addition of organic matter may produce some responses that are similar to those resulting from stimulating in situ primary production (e.g., hypoxia in bottom water), but not others (e.g., species shifts in primary producers). My definition does not assume a stoichiometric equivalence between nutrient addition and carbon addition.

Third, in preparation for the talk, it was brought to my attention that some managers object to the definition because it requires expensive monitoring of primary production to document that eutrophication is occurring or being remediated. This is an important perspective that I had not appreciated, but I do not think it is a good reason to reject the fundamentals of the definition. As a practical matter, the research community can also offer some solutions that may be useful. For example, we have known for about 50 years that primary production can, in some cases, be computed from measurements of phytoplankton biomass and light in the water column (Ryther & Yentsch, 1957), and a recent review by Brush et al. (2002) of many more studies in the United States and Europe suggests that this approach

applies across a wide variety of coastal systems. Since it is relatively inexpensive to monitor chlorophyll, vertical light attenuation coefficients, and incident radiation, perhaps in association with periodic calibrations using local ^{14}C uptake measurements, it seems that the issue of costly monitoring can be averted in phytoplankton-based systems. Macrophyte-dominated systems are another matter, but in such shallow waters data on area coverage by macroalgae and/or sea grasses may be a useful proxy for production (e.g., Sfriso et al., 1989). The increasing availability of reliable in situ oxygen monitoring equipment that can be deployed for days or weeks is also making it possible to conduct relatively inexpensive free water measurements of total system metabolism (e.g., Bergondo et al., 2005), a technique pioneered by H.T. Odum half-a-century ago to give a more macroscopic view of production than bottle incubations (e.g., Odum & Hoskin, 1958).

Andersen et al. (2006) have just reviewed some of the more common definitions used for coastal eutrophication and noted that the management community within Europe has chosen to define the phenomenon in terms of nutrient pollution and 'undesirable' changes in the biology or ecological status of an area. I suppose this is understandable from a regulator's perspective, but as they point out, it embodies an anthropocentric view of what is 'desirable' in nature and raises issues of 'reference conditions' against which change can be measured. For me such a definition is too limited and makes eutrophication a pollution problem arising from one type of pollutant (nutrients, or just nitrate in the case of the EC Nitrates Directive) rather than embedding eutrophication in the more basic ecological issue of changing the energetic base of ecosystems. One definition missed by Andersen et al. (2006) is that offered by Ulanowicz (1986, p. 80) in which eutrophication was defined as "any increase in system ascendancy due to a rise in total system throughput that more than compensates for a concomitant fall in the mutual information of the flow network." While this is certainly too much jargon to appeal to regulators, it does seem consistent with my emphasis on eutrophication increasing the supply of organic matter. Andersen et al. (2006) also argue strongly for including the measurement of primary production in European eutrophication monitoring programs. If their advice is heeded, it will increase the common ground between the research

and management communities with regard to understanding and dealing with eutrophication.

The eutrophication generation

It is customary for the president of the Coastal and Estuarine Research Federation to give a farewell address at end of his or her term of office. Linda Schaffner ended her presidency in 2005 with a talk at the biennial meeting in Norfolk, Virginia in which she exclaimed: "This is the eutrophication generation!" If we accept a generation time of 20 years (an arbitrary demographic standard), then there were probably three generations in her audience, but for those of us who began our scientific careers in the late 1960s and early 1970s, she was certainly right. A quick search of my office produced a pile of books and special journal issues on (largely) marine eutrophication about 0.75 m high. The earliest appeared in 1969 (Eutrophication: Causes, Consequences, Correctives published by the U.S. National Academy of Sciences) and the most recent are the excellent special issues of *Limnology and Oceanography* that appeared in January 2006, and *Ecological Applications* that appeared in July 2007. The first was almost completely devoted to fresh waters, while marine and freshwater systems are about equally represented in the most recent *Limnology and Oceanography*. The *Ecological Applications* special issue is completely devoted to coastal marine systems. An exhaustive history of coastal marine eutrophication and the evolving interplay between science and management related to the issue in Europe has recently been published by de Jong (2006).

A search of the Aquatic Sciences and Fisheries Abstracts (ASFA) database confirms that about half of the articles published in recent years on the topic of eutrophication appear to focus on marine systems (Fig. 1). It is tempting to interpret the rapid increase in eutrophication citations between the early 1970s and the early 1990s as the research community responding to increasing impacts of nutrient enrichment from increasing fertilizer use, fossil fuel combustion, and protein consumption (e.g., Nixon, 1995; Rabalais & Nixon, 2002; de Jong, 2006; Howarth & Marino, 2006), but I suspect that this is only part of the story. The increasing study of eutrophication is part of a larger trend that has been driven by at least three

Fig. 1 World consumption of synthetic fertilizer since 1970 (total N, P₂O₅, K₂O International Fertilizer Industry Association, http://www.fertilizer.org/ifa/statistics/indicators/ind_cn_world.asp) and the number of publications dealing with the subject of eutrophication (solid circles) and marine eutrophication (*open circles*) as reported on line by Aquatic Sciences and Fisheries Abstracts (ASFA)

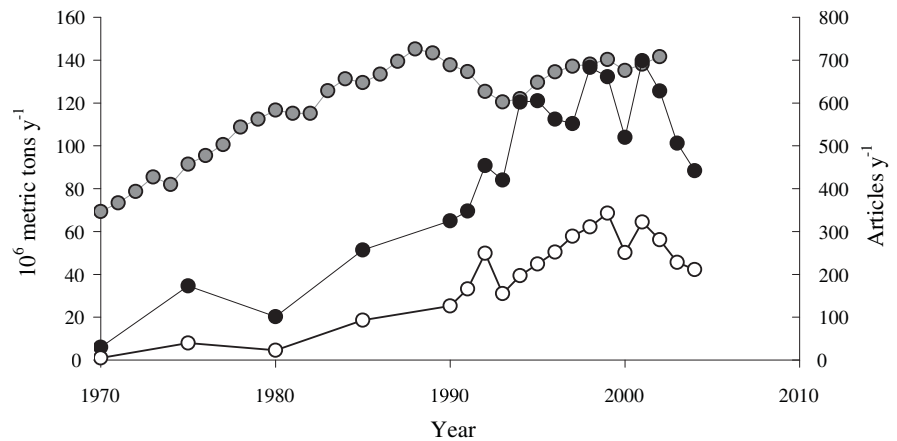
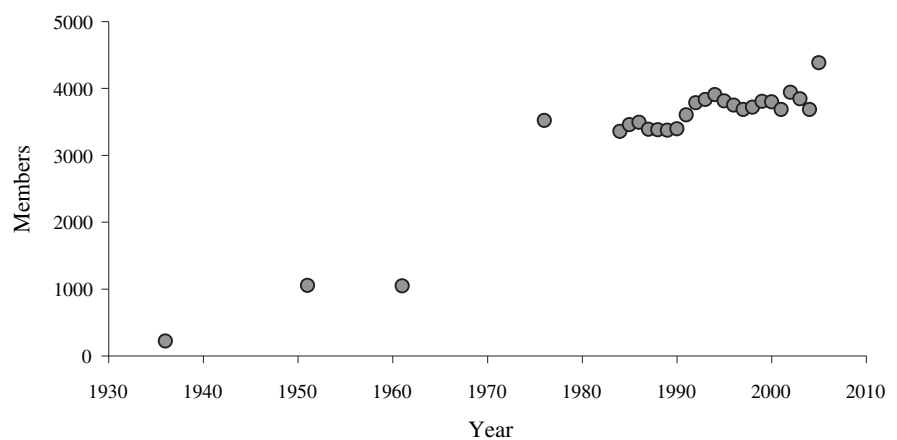


Fig. 2 Membership in the American Society of Limnology and Oceanography (ASLO) over time (data from Susan Jones, ASLO Business Office, e-mail from Sybil Seitzinger 5/30/06)



factors: the production of large numbers of ‘baby boomer’ Ph.D. research scientists after about 1975, the rise of the environmental movement in the 1970s, and the availability of increasing amounts of government funding for scientific research after the Second World War. The impact of these factors can be seen in the time series of membership in The American Society of Limnology and Oceanography (ASLO) (Fig. 2). It may also be argued that a sharply expanding economy in Europe, North America, and parts of Asia also provided the means by which societies were willing and able to invest in environmental research and the specialized education it requires. A good integrated measure of the expanding economy that does not require numerous inflation corrections is the world inventory of annual carbon dioxide (CO₂) emissions that documents the ‘Great Acceleration’ after about 1950 (Steffen et al., 2007) (Fig. 3). In macroscopic view, the wealth created by industrial expansion also made it possible to train and support the scientists who identified and documented

many of the environmental problems caused by that ‘Acceleration’. The link between the carbon dioxide problem and marine eutrophication goes well beyond the fixation and release of nitrogen during fossil fuel combustion and the use of fossil fuels in fertilizer production. The growing number of scientific articles dealing with coastal eutrophication is mirrored on a much larger scale by the growing number of articles being added each year to the ASFA database (Fig. 4), by the growing scale of human impacts on the environment, and by resource consumption on an unprecedented scale (Steffen et al., 2007).

There are, of course, fads in scientific research (Rigler & Peters, 1995), but the scientific community can also be surprisingly steadfast in its attention to some issues regardless of how their importance may change. For example, the ASFA database shows that the number of publications dealing with oil pollution has remained relatively constant at about 150–200 per year since the mid-1970s, with the exception of a brief peak of over 400 around 1980 (almost certainly

Fig. 3 Global emissions of carbon as carbon dioxide (data provided by M.E.Q. Pilson, Graduate School of Oceanography, Univ. of Rhode Island, from <http://cdiac.esd.ornl.gov/ftp/ndp030/global.17512003.ems> up to 2003. Emissions for 2004 and 2005 calculated from data published by British Petroleum in June 2006). Arrow marks the publication of Revelle and Suess's (1957) famous paper on the coming CO₂ problem

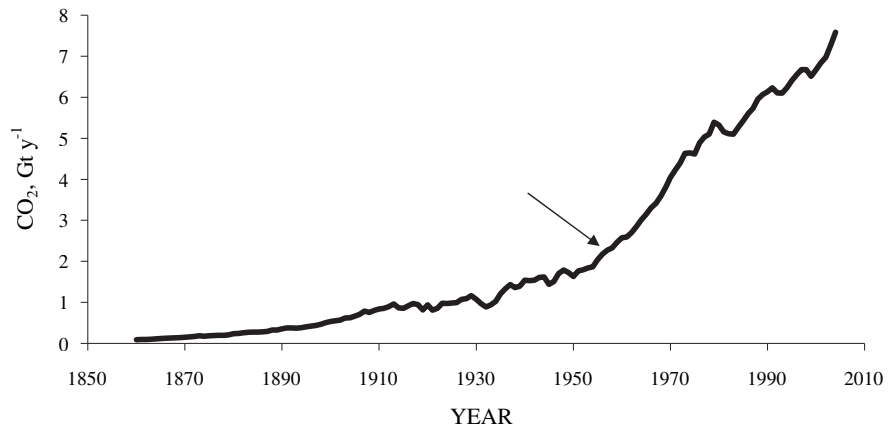


Fig. 4 The number of publications added to the Aquatic Sciences and Fisheries (ASFA) database during various years (data from Vicki Soto, Supervisor of Aquatic and Meteorological Sciences, ASFA; e-mail 5/22/06)

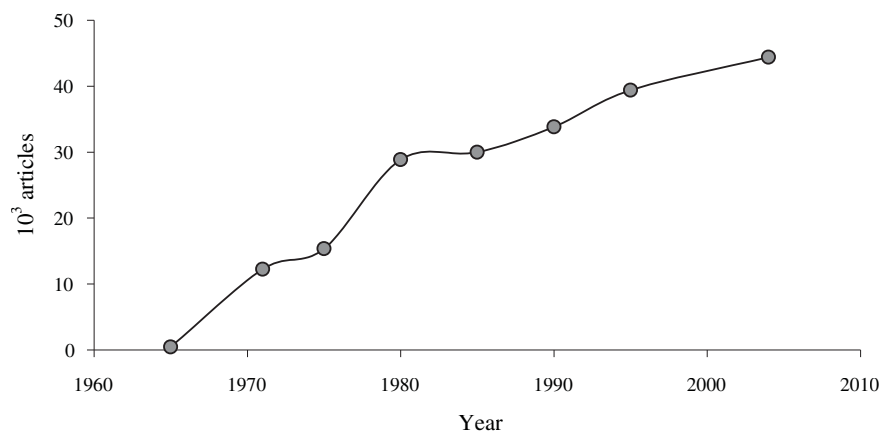
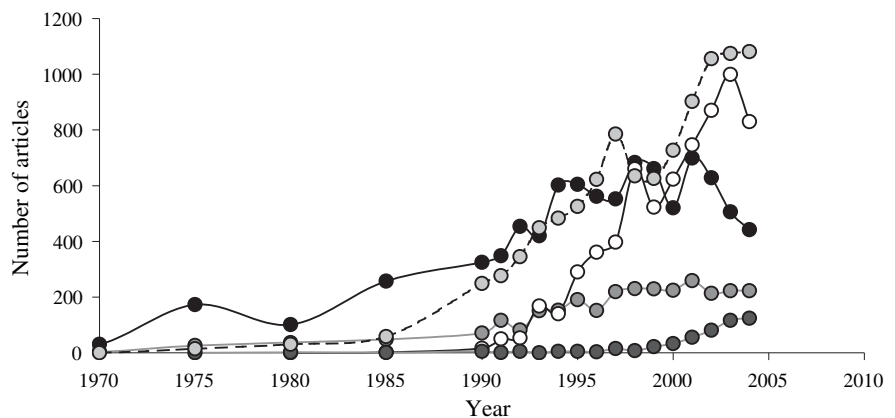


Fig. 5 The number of publications on various topics relating to aquatic ecology indexed by ASFA by year of publication. Arranged from highest to lowest the topics are: climate change, biodiversity, eutrophication, overfishing, invasive species



due to the *Amoco Cadiz* disaster on the coast of Brittany in March, 1978). The attention paid to oil pollution is steadily increasing despite the declines in the volume of oil spilled in U.S. coastal waters (and presumably also in Europe) as a result of better management, education, and engineering (Valiela, 2006). Data compiled by the U.S. Coast Guard show

oil spills averaging about 15 million gallons per year in the 1970s have declined to 1 or 2 million gallons per year around 2000. While eutrophication has received considerably more attention than overfishing for many years, eutrophication has recently been overtaken by biodiversity as a favored topic for study (Fig. 5). Publications in which climate change is a

key phrase have increased from fewer than 50 in 1980 to almost 5,000 in 2006. For the future, however, the expanding coastal populations of the developing world, an increasing world demand for meat protein, and the increasing production of biofuels assure that marine eutrophication will continue to demand the attention of marine ecologists and managers (Nixon, 1995; Valiela, 2006).

Coastal marine oligotrophication?

Despite the probable spread of marine eutrophication, especially in developing (often tropical) countries, there are beginning to be cases in which improved fertilizer and livestock management and advanced wastewater treatment are significantly reducing the flow of nutrients to coastal systems (e.g., Yamamoto, 2003; Carstensen et al., 2006; Greening & Janicki, 2006; Soetaert et al., 2006; Philippart et al., 2007). The decline in nutrient inputs to increasing numbers of temperate coastal systems in wealthier countries means that we will almost certainly begin to see an increasing number of articles dealing with marine oligotrophication or 'trend reversal' as it is called in Europe (e.g., Carstensen et al., 2006; Philippart et al., 2007). This is an almost unexplored phenomenon in marine systems. Only some 5–20 articles on this topic have appeared in the ASFA database each year since 1990, and almost all of these deal with the oligotrophication of lakes and reservoirs (e.g., Ney, 1996; Stockner et al., 2000; Anderson et al., 2005). The observations of lake oligotrophication suggest that we should anticipate important changes in both ecosystem structure and function. For example, an analysis of 35 case studies by Jeppesen et al. (2005) found declines in phytoplankton biomass and changes in taxonomic composition, increases in the ratio of zooplankton biomass to phytoplankton biomass, declines in total fish biomass, and increases in the relative abundance of piscivores. The small sample of marine systems suggests that their responses will also involve complex changes in structure in virtually all trophic levels and declining yields at least for some important species (e.g., Yamamoto et al., 2003 for the Seto Inland Sea and Philippart et al., 2007 for the Wadden Sea). In anticipation of a growing interest in the topic, I would like to offer a definition for oligotrophication that parallels that of eutrophication:

Oligotrophication (noun) – a decrease in the rate of supply of organic matter to an ecosystem.

This definition shares all of the strengths (and limitations) of the earlier eutrophication definition, and the same would preclude the use of the awkward term 're-oligotrophication' that has started to appear in the limnological literature (e.g., Jeppesen et al., 2002). By this definition, it is also clear that the coming oligotrophication due to nutrient reductions is the second oligotrophication of coastal marine systems. The first was a reduction in organic inputs from sewage that accompanied the move to full secondary treatment of urban and industrial wastes in wealthier countries during the 1970s and 1980s (National Research Council, 1993). The impact of very large expenditures on sewage infrastructure on the amount of organic matter actually reaching coastal systems is very difficult to determine. Significant improvements in sewage treatment technology were accompanied by large increases in the populations connected to sewage systems. In the United States, the overall balance may have been a net decrease in organic matter discharged by treatment plants of about 25% (Stoddard et al., 2002). Unfortunately, the research community and the resource base were much smaller during that first major change, and the ecological impacts (aside from increases in dissolved oxygen) were seldom documented.

The intersection of two great inquiries in marine ecology

Eutrophication (and oligotrophication) will continue to be a major focus of our research not just because nutrient fluxes between land and the coastal ocean will be changing (both increasing and decreasing), but also because eutrophication lies at the intersection of two of the great inquiries of marine ecology. The first of these inquiries, the basis of marine production, developed largely in Europe, within a few hundred kilometers of the meeting that stimulated this collection of articles. The second, the impact of human activities on the sea, began for eutrophication and nutrient pollution in a coastal lagoon on Long Island, New York.

It seems appropriate and useful for an introductory article to spend a little time on the development of

these two lines of inquiry, and my task is made much easier with regard to the first because of an excellent history of our understanding of primary production in marine systems written by Mills (1989).

The agricultural model

Very briefly, the development of marine ecology as a quantitative science can be traced to the establishment of the Kiel Commission for the Study of the German Seas in 1870. The major task of the Commission was to learn why some regions of the sea produced so many more fish than others. Beginning in the early 1880s, Victor Hensen at Kiel emphasized quantitative sampling of the plankton and benthos and focused on the metabolism of the sea as well as measurements of standing crops. By 1887, Karl Brandt had come to Kiel and begun the development of chemical analyses of plankton and sea water. He first identified nitrogen as the probable limiting factor for production in 1899. As Mills (1989, p. 53) noted, "The history of plankton dynamics after 1899 is largely the history of the knowledge of the nitrogen cycle." By 1901, the Norwegian botanist H. H. Gran showed that denitrification was widespread in the Baltic and the coastal North Sea. The Kiel Commission was a forerunner of the modern International Council for the Exploration of the Sea (ICES), which first met in Copenhagen in 1902. The ICES began the first regular monitoring of the coastal environment in the Baltic and North Sea and supported the efforts of Emil Rabin at Kiel to improve nutrient analyses between 1902 and 1917.

In 1893, the 'Copenhagen Program' began with the goal of relating climate to fisheries yields, which stimulated much of the early study in Scandinavia and Germany on physical oceanography and the links between physical and biological processes. Improved understanding of physical mixing led Alexander Nathansohn at Kiel in 1906 to identify its importance in supplying nutrients to surface waters and stimulating productivity. The first actual measurements of plankton production in a marine system were made in the Oslo Fjord by Gaarder and Gran in 1916 using the 'light and dark bottle' oxygen technique. Similarly, the importance of eelgrass and benthic communities in shallow-water ecosystems was quantified for the first time by Peterson and others working at the Danish Biological Station between 1915 and 1920. Analytical advances during the 1920s by Atkinson

and Harvey at the Plymouth Laboratory in England made it possible to quantify the importance of seasonal cycles in the abundance of the major nutrients and provided some of the foundation on which Gran during 1931–1935 formalized the concept of the compensation depth, the basis of our modern understanding of the spring bloom. Finally, during 1954–1960, Steemann-Nielsen obtained the first measurements of primary production by the phytoplankton over an annual cycle using the ^{14}C technique (Richardson & Heilmann, 1995). His stations were in the Kattegat, just offshore from the site of the meeting that stimulated this collection of articles. There is certainly no more appropriate venue for discussion of the important links between nutrients and the supply of organic matter to marine ecosystems. However, it is important to remember that these links were virtually always viewed in a positive light—more nutrients, more primary production, more fish—what Cushing (1975) called the 'agricultural model' of production in the sea (Nixon et al., 1986; Nixon, 1995; Nixon & Buckley, 2002).

Of ducks and dead zones—why didn't we see it coming?

While the positive link between nutrients and production was developed in Scandinavia and northern Europe, the view of nutrients as a marine pollutant first emerged in the United States. As far as I am aware, the first reference to nutrient over-enrichment came from John Ryther's (1954) identification of duck waste being responsible for the development of nuisance phytoplankton blooms ($>10^{10}$ cells l^{-1}) in Moriches Bay and Great South Bay on Long Island, NY. At that time, it was believed that ducks needed to be raised on water, and data on the duck food supplied to produce some four million ducks each year (Ryther, 1989) suggest that the N and P loading to the bays may have been about 8.8 and 1.2 $\text{mmol m}^{-2} \text{d}^{-1}$, respectively, a very intensive fertilization. The dense phytoplankton blooms were believed to have a negative impact on tourism and were also blamed for the loss of a productive oyster fishery, "...though a definitive cause-and-effect relationship was never established" (Ryther, 1989).

The impact of Ryther's 1954 article was neither immediate nor great, and when the American Association for the Advancement of Science published the

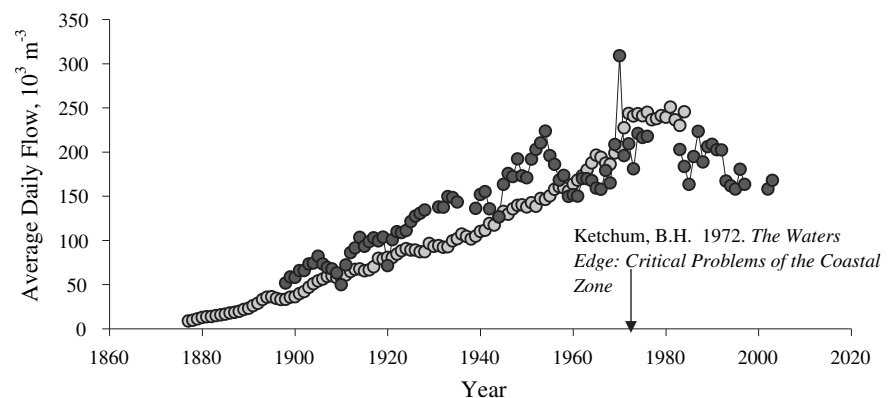
first monograph devoted to *Estuaries* (Lauff, 1967), only one chapter out of 71 dealt with nutrients as a pollutant, and that was written by B. H. Ketchum, Ryther's colleague at the Woods Hole Oceanographic Institution. Even 5 years later, when Ketchum (1972) edited the landmark volume, *The Water's Edge: Critical Problems of the Coastal Zone*, only three pages out of 393 were devoted to nutrients as a pollutant. Things were no different in Europe, where Wulff (1990) noted that "The concept of 'marine eutrophication' was unheard of until about 20 years ago." A more detailed discussion of the history of the pollution side of marine eutrophication, especially as it developed in Europe, is given by de Jong (2006).

The question naturally arises as to why it took so long for the potential negative impacts of nutrient enrichment to be widely recognized by the coastal research and management communities. As early as 1957, Revelle and Suess (Fig. 3) eloquently pointed to the coming CO₂ problem when they wrote, "...human beings are now carrying out a large scale geophysical experiment of a kind that could not have happened in the past nor be reproduced in the future. Within a few centuries we are returning to the atmosphere and the oceans the concentrated organic carbon stored in sedimentary rocks over hundreds of millions of years." With some modification, a similar statement could have been made about the human perturbation of the N and P cycles. While it is true that the anthropogenic fixation of atmospheric N for fertilizer began to increase dramatically only after 1950, the growing need for fertilizer to support the human population had been clear for decades (Smil, 2001). Moreover, large amounts of N and P from human waste began to be released from urban sewer systems during the last quarter of the 1800s (Fig. 6),

and the impacts on coastal receiving waters must have been dramatic. There were also ambitious monitoring programs in many rivers that included ammonium and nitrate as well as organic N (actually only the more easily oxidized components of organic N). These measurements were made in the belief that the various forms of N were good chemical indicators of bacterial contamination from sewage and could thus be used to separate contaminated water from sources suitable for drinking (Hamlin, 1990). While N concentration proved to be an imperfect way to judge drinking water, the early measurements often showed very large increases in N being carried into estuaries as cities grew along the rivers and the public demanded running water, flush toilets, and sewer systems to collect and dispose of the waste (Nixon et al., 2008). In spite of what must have been intense fertilization of many urban coastal areas during the first quarter of the twentieth century, virtually all of the early concerns of sewage impact focused on human disease, odors, and discoloration.

I think there are at least five reasons for our late awakening to the potential problems of nutrient enrichment. First, eutrophication is the most subtle of a suite of problems associated with the discharge of raw or partially treated sewage—it is completely understandable that the first concerns were contamination of seafood and the hydrogen sulfide odors associated with low oxygen. Second, while the role of nutrients in marine production was well established by the turn of the twentieth century, the entire focus was on the positive effects of nutrient stimulation. Third, the number of marine ecologists was very small and specialized, and many were focused on taxonomy and systematics. Pollution issues in urban estuaries were usually the province of city engineers,

Fig. 6 The average daily flows of water and sewage in the city of Providence, Rhode Island as reported in the Annual Reports of the City Engineer (Nixon et al., 2008)



public health scientists, or sanitarians. It is difficult for us to appreciate how small the marine research community was and how limited its resources were. The research groups at Kiel and, later, at Plymouth were very unusual in their interdisciplinary focus on marine production. But even these made progress slowly. Consider that over the 40 years between 1887 and 1927, when it was one of the most active marine research groups in the world, the 'Kiel school' averaged just 2.6 publications each year (Mills, 1989). Fourth, there was a general impression that the water residence times of most estuaries were too short for nutrient impacts to be as severe as they were in lakes (e.g., Schindler, 1981). Fifth, unlike the relatively well-mixed atmosphere, estuaries are isolated from each other and their pollution problems are often discovered and studied with a very local perspective. It was difficult to untangle the impact of nutrients in urban estuaries from the impacts of organic loading (also eutrophication by my definition), overfishing, dredging, filling, and various other human impacts. Nutrient-driven coastal eutrophication only emerged as an internationally recognized pollution problem when non-point sources from fertilizers and N-enriched atmospheric deposition became widespread and caused unambiguous impacts in many more coastal areas that were not urban: And that is when there were growing numbers of environmental scientists and environmental advocates watching the coast.

From a belated start, there has been much progress made during the last 35 years in understanding the causes and consequences of marine eutrophication, and there is now a widespread recognition that the phenomenon can have severe, undesirable consequences (Colombo et al., 1992; Vollenweider et al., 1992; Nixon, 1995; Cloern, 2001; Rabalais, 2002; Howarth & Marino, 2006; Schindler, 2006; Smith et al., 2006; Valiela, 2006; and many others). The major sources of N and P reaching coastal waters have been identified and, in many cases, quantified. The pathways by which N and P enter coastal waters are also well known, though some have proven difficult to quantify (e.g., groundwater). The ecological responses are still being documented—some are relatively well understood (e.g., hypoxia) while others remain speculative (e.g., links to disease). It is also safe to say that almost all of the research attention has been directed to nutrient-driven eutrophication. As a

result, we know relatively little about system responses to eutrophication that may be the result of physical changes (e.g., increases in water residence time or turbidity), climate change (e.g., changes in freshwater inflow), or 'top-down' effects (e.g., removal of filter feeders). An excellent example of how such changes can cause eutrophication in spite of decreasing nutrient enrichment from wastewater is described by Cloern et al. (2007) for San Francisco Bay, CA. We also know almost nothing about the ecological consequences of oligotrophication in coastal marine systems, and there appears to have been little discussion of what we might learn of this phenomenon from limnology. Coastal marine ecosystems differ from lakes and reservoirs in some fundamental and important ways (Nixon, 1988), but just as we learned much from the earlier manifestation of eutrophication in lakes, we can and should learn much from their first experiences with oligotrophication.

The macroscope

The great ecologist H.T. Odum used to joke (but in all seriousness) in the 1970s that ecologists needed to invent a 'macroscope' that would help them see how their studies fit into the larger scales of nature (Nixon, 1996). As Odum wrote in *The System in the Sea* (Platt, 1993), "Always select the scale one size larger than your problem, because it is half driven from the larger side. That's the first principle of the systems approach." Much of the hardware (and software) for the mythical macroscope has now been invented, including satellites, the internet, search engines, high-speed computers, underway sensors, and many other additions to the ecologist's tool box. Environmental scientists are also learning to work together on cross-disciplinary problems, on larger systems, and in bigger teams. For example, the frequency of single-authored articles in the journal *Ecology* has declined from over 90% in 1920 to about 25% today (Paine, 2005). Only 15% and 12% of the articles in *Estuaries* and *Continental Shelf Research*, respectively, were by single authors up to the early 1990s (Nixon, 1996).

Coastal marine eutrophication is the quintessential problem requiring macrosopes as well as microscopes. Moreover, its study has done a lot to engage coastal ecologists with the macroscopic approach. As defined here, eutrophication is an intellectually rich

problem that weaves together plant, animal, and microbial physiology, physics, climatology, hydrology, biogeochemistry, soil science, agriculture, forestry, urban infrastructure, demography, and nutrition. It involves every level of the ecosystem from abiotic factors to top carnivores. It draws on our skills of observation across wide scales of time and space. The eutrophication literature is full of the results of studies using microscopes, satellite images, sediment cores, stable and radioactive isotopes, analyses of shells and scales and bones, field surveys, buoy sensor data records, mesocosm experiments, long time-series analyses, historical documents, field manipulations, physiological rate measurements, plant and animal tissue analyses, growth studies, and complex numerical models of atmospheric chemistry, oceanic circulation, and ecosystems.

Part of the intellectual richness of the eutrophication (and oligotrophication) phenomenon as defined here springs from its great complexity. Because increases in the supply of organic matter are so often associated with nutrient fertilization, we are faced with the many sources of nutrients: both natural and anthropogenic, fixed and mobile, point and non-point. And with many forms—with N alone we must contend with ammonium, nitrite, nitrate, dissolved organic N (a complex mix itself), dinitrogen gas, nitrogen oxides, ions, solids, gases, vapors, aerosols, particles—all moving through many pathways including groundwater, pipes, streams, sheet flows, tidal and sub-tidal flows, dust, migrating animals, detritus, and the atmosphere. However, the macroscopic view demands that we also look beyond nutrients and keep our eyes and minds open to the possibilities of other factors (such as climate change) that may alter the supply of organic matter and energy in marine ecosystems (e.g., Schell, 2000; Schulman, 2005; Grebmeier et al., 2006; Cloern et al., 2007; Fulweiler et al., 2007). These factors may themselves change the supply of nutrients or interact in important ways with changing nutrient inputs to modify the outcome of nutrient enrichment or removal. Eutrophication is a lot more interesting than ‘nutrients in, dead fish out’.

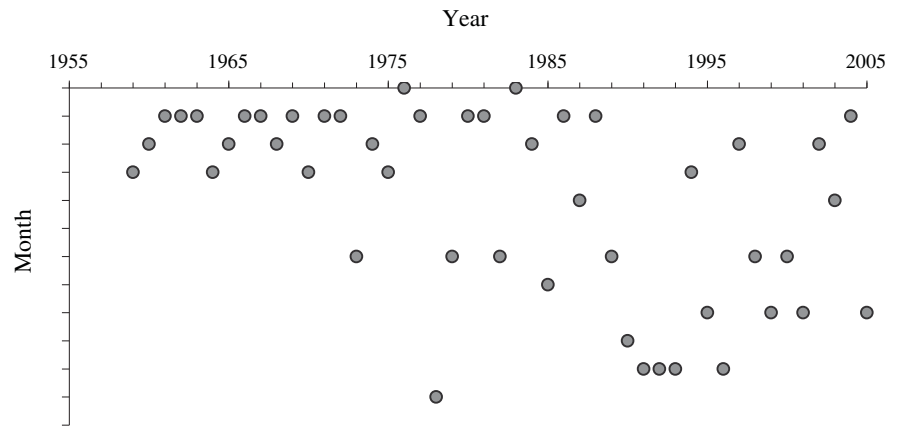
Phenology and eutrophication

In proposing a definition of eutrophication, I gave some examples of changes other than increasing nutrient inputs that might lead to eutrophication, including the

effect of dams in reducing river sediment loads and increasing downstream estuarine water clarity, and the closing of passes through barrier spits that might increase water residence time in coastal lagoons, and the over-harvesting of filter-feeding animals. A provocative recent review even concluded that, “the accumulation of plant biomass in shallow benthic habitats is more likely controlled by consumer effects than by nutrients” (Heck & Vallentine, 2007, p. 378). Of course, these factors might also change conditions in ways that lead to oligotrophication, for example, the ‘artificial lake effect’ by which dams can reduce the concentrations of inorganic nutrients and change nutrient ratios. However, I would like to close with the speculation that there may be another important and even larger scale macroscopic factor that can stimulate eutrophication or oligotrophication in coastal marine systems—climate-induced changes in ecosystem phenology. Phenology has been a common term long used by botanists, but climate change is making it popular with a much wider audience (e.g., Schwartz, 2003) and there is now a well-established European Phenology Network (http://www.dow.wau.nl/msa/epr/about_EPN.asp). Defined simply, phenology is the science of the relations between climate and periodic biological phenomena.

It is becoming clearer that climate-induced changes in phenology can have profound effects on coastal ecology (e.g., Townsend & Cammen, 1988; Sullivan et al., 2001; Grall & Chauvaud, 2002; Edwards & Richardson, 2004; Oviatt, 2004; Ozaki et al., 2004; Fulweiler & Nixon, this volume, and many of the articles cited in support of the macroscopic view). In the case of the system that I know best, Narragansett Bay, RI (USA), changing phenology may be reducing productivity at the same time that other aspects of climate change may be exacerbating hypoxia, a condition normally associated with eutrophication. Very briefly, warmer and cloudier winters seem to be delaying or eliminating the initiation of the traditional late winter–early spring diatom bloom (Li & Smayda, 1998; Keller et al., 1999; Borkman, 2002; Oviatt et al., 2002) (Fig. 7). The summer and autumn blooms that are replacing the winter–spring bloom are often less intense and of much shorter duration, with the result that the mean annual and summer chlorophyll concentrations in the middle of the bay are only about one-third of those found in the 1970s (Li & Smayda, 1998; Fulweiler

Fig. 7 The month of maximum phytoplankton bloom (chlorophyll concentration) in the West Passage of Narragansett Bay (compiled by Betty Buckley, University of Rhode Island, from various sources, including T. Smayda, MERL, and the Graduate School of Oceanography plankton monitoring program)



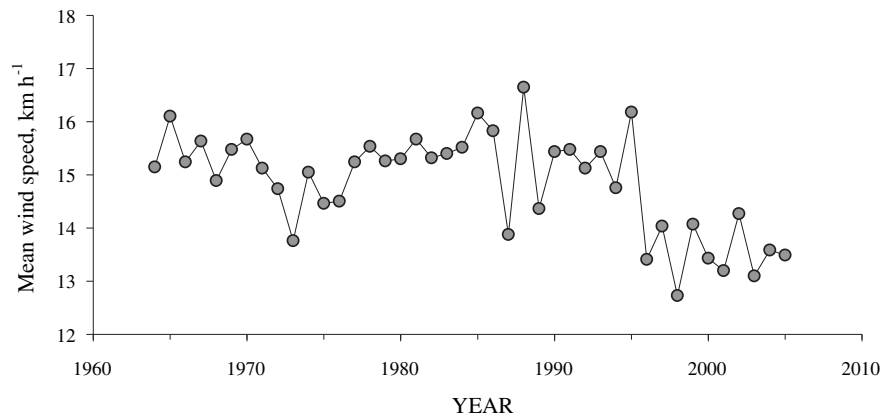
et al., 2007; Nixon et al., 2008; Fulweiler and Nixon, this volume). Declines in chlorophyll have also been documented in the lower bay (Hawk, 1998; Oviatt, 2004). Because of the high correlation between chlorophyll and ^{14}C uptake in this bay (Keller, 1988a, b), the decline in chlorophyll almost certainly means that at least the mid- and lower bay have been undergoing oligotrophication. We attribute the decline in production to climate change because reductions in anthropogenic N inputs to the bay are only just beginning and measurements show essentially unchanging N inputs during at least the last quarter century (Nixon et al., 2005, 2008). The input of anthropogenic P has declined significantly in recent decades, but ecosystem-level experiments have shown that the bay is strongly N limited during summer when productivity is the greatest (Oviatt et al., 1995) and DIN/DIP (dissolved inorganic nitrogen/dissolved inorganic phosphorus) ratios in the surface water are very low during summer (Pilson, 1985).

While chlorophyll (and presumably primary production) has been declining in mid-bay, a growing number of observations have documented that portions of the upper bay experience recurring episodic hypoxia in the bottom waters during summer, particularly during the weakest neap tides (e.g., Granger et al., 2000; Bergondo et al., 2005; Deacutis & Kiernan, 2006; Melrose et al., 2007). These measurements are too recent to establish whether the extent, duration, or intensity of hypoxia is increasing, but the general impression has long been that the major passages of the bay have been too well mixed vertically to develop hypoxia.

If it is correct that hypoxia has increased even as the standing crop of phytoplankton has declined (and

macrophyte production is very small relative to phytoplankton in this system), two climate-related changes may be involved. First, water temperatures have been increasing during the last 30 years at about $0.04^\circ\text{C y}^{-1}$ (Nixon et al., 2004). Based on theoretical considerations, heterotrophic respiration increases with temperature at twice the rate of net primary production (Harris et al., 2006), so that the temperature increase may have made some contribution to an increasing oxygen demand in the bay. There also appear to have been significant declines in the mean wind speed over the upper bay. During the windiest months (F, M, A) mean speed has been declining since about 1970 (from about 20 to about 16 km h^{-1}) while the wind speed declined markedly only after 1996 for the least windy months (J, A, S) (Pilson, 2008) (Fig. 8). Between 1964 and 1995, the mean ($\pm\text{SD}$) summer wind speed was about $15.3 \pm 0.6 \text{ km h}^{-1}$, while it averaged only about $13.5 \pm 0.5 \text{ km h}^{-1}$ during the next 10 years, the period during which regular oxygen monitoring really began. Since the power of the wind to mix the water column vertically varies approximately as the cube of the speed, this would represent a drop of some 30% in summer wind mixing potential during the last decade (Niiler & Kraus, 1977; Husby & Nelson, 1982). The decline in speed appears to have been associated with the easterly, cross-bay component of the wind rather than the north-directed component that aligns with the greatest fetch of the bay (Pilson, 2008). While intriguing, these changes have so far only been described for a single monitoring station and the ‘step function’ decline in the summer mean wind speed is particularly suspicious. If the wind reduction is confirmed, the situation in Narragansett Bay will

Fig. 8 Mean wind speed measured by the National Weather Service (NOAA) at the T. F. Green airport in Warwick, RI during the least windy months of the year; July, August, and September 1964 through 2005 (modified from Pilson, 2008)



not be unique. Conley et al. (2007) have shown that changes in wind were probably responsible for declining concentrations of dissolved oxygen in some Danish coastal waters despite significant nitrogen discharge reductions and a similar situation is emerging in western Long Island Sound in the United States. (L. Swanson, State University of New York, Stony Brook, pers. comm.).

With such a possible decline in wind mixing, the question arises about other factors that might potentially increase vertical stratification in the upper bay. The warming surface water is obvious, but the most dramatic warming is in winter and the summer increase is small. Stratification in the Upper Bay is much more related to vertical salinity differences that are largely a function of river flow. While there has been a long-term increase in rainfall in this area (about 30% over the last century), the average annual river flow does not appear to have increased, at least since 1970 (Pilson, 2008).

In response to concerns about the hypoxia in the Upper Bay and fears that it may be spreading down the bay, as well as a dramatic fish kill in a side embayment of Narragansett Bay proper, state regulators and politicians have mandated major reductions in N discharge from the larger sewage treatment plants. When fully implemented, these reductions may lower the amount of N entering Narragansett Bay from land between May and October by about 35–40% (Nixon et al., 2008). This will almost certainly have a major impact on primary production in the bay during the time when benthic and pelagic animals are growing. However, this impact will fall on a bay that has already had a large reduction in primary production, or at least in the mean standing

crop of phytoplankton in mid bay. We believe that this has already had an impact on the benthos and on benthic-pelagic coupling, at least in the mid-bay (Fulweiler et al., 2007; Fulweiler & Nixon, this volume).

Neither the reduced concentrations of chlorophyll nor the potential role of changes in the wind have been part of a management/policy discussion that has focused only on the traditional picture of nutrients (which activists imply have been increasing in spite of the evidence to the contrary), hypoxia (which may or may not have been getting worse), and dead fish (one significant kill in the last 100 years). The macroscopic view would include the dramatic impacts of climate change (warming and cloud cover) on the timing and magnitude of primary production, the consequences of these changes on higher trophic levels (e.g., warmer winters allow the earlier appearance and greater abundance of ctenophores that prey on the herbivorous copepods during spring and summer; Sullivan et al., 2001), and the potential role of changes in the wind on hypoxia. What are the ecological and economic trade-offs in possibly improving hypoxia in the upper bay at the cost of increasing food limitation over the mid and lower bay (about 70% of the total area of the system)?

We must also be mindful that Narragansett Bay and virtually all the coastal systems have been exposed to many pressures in addition to climate change since they began receiving large amounts of nutrients in the second half of the 1800s (urban, point-source dominated) or the second half of the 1900s (non-point-source dominated). In a talk at the most recent (autumn 2007) meeting of the Coastal and Estuarine Research Federation, Carlos Duarte

pointed out that those who believe that reducing nutrient inputs will return coastal ecosystems to some pristine state have forgotten that the ‘baseline’ has been changing. Like Peter Pan, he said, they want to return to ‘Neverland’ where time stands still and nothing ever changes. The scientific community, of course, does not make decisions about policy. However, I do suggest that all of us, scientists, regulators, politicians, and even the activists need to consider coastal marine eutrophication and oligotrophication as the fundamental ecological processes they are. They are not simple ‘pollution problems’ but major ecological changes that must be viewed through the microscope.

Acknowledgements I thank the organizers of the International Symposium on Research and Management of Eutrophication in Coastal Ecosystems as well as the sponsors (Danish EPA, Swedish EPA, Fyn County, and DHI Water & Environment) for making it possible for me to participate and present an introductory talk, on which this article is based. Robinson Fulweiler contributed much to this article through discussions and assistance with graphics and editing. I also thank Jesper Andersen and an anonymous reviewer for helpful comments on an earlier draft. Vicki Soto, Sybil Seitzinger, and Michael Pilson made unpublished data available. T. Yamamoto and T. Nagai helped me to understand the situation in the Seto Inland Sea. Bob Beardsley and Dan Codiga shared information on wind speed and vertical mixing. The RI Sea Grant Program has long supported my studies of coastal marine eutrophication. My study on this topic has also benefited greatly from a long collaboration with Betty Buckley, Steve Granger, Michael Pilson, Candace Oviatt, and Virginia Lee, and for this I am most grateful.

References

- Anderson, N. J., E. Jeppesen & M. Soendergaard, 2005. Ecological effects of reduced nutrient loading (oligotrophication) on lakes: an introduction. *Freshwater Biology* 50: 1589–1593.
- 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.
- Bergondo, D. L., D. Kester, H. Stoffel & W. Woods, 2005. Time-series observations during the low sub-surface oxygen events in Narragansett Bay during summer 2001. *Marine Chemistry* 97: 90–103.
- Borkman, D., 2002. Analysis and simulation of *Skeletonema costatum* (Grev.) Cleve annual abundance patterns in Lower Narragansett Bay 1959–1996. Ph.D. Thesis, University of Rhode Island.
- Brush, M. J., J. W. Brawley, S. W. Nixon & J. N. Kremer, 2002. Modeling phytoplankton production: problems with the Eppley curve and an empirical alternative. *Marine Ecology Progress Series* 238: 31–45.
- Carstensen, J., D. Conley, J. Andersen & G. Ærtebjerg, 2006. Coastal eutrophication and trend reversal: a Danish case study. *Limnology and Oceanography* 51: 398–408.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Cloern, J. E., A. Jassby, J. Thompson & K. Hieb, 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. *Proceedings on National Academy of Sciences* 104: 18561–18565.
- Colombo, G., I. Ferrari, V. Ceccherelli & R. Rossi (eds), 1992. *Marine Eutrophication and Population Dynamics*. Olsen and Olsen, Fredensborg
- Conley, D., J. Carstensen, G. Ærtebjerg, P. Christensen, T. Dalsgaard, J. Hansen & A. Josefson, 2007. Long-term changes and impacts of hypoxia in Danish coastal waters. *Ecological Applications* 17: S165–S184.
- Cushing, D. H., 1975. *Marine Ecology and Fisheries*. Cambridge University Press, London.
- de Jong, F., 2006. *Marine Eutrophication in Perspective—On the Relevance of Ecology and Environmental Policy*. Springer, Berlin.
- Deacutis, C. & S. Kiernan, 2006. Sensors monitor water quality in Narragansett Bay. 410 N. 3: 7–8. Rhode Island Sea Grant, University of Rhode Island. Available on line at: <http://seagrant.gso.uri.edu/41N>.
- Edwards, M. & A. Richardson, 2004. Impact of climate change in marine pelagic phenology and trophic mismatch. *Nature* 430: 881–884.
- Fulweiler, R. W. & S. W. Nixon, 2008. Responses of benthic-pelagic coupling to climate change in a temperate estuary. *Hydrobiologia* (this volume).
- Fulweiler, R. W., S. Nixon, B. Buckley & S. Granger, 2007. Reversal of the net dinitrogen gas flux in coastal marine sediments. *Nature* 448: 180–182.
- Grall, J. & L. Chauvaud, 2002. Marine eutrophication and benthos: the need for new approaches and concepts. *Global Change Biology* 8: 813–830.
- Granger, S., M. Brush, B. Buckley, M. Traber, M. Richardson & S. Nixon, 2000. An assessment of eutrophication in Greenwich Bay. In Schwartz, M. (ed.), *Restoring Water Quality in Greenwich Bay: A White Paper Series*. Paper No. 1. Rhode Island Sea Grant, Narragansett.
- Grebmeier, J., J. Overland, S. Moore, E. Farley, E. Carmack, L. Cooper, K. Frey, J. Helle, F. McLaughlin & S. McNutt, 2006. A major ecosystem shift in the northern Bering Sea. *Science* 311: 1461–1464.
- Greening, H. & A. Janicki, 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. *Environmental Management* 38: 163–178.
- Hamlin, C., 1990. *A Science of Impurity—Water Analysis in Nineteenth Century Britain*. University of California Press, California.
- Harris, L. A., C. M. Duarte & S. W. Nixon, 2006. Allometric laws and prediction in estuarine and coastal ecology. *Estuaries and Coasts* 29: 340–344.
- Hawk, J., 1998. The role of the North Atlantic Oscillation in winter climate variability as it relates to the winter-spring

- bloom in Narragansett Bay. M.S. Thesis, University of Rhode Island.
- Heck, K. L. & J. Vallentine, 2007. The primacy of top-down effects in shallow benthic ecosystems. *Estuaries and Coasts* 30: 371–381.
- Heip, C., 1995. Eutrophication and zoobenthos dynamics. *Ophelia* 41: 113–136.
- Howarth, R. W. & R. Marino, 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. *Limnology and Oceanography* 51: 364–376.
- Husby, D. M. & C. Nelson, 1982. Turbulence and vertical stability in the California Current. *CalCOFI Report* 23: 113–129.
- Jeppesen, E., J. Jensen & M. Soendergaard, 2002. Response of phytoplankton, zooplankton, and fish to re-oligotrophication: an 11 year study of 23 Danish lakes. *Aquatic Ecosystem Health and Management* 5: 31–43.
- Jeppesen, E., M. Soendergaard, J. Jensen, K. Havens, O. Anneville, L. Carvalho, M. Coveney, R. Deneke & M. Dokulil, 2005. Lake responses to reduced nutrient loading—An analysis of contemporary long-term data from 35 case studies. *Freshwater Biology* 50: 1747–1771.
- Keller, A., 1988a. An empirical model of primary productivity (^{14}C) using mesocosm data along a nutrient gradient. *Journal of Plankton Research* 10: 813–834.
- Keller, A., 1988b. Estimating phytoplankton productivity from light availability and biomass in the MERL mesocosms and Narragansett Bay. *Marine Ecology Progress Series* 45: 159–168.
- Keller, A., C. Oviatt, H. Walker & J. Hawk, 1999. Predicted impacts of elevated temperature on the magnitude of the winter-spring phytoplankton bloom in temperate coastal waters: a mesocosm study. *Limnology and Oceanography* 44: 344–356.
- Ketchum, B. H., 1972. *The Water's Edge: Critical Problems of the Coastal Zone*. MIT Press, Cambridge.
- Lauff, G. H. (ed.), 1967. *Estuaries*. American Association for the Advancement of Science, Pub. No 83.
- Li, Y. & T. Smayda, 1998. Temporal variability of chlorophyll in Narragansett Bay 1973–1990. *ICES Journal of Marine Science* 55: 661–667.
- Melrose, D., C. Oviatt & M. Berman, 2007. Hypoxic events in Narragansett Bay, Rhode Island, during the summer of 2001. *Estuaries and Coasts* 30: 47–53.
- Mills, E. L., 1989. *Biological Oceanography: An Early History, 1870–1960*. Cornell University Press, NY.
- National Research Council, 1993. *Managing Wastewater in Coastal Urban Areas*. National Academy Press, Washington DC.
- Ney, J. J., 1996. Oligotrophication and its discontents: effects of reduced nutrient loading on reservoir fisheries. *American Fisheries Society Symposium* 16: 285–295.
- Niiler, P. P. & E. Kraus, 1977. One-dimensional models of the upper ocean. In Krauss, E. (ed.), *Modelling and Prediction of the Upper Layers of the Ocean*. Pergamon Press, NY: 1443–1472.
- Nixon, S. W., 1988. Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnology and Oceanography* 33: 1005–1025.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Nixon, S. W., 1996. Regional coastal research—What is it? Why do it? What role should NAML play? *Biological Bulletin* 190: 252–259.
- Nixon, S. & B. Buckley, 2002. 'A strikingly rich zone'—Nutrient enrichment and secondary production in coastal marine ecosystems. *Estuaries* 25: 782–796.
- Nixon, S. W., C. A. Oviatt, J. Frithsen & B. Sullivan, 1986. Nutrients and the productivity of estuarine and coastal marine ecosystems. *Journal of the Limnological Society of Southern Africa* 12: 43–71.
- Nixon, S. W., B. Buckley, S. Granger & J. Bintz, 2001. Responses of very shallow marine ecosystems to nutrient enrichment. *Human and Ecological Risk Assessment* 7: 1457–1481.
- Nixon, S. W., S. Granger, B. Buckley, M. Lamont & B. Rowell, 2004. A one hundred and seventeen year coastal water temperature record from Woods Hole, Massachusetts. *Estuaries* 27: 397–404.
- Nixon, S., B. Buckley, S. Granger, L. Harris, A. Oczkowski, L. Cole & R. Fulweiler, 2005. Anthropogenic Nutrient Inputs to Narragansett Bay: A Twenty-Five Year Perspective. Report to the Narragansett Bay Commission and Rhode Island Sea Grant. Available at <http://seagrant.gso.uri.edu/research/news.html>.
- Nixon, S., B. Buckley, S. Granger, L. Harris, A. Oczkowski, R. Fulweiler & L. Cole, 2008. Nutrient inputs to Narragansett Bay: past, present, and future. In Desbonnet, A. & B. Costa-Pierce (eds), *Science for Ecosystem-Based Management: Narragansett Bay in the 21st Century*. Springer-Verlag, NY: 101–175.
- Odum, H. T., 1993. In Platt, D. (ed.), 1993. *The System in the Sea*. Island Institute, Rockland: 34–40.
- Odum, H. T. & C. Hoskin, 1958. Comparative studies of the metabolism of Texas bays. *Publications of the Institute of Marine Science, University of Texas* 5: 16–46.
- Oviatt, C. A., 2004. The changing ecology of temperate coastal waters during a warming trend. *Estuaries* 27: 895–904.
- Oviatt, C. A., P. Doering, B. Nowicki, L. Reed, J. Cole & J. Frithsen, 1995. An ecosystem level experiment on nutrient limitation in temperate coastal marine environments. *Marine Ecology Progress Series* 16: 247–254.
- Oviatt, C. A., A. Keller & L. Reed, 2002. Annual primary production in Narragansett Bay with no bay-wide winter-spring phytoplankton bloom. *Estuarine, Coastal and Shelf Science* 54: 1013–1026.
- Ozaki, K., S. Uye, T. Kusumoto & T. Hagino, 2004. Interannual variability of the ecosystem of the Kii Channel, the Inland Sea of Japan, as influenced by bottom intrusion of cold and nutrient-rich water from the Pacific Ocean, and a recent trend of warming and oligotrophication. *Fisheries Oceanography* 13: 65–79.
- Paine, R. T., 2005. Cross environmental talk in ecology: fact or fantasy? *Marine Ecology Progress Series* 304: 271–307.
- Pearson, T. H. & R. Rosenberg, 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Annual Review of Oceanography and Marine Biology* 16: 229–311.

- Philippart, C., J. Beukema, G. Cadée, R. Dekker, P. Goedhart, J. van Iperen, M. Leopold & P. Herman, 2007. Impacts of nutrient reduction on coastal communities. *Ecosystems* 10: doi:10.1007/s10021-006-9006-7.
- Pilson, M., 1985. Annual cycles of nutrients and chlorophyll in Narragansett Bay, Rhode Island. *Journal of Marine Research* 43: 849–873.
- Pilson, M., 2008. Narragansett Bay amidst a globally changing climate. In Desbonnet, A. & B. Costa-Pierce (eds), *Science for Ecosystem-Based Estuary Management*. Narragansett Bay in the 21st Century. Springer-Verlag, NY: 35–46.
- Platt, D. D. (ed.), 1993. *The System in the Sea: Applying Ecosystems principles to Marine Fisheries*. The Island Institute, Rockland.
- Rabalais, N., 2002. Nitrogen in aquatic ecosystems. *Ambio* 31: 102–112.
- Rabalais, N. N. & S. W. Nixon, 2002. Nutrient over-enrichment in coastal waters: global patterns of cause and effect. *Estuaries* 25: 4B.
- Revelle, R. & H. E. Suess, 1957. Carbon dioxide exchange between atmosphere and ocean and the question of an increase in atmospheric CO₂ during the past decades. *Tellus* 9: 18–27.
- Richardson, K. & J. P. Heilmann, 1995. Primary production in the Kattegat: past and present. *Ophelia* 41: 317–328.
- Rigler, F. H. & R. H. Peters, 1995. *Science and Limnology. Excellence in Ecology*, Vol. 6. The Ecology Institute, Oldendorf/Luhe, Germany.
- Ryther, J. H., 1954. The ecology of phytoplankton blooms in Moriches bay and Great South Bay, Long Island, New York. *Biological Bulletin* 106: 198–209.
- Ryther, J. H., 1989. Historical perspective of phytoplankton blooms on Long Island and the Green Tides of the 1950s. In Coper, E. M., V. M. Bricelj & E. J. Carpenter (eds), *Novel Phytoplankton Blooms*. Springer-Verlag, NY: 375–382.
- Ryther, J. H. & C. S. Yentsch, 1957. The estimation of phytoplankton production in the ocean from chlorophyll and light data. *Limnology and Oceanography* 2: 282–286.
- Sand-Jensen, K. & J. Borum, 1991. Interactions among phytoplankton, periphyton, and macrophytes in temperate freshwaters and estuaries. *Aquatic Botany* 41: 137–175.
- Schell, D. M., 2000. Declining carrying capacity in the Bering Sea: isotopic evidence from whale baleen. *Limnology and Oceanography* 45: 459–462.
- Schindler, D. W., 1981. Studies of eutrophication in lakes and their relevance to the estuarine environment. In Neilson, B. & L. E. Cronin (eds), *Estuaries and Nutrients*. Humana Press, NY: 71–82.
- Schindler, D. W., 2006. Recent advances in the understanding and management of eutrophication. *Limnology and Oceanography* 51: 356–363.
- Schulman, A., 2005. A warm unwelcome – Seabirds suffer as climate change unravels North Sea food web. *Grist Environmental News & Commentary*. <http://www.grist.org/news/maindish2005/01/25/schulman-seabirds/>.
- Schwartz, M. D. (ed.), 2003. *Phenology: An Integrative Environmental Science*. Kluwer, Dordrecht.
- Sfriso, A., B. Pavoni & A. Marcomini, 1989. Macroalgae and phytoplankton standing crops in the central Venice Lagoon: primary production and nutrient balance. *Science of the Total Environment* 80: 139–159.
- Smil, V., 2001. *Enriching the Earth*. MIT Press, Cambridge.
- Smith, V. H., S. B. Joye & R. W. Howarth, 2006. Eutrophication of freshwater and marine ecosystems. *Limnology and Oceanography* 51: 351–355.
- Soetaert, K., J. Middelburg, C. Heip, P. Meire, S. Van Damme & T. Maris, 2006. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt Estuary (Belgium, The Netherlands). *Limnology and Oceanography* 51: 409–423.
- Steffen, W., P. Crutzen & J. McNeill, 2007. The anthropocene: are humans now overwhelming the great forces of nature? *Ambio* 36: 614–621.
- Stockner, J. G., E. Rydin & P. Hyenstrand, 2000. Cultural oligotrophication: causes and consequences for fisheries resources. *Fisheries* 25: 7–14.
- Stoddard, A., J. Harcum, J. Simpson, J. Pagenkopf & R. Bastian, 2002. *Municipal Wastewater Treatment: Evaluating Improvements in National Water Quality*. John Wiley & Sons, New York.
- Sullivan, B. K., D. Van Keuren & M. Clancy, 2001. Timing and size of blooms of the ctenophore *Mnemiopsis leidyi* in relation to temperature in Narragansett Bay, Rhode Island. *Hydrobiologia* 451: 113–120.
- Townsend, D. & L. Cammen, 1988. Potential importance of the timing of spring plankton blooms to benthic-pelagic coupling and recruitment of juvenile demersal fishes. *Biological Oceanography* 5: 215–229.
- Ulanowicz, R. E., 1986. A phenomenological perspective of ecological development. In Poston, T. & R. Purdy (eds), *Aquatic Toxicology and Environmental Fate*, Vol. 9. American Society for Testing and Materials, Philadelphia: 73–81.
- Valiela, I., 2006. *Global Coastal Change*. Blackwell Publishing, Oxford.
- Vollenweider, R. A., R. Marchetti & R. Viviani (eds), 1992. *Marine Coastal Eutrophication*. Elsevier, Amsterdam.
- Wulff, F., 1990. Overall conclusions. In Lancelot, C., G. Billen & H. Barth (eds), *Eutrophication and Algal Blooms in North Sea Coastal Zones, the Baltic and Adjacent Areas: Prediction and Assessment of Preventive Actions*. Water Pollution Research Report 12. Commission of the European Communities' Environmental Program, Brussels: 267–271.
- Yamamoto, T., 2003. The Seto Inland Sea—Eutrophic or oligotrophic? *Marine Pollution Bulletin* 47: 37–42.

Ecosystem thresholds with hypoxia

Daniel J. Conley · Jacob Carstensen ·
Raquel Vaquer-Sunyer · Carlos M. Duarte

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 21–29.
DOI: 10.1007/s10750-009-9764-2 © Springer Science+Business Media B.V. 2009

Abstract Hypoxia is one of the common effects of eutrophication in coastal marine ecosystems and is becoming an increasingly prevalent problem worldwide. The causes of hypoxia are associated with excess nutrient inputs from both point and non-point sources, although the response of coastal marine ecosystems is strongly modulated by physical processes such as stratification and mixing. Changes in climate, particularly temperature, may also affect the susceptibility of coastal marine ecosystems to hypoxia. Hypoxia is a particularly severe disturbance because it causes death of biota and catastrophic changes in the ecosystem. Bottom water oxygen deficiency not only influences the

habitat of living resources but also the biogeochemical processes that control nutrient concentrations in the water column. Increased phosphorus fluxes from sediments into overlying waters occur with hypoxia. In addition, reductions in the ability of ecosystems to remove nitrogen through denitrification and anaerobic ammonium oxidation may be related to hypoxia and could lead to acceleration in the rate of eutrophication. Three large coastal marine ecosystems (Chesapeake Bay, Northern Gulf of Mexico, and Danish Straits) all demonstrate thresholds whereby repeated hypoxic events have led to an increase in susceptibility of further hypoxia and accelerated eutrophication. Once hypoxia occurs, reoccurrence is likely and may be difficult to reverse. Therefore, elucidating ecosystem thresholds of hypoxia and linking them to nutrient inputs are necessary for the management of coastal marine ecosystems. Finally, projected increases in warming show an increase in the susceptibility of coastal marine ecosystems to hypoxia such that hypoxia will expand.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

D. J. Conley (✉)
GeoBiosphere Science Centre, Department of Geology,
Lund University, Sölvegatan 12, 22362 Lund, Sweden
e-mail: daniel.conley@geol.lu.se

J. Carstensen
Department of Marine Ecology, National Environmental
Research Institute, Aarhus University, P.O. Box 358,
4000 Roskilde, Denmark

R. Vaquer-Sunyer · C. M. Duarte
Instituto Mediterraneo de Estudios Avanzados,
CSIC-Univ. Illes Balears, C/Miquel Marqués 21,
07190 Esporles, Islas Balears, Spain

Keywords Hypoxia · Biogeochemistry ·
Nitrogen · Phosphorus · Regime shift ·
Resilience

Introduction

Hypoxia is one of the common effects of eutrophication in coastal marine ecosystems, and dead zones

created by hypoxia have spread exponentially globally since the 1960s (Diaz & Rosenberg, 2008). Increased prevalence of hypoxia is directly a result of excess nutrient inputs from both point and diffuse sources, thus increasing the production of organic matter. The increased flux of organic matter to bottom waters changes the balance between oxygen supply through physical forcing and oxygen consumption from decomposition of organic material. It should be stressed that hypoxia in coastal marine ecosystems is strongly modulated by physical processes such as stratification and mixing (Smith et al., 1992; Hagy et al., 2004); without stratification or with intense mixing renewing oxygen supplies, hypoxia will not occur. Hypoxia is a particularly severe disturbance because it causes death of organisms and catastrophic changes in the ecosystem (Gray et al., 2002).

In this review, we will explore the consequences of hypoxia to changes in the remineralization of organic matter and to the modification of the biogeochemical cycles of phosphorus and nitrogen, and show how they act as feedback mechanisms to further eutrophication. In addition, we will demonstrate that repeated hypoxic events will increase the susceptibility of coastal marine ecosystems to further hypoxia through alteration of ecosystem functioning of the sediments and show that this has already occurred in a number of coastal marine ecosystems. Finally, we will examine the effects of predicted increases in global temperature on hypoxia.

Biogeochemical consequences

Hypoxia and bottom water oxygen deficiency not only negatively influence organisms (Vaquer-Sunyer & Duarte, 2008) and the habitat of living resources, but also the biogeochemical processes that control nutrient concentrations in the water column. Internal feedbacks on biogeochemical processes occur with oxygen depletion, with one of the most prominent being the effect on the phosphorus (P) biogeochemical cycle. Increased P fluxes from sediments into overlying waters with hypoxia is a classic response observed in freshwater systems (Mortimer, 1941) and has been well documented in coastal marine ecosystems (Nixon et al., 1980; Conley et al., 2002). When oxygen conditions in sediments become low,

dissolved inorganic phosphorus (DIP) is released from iron-bound phosphorus in sediments (Jensen et al., 1995). In the Baltic Sea, which is the largest coastal area in the world to suffer from eutrophication-induced hypoxia, large internal P loading occurs with hypoxia (Fig. 1). The amount of DIP released from sediments in the Baltic is an order of magnitude larger than external inputs from rivers (Conley et al., 2002). The increased sediment–water fluxes of phosphorus with hypoxia return DIP to surface waters alleviating P limitation (Conley, 1999), stimulating phytoplankton production and acting as a positive feedback to increase hypoxia.

Hypoxia also influences rates of denitrification and anaerobic ammonium oxidation (anammox). Denitrification is one of the major routes of loss of fixed nitrogen (N) in the oceans (Seitzinger & Giblin, 1996); however, its measurement is difficult (Groffman et al., 2006). Denitrification is the reductive respiration of nitrate or nitrite to N₂ or nitrous oxide (N₂O). In addition, fixed nitrogen removal also occurs through the anammox process (Dalsgaard et al., 2003). The rates of denitrification are dependent on a variety of factors, but the availability of starting products, for example nitrate (Kemp et al., 1990) and carbon (Smith & Hollibaugh, 1989; Sloth et al., 1995), is a major controlling factor.

In a review of continental shelf sediments, Seitzinger & Giblin (1996) found a linear relationship between denitrification rate and sediment oxygen uptake. In general, the efficiency rate of denitrification increases with sediment respiration. However, as

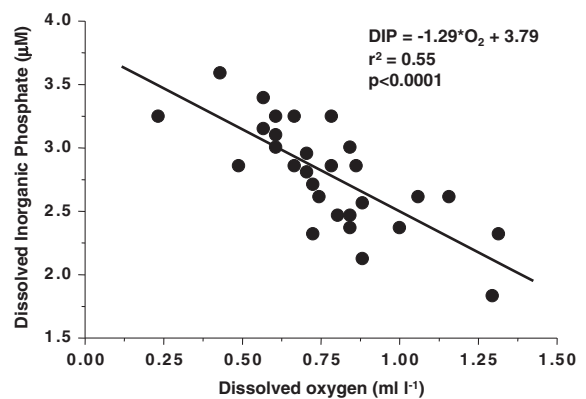


Fig. 1 Relationship between bottom water oxygen concentrations and DIP concentrations in hypoxic bottom waters from the Baltic Sea (redrawn from Conley et al., 2002)

the sediments become more reducing and more N is remobilized as ammonium and less as nitrate, the rates of denitrification slow such that denitrification can be shut down at high respiration rates. Several studies have demonstrated that in estuarine systems, denitrification displays a threshold-like behavior (Webster & Harris, 2004; Eyre & Ferguson, 2009), increasing to a maximum of carbon decomposition, and then decreasing as sediments become more reducing (Fig. 2). Thus, when coastal and estuarine systems become hypoxic, there is a large risk that the loss of nitrogen will decrease (Smith & Hollibaugh, 1999), thus increasing the availability of DIN and acting as a positive feedback that increases the potential for eutrophication.

The conventional belief that the efficiency of denitrification is reduced when oxygen conditions are low has been challenged with the recent observation that massive amounts of denitrification occur in the water column of hypoxic zones in the open ocean (Deutsch et al., 2007). Recently, Vahtera et al. (2007) found a negative correlation between hypoxic water volume and the total dissolved inorganic nitrogen pool in the Baltic Sea. This suggests greater overall nitrogen removal as the area of hypoxia increases, although current measurements of water column denitrification do not support this hypothesis (Hannig et al., 2007). However, we have no information on the potential for denitrification in the sediments prior to large-scale water column hypoxia in the Baltic Sea. The enhancement or reduction in the rate of denitrification and anammox may depend upon whether systems are seasonally hypoxic or permanently hypoxic and the development of these processes in

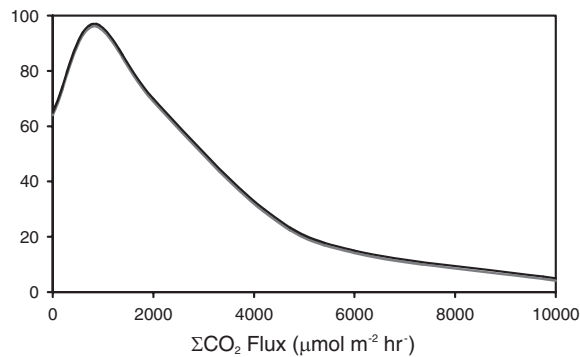


Fig. 2 System average denitrification efficiency as a function of ΣCO_2 efflux for 22 shallow coastal Australia systems (redrawn from Eyre & Ferguson, 2009)

the water column or the sediments. The other important controlling factor governing the loss of nitrogen is certainly the availability of nitrate (Kemp et al., 1990), which is low in many hypoxic zones.

Thresholds of ecosystem hypoxia

The relationship between nutrient inputs and hypoxia may not be a simple one, as there is evidence that once an ecosystem experiences hypoxia, it becomes more vulnerable to future hypoxic events (Conley et al., 2007; Díaz & Rosenberg, 2008). This suggests that the relationship between ecosystem hypoxia and nutrient inputs may show a threshold-like behavior, with a regime shift (Scheffer et al., 2001) involving an abrupt transition from oxic to hypoxic status upon exceeding specific thresholds of nutrient inputs. Regime shifts derive from non-linear responses in ecosystems (Muradian, 2001) where qualitative changes, involving changes in ecosystem buffers and ecosystems processes, lead to an abrupt change in status. These non-linear responses to pressures also imply that ecosystems displaying a threshold-like behavior typically show resistance to return to the original status, resulting in a hysteresis in the recovery to reduced pressures (Muradian, 2001). We argue here that the development of coastal hypoxia can be best conceptualized and examined as a case for ecosystem thresholds conducive to a regime shift that may affect the reversibility of the problem.

As shown above, hypoxia can act as a positive feedback to enhance the effects of eutrophication. In the Baltic Sea, for example, excess DIP remaining after the spring bloom stimulates the growth of large nuisance blooms of nitrogen-fixing cyanobacteria during the summer, with fixation of new nitrogen and the sinking carbon further enhancing hypoxic conditions (Vahtera et al., 2007). Large sediment–water fluxes of DIP occur with hypoxia, returning DIP to a partially P-limited water column, stimulating phytoplankton growth and acting as a positive feedback to increase hypoxia. It has been hypothesized that the Baltic Sea has experienced several ecosystem states with respect to fish production, with eutrophication and enhanced DIP release from anoxic bottoms together with enhanced nitrogen fixation providing a stabilizing mechanism (Österblom et al., 2007).

Recent studies in other coastal marine ecosystems suggest that repeated hypoxic events can help to sustain future hypoxic conditions (Hagy et al., 2004; Kemp et al., 2005). Large-scale changes in benthic communities occur with hypoxia reducing the abundance of large, slow-growing, deeper-dwelling animals, and changing to smaller, fast-growing species that can colonize surface sediments rapidly following hypoxia (Díaz & Rosenberg, 1995). These smaller species do not have the capability to irrigate and bring oxygen downward into the sediments, so that the change in the community structure implies a change in functioning, with the loss of a major buffer against hypoxia. The loss of benthic communities and the inability of the communities to recover with repeated hypoxic events (Karlson et al., 2002) may render ecosystems more vulnerable to future development and persistence of hypoxia. A further loss of sediment-buffering capacity against hypoxia results from the consumption of electron acceptors during the onset of hypoxia (NO_3 , O_2 , Fe^{2+} , and Mn^{2+}). Once these are depleted during hypoxic events, there is a qualitative change in the sediment metabolism from aerobic to anaerobic pathways involving profound changes in the rates and processing of organic matter (Soetaert & Middelburg, 2009). Once sediments and water become anoxic, anaerobic metabolism leads to the accumulation of reduced metabolites such as sulfides in the system, which titrate oxygen diffusing into the system, thus acting as buffers preventing re-oxygenation of the environment. For example, sulfide accumulation has been shown to be a major buffering process acting to maintain sediments anoxic under seagrass beds that receive excess organic inputs, as sulfide is rapidly oxidized thereby consuming oxygen inputs (Holmer et al., 2003).

Hence, ecosystem hypoxia leads to the loss of important ecosystem buffers that favor the maintenance of oxic conditions (sediment bio-irrigation and electron acceptors) and the appearance of new buffers that act in the opposite direction to maintain ecosystems hypoxic (Fig. 3). Ecosystem buffers are responsible for the resilience of ecosystems. Once they are changed with hypoxia, the resilience of the ecosystems against hypoxia is eroded and new buffers, acting to build resilience to the hypoxic state, appear that increase ecosystem vulnerability to this disturbance and tend to perpetuate hypoxia.

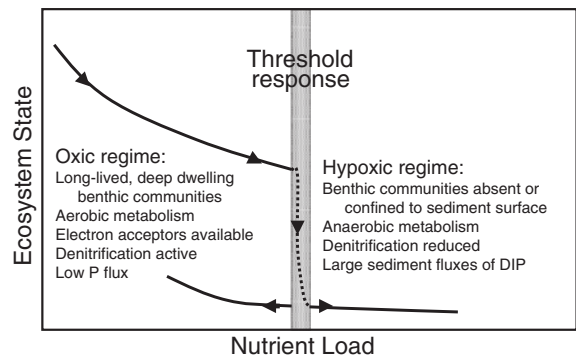


Fig. 3 Hypothetical diagram showing change in ecosystem state as represented by bottom water oxygen concentrations versus nutrient concentrations. An ecosystem is believed to reach a threshold when bottom-living organisms are killed and the system switches to one dominated by anaerobic processes

We have tested for changes in ecosystem state in three large coastal marine ecosystems that have experienced hypoxia over the last decades (Fig. 4A–C): the Danish Straits (Conley et al., 2007), Chesapeake Bay (Hagy et al., 2004), and the Gulf of Mexico (Rabalais et al., 2002). We used a change-point detection method (Gombay and Horvath, 1996) to test for the significance of thresholds in time-series data assuming observations to be normally distributed and a change in the mean or in the linear relation to the driver to occur at an unknown point in time. For the Danish Straits, two significant change points were detected (Fig. 4A): in 1986 (Conley et al., 2007) and again after the record hypoxia in 2002 (HELCOM, 2003). Note that the second shift in 2002 was not found in Conley et al. (2007) because there were not as many observations reported following the 2002 event. In the Danish Straits, hypoxia has worsened following the appearance of large-scale hypoxic events (Fig. 4A; Conley et al., 2007). For Chesapeake Bay, we determined that the significant change occurred in 1986 (Fig. 4B). A persistent increase in the susceptibility of Chesapeake Bay has previously been noted by Hagy et al. (2004) and Kemp et al. (2005), although the time-point of change was not previously pinpointed. Finally, the Gulf of Mexico has exhibited a significant increase in hypoxia (Fig. 4C) following the 1993 record flooding in the Mississippi River (Rabalais et al., 2002). Higher sedimentary oxygen demand in the Gulf of Mexico has been hypothesized

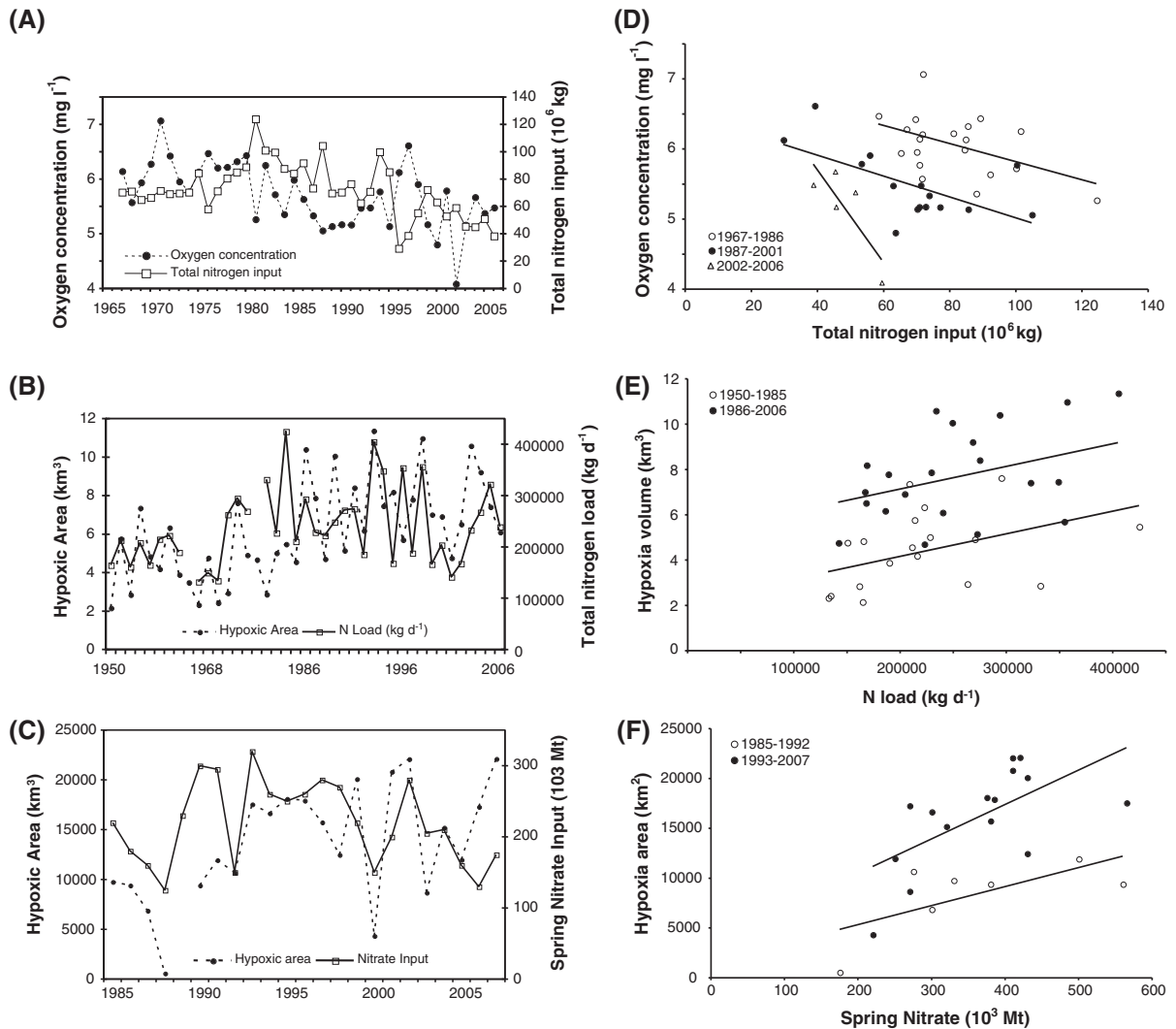


Fig. 4 Systems displaying significant thresholds ($P < 0.05$) in hypoxia through time (A–C). The significance of the thresholds was tested with a change-point detection method (Gombay & Horvath, 1996). **A** and **D**: Danish Straits (Conley et al., 2007); **B** and **E**: Chesapeake Bay (Hagy et al., 2004; Scavia et al., 2006; D. Scavia, University of Michigan, unpubl. data); and **C** and **F**: size of the hypoxic zone in the northern Gulf of Mexico (N.N. Rabalais, Louisiana Universities Marine Consortium

to result in an increase in the size of the hypoxic zone and can be considered a shift to an alternative steady state (Turner et al., 2008). Hypoxia in the Gulf of Mexico has worsened after 1993; for example, smaller river flows have induced a larger response in hypoxia since then (Fig. 4C), with the occurrence of large hypoxic areas over 15,000 km² observed in many of the following years. Subsequently, we examined the slope of nitrogen loading versus oxygen

(LUMCON), unpubl. data, <http://www.gulphypoxia.net/>) and spring (April, May, June) nitrate + nitrite inputs from the Mississippi and Atchafalya rivers (Alexander et al., 2008). **D–F** examines the relationship between nitrogen loading and hypoxia. Slopes for the separate periods were not significantly different, and therefore, the thresholds were manifested as a change in the mean of hypoxic conditions only. (The unit Mt signifies metric tonnes)

concentration for the different periods (Fig. 4D–F). In all cases, significant relationships were observed between hypoxia and nitrogen loading, although the slope of the relationship was not significantly different, and therefore, the thresholds were manifested as a change in the mean hypoxic conditions only.

These changes suggest that a regime shift occurs in coastal marine ecosystems affected by large-scale hypoxia (Conley et al., 2007). Regime shifts are often

rapid transitions that occur in an ecosystem state to new alternative states, with changes in biological variables that propagate through several trophic levels (Scheffer et al., 2001; Collie et al., 2004). Regime shifts can have large consequences for fisheries (Collie et al., 2004; Oguz & Gilbert, 2007). The recovery of ecosystems from hypoxia may occur only after long time periods (Díaz, 2001) or with further reductions in nutrient inputs. Although it is possible for coastal marine ecosystems to respond rapidly to large reductions in nutrient loading (Rask et al., 1999; Mee, 2006), experience has generally shown recovery to be greatly delayed, taking years to decades for ecosystems to recover after nutrient inputs are reduced, and probably less than complete recovery is possible (Díaz, 2001; Duarte et al., 2009). One potential concern with regime shifts is that the condition is not easily reversible. The ability of coastal marine ecosystems to recover from hypoxia may be partly related to the ability of organisms to recolonize and the availability of nearby refuges. These observations suggest that the thresholds for hypoxia are dynamic, and that once trespassed, the changes in internal buffers discussed above render the ecosystems more prone to experience future hypoxia, effectively lowering the thresholds of nutrient inputs at which hypoxia takes place and increasing the recurrence of the problem.

Impact of global warming

Coastal hypoxia has been primarily linked to increased nutrient inputs. Yet, this is not the sole control on hypoxia. Temperature is one of the major factors controlling the extent of hypoxia (Conley et al., 2007), acting through a multitude of interacting processes. Temperature increases may enhance stratification, thereby reducing the supply of oxygen to bottom waters by vertical mixing. However, stratification also potentially reduces the upward mixing of nutrients across the pycnocline, which constitutes a considerable source for production, and consequently decreases the export of particulate organic matter to bottom waters, particularly in the summer period. Temperature also affects the respiration of organisms in the water column. A consequence of this could be higher remineralization of organic matter in the upper mixed layer and lower sedimentation of particulate

organic matter, whereby a larger fraction of the organic matter sedimenting out of the upper mixed layer may be respired instead of permanently buried. Increasing temperature may also affect phytoplankton grazing, but it is not clear whether matching of primary production and grazing will improve or worsen. Finally, the solubility of oxygen decreases with temperature, reducing the margin in oxygen concentration separating well-ventilated waters from hypoxia, such that there will be less oxygen for replenishing bottom waters, no matter whether this occurs by vertical mixing or advective transport of subducted surface water.

The latter process is probably the most important effect in relation to temperature for coastal waters with estuarine character. Conley et al. (2007) analyzed long-term oxygen concentrations in the Danish straits with respect to temperature and found, using multiple regression, that they declined by $0.238 (\pm 0.084) \text{ mg O}_2 \text{ l}^{-1} \text{ } ^\circ\text{C}^{-1}$, as predicted by the effect of temperature on the solubility of oxygen (approximately $-0.26 \text{ mg O}_2 \text{ l}^{-1} \text{ } ^\circ\text{C}^{-1}$). The effect of temperature on oxygen production and consumption may also be important. Harris et al. (2006) calculated, using metabolic theory, that a 4°C increase in the summer water temperatures of a northeastern Atlantic estuary will result in a 20% increase in net primary production and a 43% increase in respiration, resulting in an increasing likelihood of system hypoxia. However, in contrast to the effect of temperature increases on oxygen solubility, which is well known and precise, the effects on ecosystem metabolism still need to be verified. In the text below, consequences of global warming on hypoxia will be assessed assuming oxygen solubility to be the major effect of temperature increase and that the other temperature-dependent processes more or less even each other out.

Over the last decades, substantial evidence from both observations and model results has documented an ongoing global warming (Christensen et al., 2007), which is now generally accepted as a fact in the scientific community. However, there is still much debate about how much the globe will warm. Here, we used the temperature projections obtained by the ECHAM4/OPYC3 model (Roeckner et al., 1999) that couples atmosphere, sea-ice, and oceans assuming the atmospheric pCO_2 level to reach $750 \mu\text{atm}$ by 2100. Using this model, Hansen & Bendtsen (2006) obtained regional forecasts for the Danish straits of

surface water temperature increases around 3.5°C, with seasonal variation from 4°C in winter to 3°C in summer. Physical models of the area using aging tracers have shown that bottom waters in the hypoxia areas (July–October) originate as Skagerrak surface waters penetrating into the straits during winter and spring (Hansen & Bendtsen, 2006). With a temperature increase of 4°C, oxygen solubility is lowered by 0.96 mg O₂ l⁻¹, which corresponds well to the extrapolated value (0.95 mg O₂ l⁻¹) using the regression analysis in Conley et al. (2007).

The spatial coverage of hypoxia in the Danish straits has been analyzed with a weekly resolution from 1998 to 2005 using a spatial and temporal interpolation technique of oxygen profiles described in HELCOM (2003). The effect of a 4°C temperature increase was analyzed by subtracting 0.952 mg O₂ l⁻¹ from the estimated oxygen concentrations at the bottom. Areas exposed to hypoxic conditions (<2 mg O₂ l⁻¹) any time during a specific year were identified. During the 8-year period, 1998 had the best oxygen conditions with 968 km² exposed to hypoxia, whereas 2002 was the worst year with 8256 km² (Fig. 5). With a projected decrease in oxygen solubility with warming and all other factors unchanged, the area of hypoxia would have been 2269 km² and 16,720 km² for the 2 years, respectively. In the latter case, this corresponds to 38% of the entire bottom area and, in fact, most of the Danish straits except the Kattegat would be hypoxic below the pycnocline located at 15–20 m depth. In a normal year, 5–10% of the bottom area exposed to hypoxia under present conditions will increase to 10–15% for the projected temperature increase. Overall, the area of hypoxia will approximately double with a temperature increase of 4°C.

Future perspectives

Hypoxia is becoming an increasingly widespread phenomenon throughout the world (Díaz, 2001), and evidence for eutrophication as an important causal factor is increasing (Scavia et al., 2003; Hagy et al., 2004; Conley et al., 2007). Despite nutrient reductions in some areas affected by hypoxia (e.g., Carstensen et al., 2006), there has still been no reported recovery from hypoxia in coastal ecosystems. This does not mean that nutrient reductions are

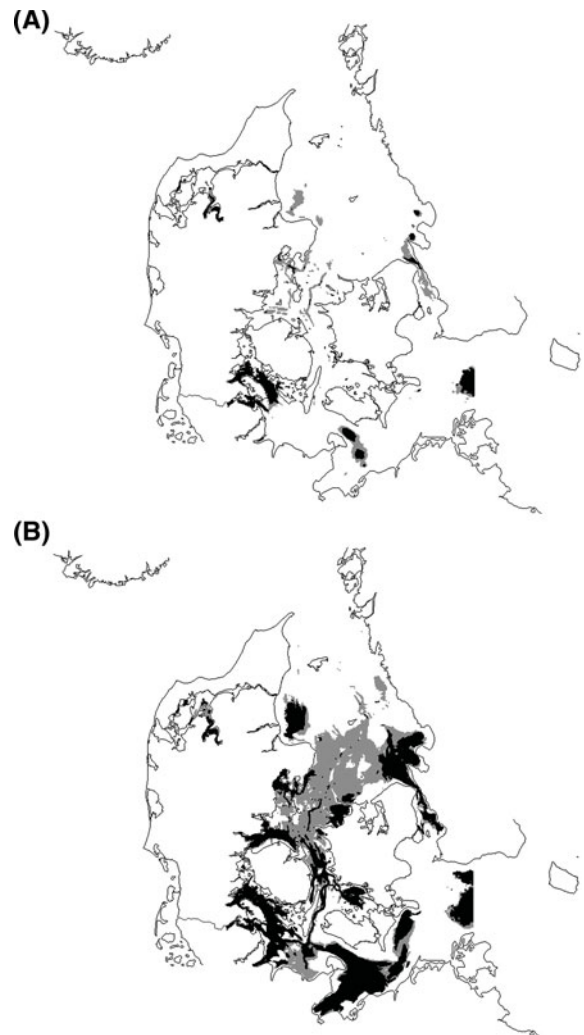


Fig. 5 Areas exposed to hypoxic conditions at present (*black shading*) and projected for a 4°C temperature increase (*grey shading*) in the best year, 1998 (A), and in the worst year, 2002 (B)

not important for improving oxygen concentrations; hypoxia would have been even worse if nutrient inputs had not been reduced. Nutrient reductions may to some extent have been counteracted by temperature increases and regime shifts in the sequestering of organic material, making hypoxia a self-sustaining process. Failure to reduce nutrient inputs may lead to cascading effects of increasing hypoxia with the return to a desired status becoming more and more unlikely, and nutrient reduction is the only realistic management measure for improving oxygen conditions. Consequently, hypoxia can only be remedied by reducing nutrient inputs to levels that will allow

for recolonization of macrozoobenthos and a balance between nutrient input and removal rates. The sooner significant reductions are implemented, the more likely is the recovery. Finally, monitoring of oxygen conditions and nutrient management responses are important for coastal ecosystems showing early signs of beginning hypoxia. Nutrient reductions required to restore coastal ecosystem health may be larger if thresholds of hypoxia are exceeded.

References

- Alexander, R. B., R. A. Smith, G. E. Schwarz, E. W. Boyer, J. W. Nolan & J. W. Brakebill, 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River basin. *Environmental Science & Technology* 42: 822–830.
- Carstensen, J., D. J. Conley, J. H. Andersen & G. Ærtebjerg, 2006. Coastal eutrophication and trend reversal: a Danish case study. *Limnology & Oceanography* 51: 398–408.
- Christensen, J. H., B. Hewitson, A. Busuioac, A. Chen, X. Gao, I. Held, R. Jones, R. K. Kolli, W. T. Kwon, R. Laprise, V. Magaña Rueda, L. Mearns, C. G. Menéndez, J. Räisänen, A. Rinke, A. Sarr & P. Whetton, 2007. Regional climate projections. In Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor & H. L. Miller (eds), *Climate Change 2007: The Physical Science Basis*. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge: 847–940.
- Collie, J. S., K. Richardson & J. H. Steele, 2004. Regime shifts: can ecological theory illuminate the mechanisms? *Progress in Oceanography* 60: 281–302.
- Conley, D. J., 1999. Biogeochemical nutrient cycles and nutrient management strategies. *Hydrobiologia* 410: 87–96.
- Conley, D. J., C. Humborg, L. Rahm, O. P. Savchuk & F. Wulff, 2002. Hypoxia in the Baltic Sea and basin-scale changes in phosphorus biogeochemistry. *Environmental Science & Technology* 36: 5315–5320.
- Conley, D. J., J. Carstensen, G. Ærtebjerg, P. B. Christensen, T. Dalsgaard, J. L. S. Hansen & A. B. Josefson, 2007. Long-term changes and impacts of hypoxia in Danish coastal waters. *Ecological Applications* 17: S165–S184.
- Dalsgaard, T., D. E. Canfield, J. Petersen, B. Thamdrup & J. Acuña-Gonzalez, 2003. N₂ production by the anammox reaction in the anoxic water column of Golfo Dulce, Costa Rica. *Nature* 422: 606–608.
- Deutsch, C., J. L. Sarmiento, D. M. Sigman, N. Gruber & J. P. Dunne, 2007. Spatial coupling of nitrogen inputs and losses in the ocean. *Nature* 445: 163–167.
- Díaz, R. J., 2001. Overview of hypoxia around the world. *Journal of Environmental Quality* 30: 275–281.
- Díaz, R. J. & R. Rosenberg, 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: Annual Review* 33: 245–303.
- Díaz, R. J. & R. Rosenberg, 2008. Spreading dead zone and consequences for marine ecosystems. *Science* 321: 926–929.
- Duarte, C. M., D. J. Conley, J. Carstensen & M. Sánchez-Camacho, 2009. Return to Neverland: shifting baselines affect ecosystem restoration targets. *Estuaries & Coasts* 32: 29–36.
- Eyre, B. D. & A. J. P. Ferguson, 2009. Denitrification efficiency for defining critical loads of carbon in shallow coastal ecosystems. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9765-1.
- Gombay, E. & L. Horvath, 1996. On the rate of approximations for maximum likelihood tests in change-point models. *Journal of Multivariate Analysis* 56: 120–152.
- Gray, J. S., R. S.-s Wu & Y. Y. Or, 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series* 238: 249–279.
- Groffman, P., M. A. Altabet, J. K. Böhlke, K. Butterbach-Bahl, M. B. David, M. K. Firestone, A. E. Giblin, T. M. Kana, L. P. Nielsen & M. A. Voytek, 2006. Methods for measuring denitrification: diverse approaches to a difficult problem. *Ecological Applications* 9: 1–13.
- Hagy, J. D., W. R. Boynton, C. W. Keefe & K. V. Wood, 2004. Hypoxia in Chesapeake Bay, 1950–2001: long-term change in relation to nutrient loading and river flow. *Estuaries* 27: 634–658.
- Hannig, M., G. Lavik, M. M. M. Kuypers, D. Woebken, W. Martens-Habbena & K. Jürgens, 2007. Shift from denitrification to anammox after inflow events in the central Baltic Sea. *Limnology & Oceanography* 52: 1336–1345.
- Hansen, J. L. S. & J. Bendtsen (2006). Climatic induced effects on marine ecosystems. National Environmental Research Institute, Roskilde, Denmark. Technical Report No. 598 (in Danish). <http://technical-reports.dmu.dk>.
- Harris, L. A., C. M. Duarte & S. Nixon, 2006. Allometric laws and prediction in coastal and estuarine ecology. *Estuaries* 29: 343–347.
- HELCOM, 2003. The 2002 oxygen depletion event in the Kattegat, Belt Sea and Western Baltic. *Baltic Sea Environment Proceedings* No. 90. Helsinki Commission for Protection of the Baltic Sea.
- Holmer, M., C. M. Duarte & N. Marbá, 2003. Sulfur cycling and seagrass (*Posidonia oceanica*) status in carbonate sediments. *Biogeochemistry* 66: 223–239.
- Jensen, H. S., P. B. Mortensen, F. Ø. Andersen, E. Rasmussen & A. Jensen, 1995. Phosphorus cycling in a coastal marine sediment, Aarhus Bay, Denmark. *Limnology & Oceanography* 40: 908–917.
- Karlson, K., R. Rosenberg & E. Bonsdorff, 2002. Temporal and spatial large-scale effects of eutrophication and oxygen deficiency on benthic fauna in Scandinavian and Baltic waters—A review. *Oceanography and Marine Biology Annual Review* 40: 427–489.
- Kemp, W. M., P. Sampou, J. Caffrey & M. Mayer, 1990. Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnology & Oceanography* 35: 1545–1563.
- Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M.

- Glibert, J. D., Hagy, L. W., Harding, E. D., Houde, D. G., Kimmel, W. D., Miller, R. I. E., Newell, M. R., Roman, E. M., Smith & J. C. Stevenson, 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303: 1–29.
- Mee, L., 2006. Reviving Dead Zones. *Scientific American*. pp. 79–86.
- Mortimer, C. H., 1941. The exchange of dissolved substances between mud and water. *Journal of Ecology* 29: 280–329.
- Muradian, R., 2001. Ecological thresholds: a survey. *Ecological Economics* 38: 7–24.
- Nixon, S. W., J. R. Kelly, B. N. Furnas, C. A. Oviatt & S. S. Hale, 1980. Phosphorus regeneration and the metabolism of coastal marine bottom communities. In K. R. Tenore & B. C. Coull (eds), *Marine Benthic Dynamics*. University of South Carolina Press, USA: 219–242.
- Oguz, T. & D. Gilbert, 2007. Abrupt transitions of the top-down controlled Black Sea pelagic ecosystem during 1960–2000: evidence for regime-shifts under strong fishery exploitation and nutrient enrichment modulated by climate-induced variations. *Deep Sea Research Part I* 54: 220–242.
- Österblom, H., S. Hansen, U. Larsson, O. Hjerne, F. Wulff, R. Elmgren & C. Folke, 2007. Human-induced trophic cascades and ecological regime shifts in the Baltic Sea. *Ecosystems* doi:10.1007/s10021-007-9069-0.
- Rabalais, N. N., R. E. Turner & W. J. Wiseman Jr., 2002. Gulf of Mexico hypoxia, A.K.A. ‘The Dead Zone’. *Annual Review of Ecology and Systematics* 33: 235–263.
- Rask, N., S. T. Petersen & M. H. Jensen, 1999. Responses to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. *Hydrobiologia* 393: 69–81.
- Roeckner, E., L. Bengtsson & J. Feichter, 1999. Transient climate change simulations with a coupled atmosphere-ocean GCM including the tropospheric sulfur cycle. *Journal of Climate* 12: 3004–3032.
- Scavia, D., N. N. Rabalais, R. E. Turner, D. Justic & W. Wiseman Jr., 2003. Predicting the response of Gulf of Mexico hypoxia to variations in Mississippi River nitrogen load. *Limnology & Oceanography* 48: 951–956.
- Scavia, D., E. A. Kelly & J. D. Hagy III, 2006. A simple model for forecasting the effects of nitrogen loads on Chesapeake Bay hypoxia. *Estuaries and Coasts* 29: 674–684.
- Scheffer, M., S. Carpenter, J. A. Foley, C. Folke & B. Walker, 2001. Catastrophic shifts in ecosystems. *Nature* 413: 591–596.
- Seitzinger, S. P. & A. E. Giblin, 1996. Estimating denitrification in North Atlantic continental shelf sediments. *Biogeochemistry* 35: 235–260.
- Sloth, N. P., H. Blackburn, L. S. Hansen, N. Risgaard-Petersen & B. A. Lomstein, 1995. Nitrogen cycling in sediments with different organic loading. *Marine Ecology Progress Series* 116: 163–170.
- Smith, S. V. & J. T. Hollibaugh, 1989. Carbon-controlled nitrogen cycling in a marine ‘macrocosm’: an ecosystem-scale model for managing cultural eutrophication. *Marine Ecology Progress Series* 52: 103–109.
- Smith, D. E., M. Leffler & G. Mackiernan, 1992. Oxygen dynamics in the Chesapeake Bay. A synthesis of recent research. University of Maryland, Maryland Sea Grant.
- Soetaert, K. & J. J. Middelburg, 2009. Modeling eutrophication and oligotrophication of shallow-water marine systems: the importance of sediments under stratified and well mixed conditions. *Hydrobiologia* (this issue). doi: 10.1007/s10750-009-9777-x.
- Turner, R. E., N. N. Rabalais & D. Justic, 2008. Gulf of Mexico hypoxia: alternative states and a legacy. *Environmental Science & Technology* 42: 2323–2327.
- Vahtera, E., D. J. Conley, B. Gustaffson, H. Kuosa, H. Pitkanen, O. Savchuk, T. Tamminen, N. Wasmund, M. Viitalo, M. Voss & F. Wulff, 2007. Internal ecosystem feedbacks enhance nitrogen-fixing cyanobacteria blooms and complicate management in the Baltic Sea. *Ambio* 36: 186–194.
- Vaquer-Sunyer, R. & C. M. Duarte, 2008. Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of Science* 105: 15452–15457.
- Webster, I. T. & G. P. Harris, 2004. Anthropogenic impacts on the ecosystems of coastal lagoons: modeling fundamental biological processes and management implications. *Marine and Freshwater Research* 55: 67–78.

Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management

D. L. Breitburg · J. K. Craig · R. S. Fulford · K. A. Rose · W. R. Boynton ·
D. C. Brady · B. J. Ciotti · R. J. Diaz · K. D. Friedland · J. D. Hagy III ·
D. R. Hart · A. H. Hines · E. D. Houde · S. E. Kolesar · S. W. Nixon ·
J. A. Rice · D. H. Secor · T. E. Targett

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 31–47.
DOI: 10.1007/s10750-009-9762-4 © Springer Science+Business Media B.V. 2009

Abstract Both fisheries exploitation and increased nutrient loadings strongly affect fish and shellfish abundance and production in estuaries. These stressors do not act independently; instead, they jointly influence food webs, and each affects the sensitivity of species and ecosystems to the other. Nutrient enrichment and the habitat degradation it sometimes causes can affect sustainable yields of fisheries, and fisheries exploitation can affect the ability of estuarine systems to process nutrients. The total biomass

of fisheries landings in estuaries and semi-enclosed seas tends to increase with nitrogen loadings in spite of hypoxia, but hypoxia and other negative effects of nutrient over-enrichment cause declines in individual species and in parts of systems most severely affected. More thoroughly integrated management of nutrients and fisheries will permit more effective management responses to systems affected by both stressors, including the application of fisheries regulations to rebuild stocks negatively affected by eutrophication. Reducing fishing mortality may lead to the recovery of depressed populations even when eutrophication contributes to population declines if actions are taken while the population retains sufficient reproductive potential. New advances in modeling, statistics, and technology promise to provide the information needed to improve the understanding and management of systems subject to both nutrient enrichment and fisheries exploitation.

Electronic supplementary material The online version of this article (doi:10.1007/s10750-009-9762-4) contains supplementary material, which is available to authorized users.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

D. L. Breitburg (✉) · R. S. Fulford · A. H. Hines ·
S. E. Kolesar
Smithsonian Environmental Research Center,
P.O. Box 28, Edgewater, MD 21037, USA
e-mail: breitburgd@si.edu

J. K. Craig
North Carolina State University,
303 College Circle, Morehead City, NC 28557, USA

J. K. Craig
Florida State University Coastal and Marine Laboratory,
3618 Highway 98, St. Teresa, FL 32358-2702, USA

R. S. Fulford
Department of Coastal Sciences, University of Southern
Mississippi, Gulf Coast Research Laboratory,
703 East Beach Road, Ocean Springs, MS 39566, USA

K. A. Rose
Department of Oceanography and Coastal Sciences,
Louisiana State University, Baton Rouge, LA 70803, USA

W. R. Boynton · E. D. Houde · D. H. Secor
Chesapeake Biological Laboratory, University
of Maryland Center for Environmental Science,
P.O. Box 38, Solomons, MD 20688, USA

Keywords Eutrophication · Hypoxia · Fisheries · Estuary · Management

Introduction

Estuaries are crossroads where land meets the sea, and where human influences on the terrestrial landscape and on aquatic food webs intersect. Both nutrient over-enrichment (Nixon & Buckley, 2002; Seitzinger et al., 2002) and declining populations of species targeted by fisheries (Jackson et al., 2001; Lotze et al., 2006) are common consequences of human influence in these coastal systems. Estuaries are particular targets for these and other stressors because of their proximity to population centers and because of their historical importance as sources of protein and as places for disposal of human waste (Kennish, 2002).

The term eutrophication encompasses a variety of ecological changes that occur when nutrient-stimulated primary production exceeds removal through biotic and physical processes (NRC, 2000). In estuaries, these changes are associated with or caused by an increase in the supply of organic carbon (Nixon, 1995) and include increased biomasses of phytoplankton, macroalgae, and epiphytes; increased occurrences of toxic and nuisance algal blooms; decreased dissolved oxygen concentrations (hypoxia and anoxia); and decreased abundance of submerged

vascular plants (Cloern, 2001; Rabalais & Turner, 2001). More than two-thirds of the 122 U.S. estuaries evaluated in the National Eutrophication Assessment were rated as moderately or highly eutrophic due to nutrient over-enrichment (Bricker et al., 1999). Worldwide, nitrogen loadings resulting from human activities, as well as the number of estuaries and coastal seas reporting low dissolved oxygen concentrations have dramatically increased since the 1950s (Diaz, 2001; Boesch, 2002; Seitzinger et al., 2002; Diaz and Rosenberg, 2008; Fig. 1a).

In many of the same estuaries, but often over longer periods of time, fishing has contributed to declining abundances of species that spend all or part of their life cycle in estuaries (e.g., Secor & Waldman, 1999; Lotze et al., 2006). The historical progression of fishery expansion and collapse follows the temporal pattern of improvements in fishing technology (Pitcher, 2001) and spatial patterns of human population densities (Barrett et al., 2004; Kirby, 2004; Lotze et al., 2006).

The combined effect of eutrophication and fisheries exploitation is to simultaneously increase algal production, degrade habitat, and remove fish and shellfish biomass. In this review, we ask how nutrient enrichment affects fish (including mobile mollusks and crustaceans throughout this article unless otherwise noted) abundances and fisheries, how fisheries affect population responses to nutrient enrichment,

D. C. Brady · B. J. Ciotti · T. E. Targett
College of Marine and Earth Studies, University of Delaware, Lewes, DE 19958, USA

Present Address:

D. C. Brady
Department of Civil and Environmental Engineering,
University of Delaware, 301 DuPont Hall, Newark,
DE 19716, USA

R. J. Diaz
College of William and Mary, Virginia Institute of Marine
Science, Gloucester Point, VA 23062, USA

K. D. Friedland
National Marine Fisheries Service, 28 Tarzwell Dr.,
Narragansett, RI 02882, USA

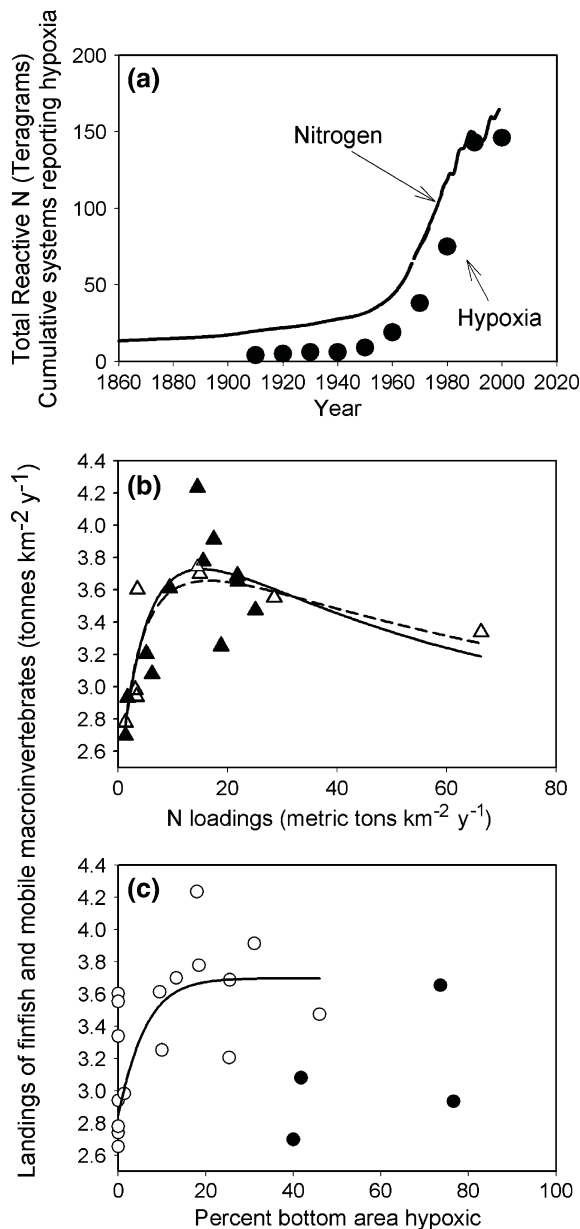
J. D. Hagy III
US Environmental Protection Agency, National Health
and Environmental Effects Research Laboratory, Gulf
Ecology Division, 1 Sabine Island Drive, Gulf Breeze,
FL 32561, USA

D. R. Hart
Northeast Fisheries Science Center, National Marine
Fisheries Service, 166 Water St., Woods Hole, MA 02543,
USA

S. E. Kolesar
St. Mary's College of Maryland, 18952 E. Fisher Road,
St. Mary's City, MD 20686, USA

S. W. Nixon
Graduate School of Oceanography, University of Rhode
Island, South Ferry Rd., Narragansett, RI 02882, USA

J. A. Rice
Biology Department, North Carolina State University,
Box 7617, Raleigh, NC 27695, USA



and how the co-occurrence of eutrophication and fisheries exploitation influences management options. We include a specific consideration of hypoxia, which is often caused or exacerbated by high loads of anthropogenic nutrients. Finally, we discuss recent advances in analysis and technology that may help predict the separate and combined effects of eutrophication and fisheries exploitation as well as outcomes of alternate management actions.

Several recent articles have addressed the effects of nutrient enrichment and fisheries removals in

◀ **Fig. 1** Relationships among nitrogen loadings, hypoxia, and fisheries landings in estuaries. **a** Both anthropogenic loadings of nutrients to estuaries (adapted from Boesch, 2002) and the number of estuaries reporting hypoxia (from Diaz, 2001) have increased greatly since the mid-twentieth century. **b** Relationship between N loadings and fisheries landings. *Filled triangle* = systems with $\geq 10\%$ of area hypoxic (dissolved oxygen concentration $< 3 \text{ mg l}^{-1}$); *open triangle* = systems with $< 10\%$ bottom area hypoxic; *grey triangle* = Chesapeake Bay landings excluding Atlantic menhaden (*Brevoortia tyrannus*). The *solid line* is a four-parameter lognormal curve (SigmaPlot 9.0) including all landings of mobile species from Chesapeake Bay (highest landings in figure; adjusted $R^2 = 0.72$). The *dashed line* excludes Atlantic menhaden landings from Chesapeake Bay to make the data more comparable to other U.S. estuaries that do not have purse-seine fisheries ($R^2 = 0.76$). Systems plotted from lowest to highest N loadings: Azov Sea, Gulf of Bothnia, Black Sea, Adriatic Sea, Irish Sea, North Sea, Gulf of Finland, Baltic Proper (including Gulf of Riga), Kattegat/Skagerrak, Chesapeake Bay, Tampa Bay, Northern Gulf of Mexico, Long Island Sound, Corpus Christi Bay, Neuse River Estuary, Sea of Marmara, Danish Straits, Delaware Bay, and Galveston Bay. **c** Positive relationship between the spatial extent of hypoxia (dissolved oxygen concentration $< 3 \text{ mg l}^{-1}$) and fisheries landings in estuaries without persistent anoxic basins or widespread release of sulfides. *Open circle* = systems included in regression in the order of extent of hypoxia: 10 systems with $\ll 1\%$ regularly occurring hypoxia (Aegean, Gulf of Bothnia, Balearic, Ionian, Irish, Levant, Sardinian, North Sea, Delaware Bay, and Galveston Bay), the Adriatic Sea, Kattegat/Skagerrak, Corpus Christi Bay, Chesapeake Bay, Northwest Gulf of Mexico, Gulf of Finland, Neuse River Estuary, and Long Island Sound, Danish Straits; *filled circle* = systems with persistent anoxic or severely hypoxic basins or widespread release of sulfides from lowest to highest spatial extent of hypoxia: the Sea of Azov, Baltic Proper, Sea of Marmara, and Black Sea. See supplementary material—Appendix—for the full list of references used for this figure

coastal systems (e.g., Diaz & Rosenberg, 1995, 2008; Peterson et al., 2000; Jackson et al., 2001; Nixon & Buckley, 2002; Lotze et al., 2006). The present synthesis differs by focusing on the potential interaction of these two stressors, and how the co-occurrence of nutrient enrichment and fisheries exploitation affects management options. We also emphasize eutrophication effects on finfish and mobile macroinvertebrates, which unlike sessile species have the potential to behaviorally avoid habitat patches negatively impacted by eutrophication. As a result of this behavioral flexibility, the responses of mobile species to eutrophication can be fundamentally different, and interactions with fisheries potentially more complex, than those of sessile species. Furthermore, unlike marine mammals and reptiles

whose recovery is discussed by Lotze et al. (2006), fish and mobile macroinvertebrates generally continue to be managed for their harvest potential, rather than as species deserving of protection primarily because of intrinsic or ecological value. Our focus is on estuaries and semi-enclosed seas, although many of the issues we raise also apply to other aquatic systems.

Interactions between nutrient enrichment and fishing effects

Although seemingly disparate stressors, nutrient enrichment and fisheries removals are tightly coupled both ecologically and in management forums. Sustainable levels of fisheries removals are dependent on rates of production and mortality—both of which can be affected by the ecological changes associated with eutrophication (Fig. 2). Fisheries removals can affect both system and species' sensitivity to nutrient enrichment by altering the trophic structure, as well as the size and age distributions of key consumers (Pauly et al., 1998; Vallin & Nissling, 2000; Szmant, 2002; Birkeland & Dayton, 2005). Both nutrient enrichment and fisheries exploitation can cause similar changes in food webs, such as increased

dominance of pelagic planktivores including gelatinous zooplankton (Caddy, 2000; de Leiva Moreno et al., 2000; Oguz, 2005). Since these stressors typically co-occur, mitigating management actions can be delayed or rendered less effective, as various stakeholders ascribe responsibility for declining fish landings to different causes and promote solutions that shift the greatest burden to other interest groups.

Nutrient enrichment effects on fish and fisheries

Production and mortality

Both cross-system comparisons and temporal trends indicate that nutrient enrichment in estuaries and semi-enclosed seas is associated with increases in total fisheries landings across a broad range of nutrient loading rates (Nixon, 1988; Nixon & Buckley, 2002; Fig. 1b). In much the same way that agricultural production increases with fertilizer use, fish landings are generally higher in systems with high nutrient loadings (Nixon & Buckley, 2002). This pattern suggests that nutrient enrichment typically increases fish growth, reproduction, or survival. Temporal patterns of fisheries landings and fish growth support this contention; in systems such as the Nile, Seto Inland Sea, North Sea, and until

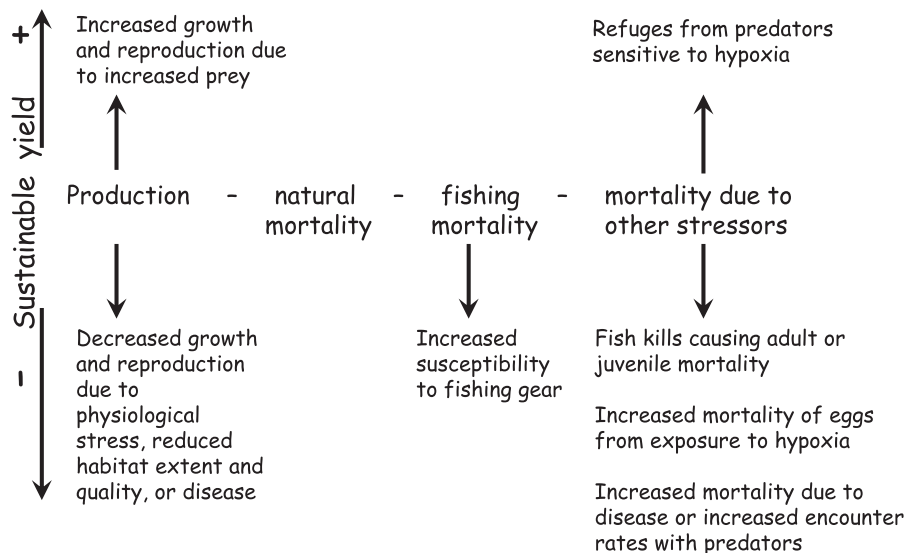


Fig. 2 Conceptual model of the relationship between eutrophication and sustainable fisheries yields. Nutrient enrichment can either increase (*upward arrows*) or decrease (*downward arrows*) sustainable yields of exploited species by affecting growth, reproduction, and fishing mortality as well as mortality

due to other interacting stressors. The effect of nutrient enrichment on sustainable yields of fish species will depend on both the severity of eutrophication and whether critical habitat is negatively affected

recently, the Kattegat–Skagerrak, landings have increased and decreased in concert with temporal patterns of nutrient loadings (Nixon & Buckley, 2002; Nagai, 2003; Oczkowski & Nixon, 2008). These results are consistent with theoretical food-web models that indicate that a number of processes common in estuarine food webs such as cannibalism, density-dependent predator mortality, intraguild predation, grazer-resistance of algae, and predator-dependent functional responses tend to increase bottom-up effects (McCauly et al., 1988; Gatto, 1991; Ginzburg & Akçakaya 1992; McCann et al., 1998; Hart, 2002). Based on analyses of a large number of coastal systems (Fig. 1b; Caddy, 2000; Nixon & Buckley, 2002), fish data that represent a range of trophic levels, and a consideration of biomass removal by fisheries, we concur with the assessment by Nixon & Buckley (2002) that nutrient enrichment of estuaries generally increases fish biomass, a conclusion that differs from that of Micheli (1999).

High yields of fishable biomass characterize most highly nutrient-enriched estuaries in spite of the potential negative effects of nutrient over-enrichment (Fig. 1b, c). We compared nitrogen loadings (watershed, upstream, and direct atmospheric deposition), the spatial extent of hypoxia (percent of bottom area with <3 mg dissolved oxygen l^{-1}), and landings of finfish and mobile invertebrates (collectively referred to as “fish”) in 19 estuaries and semi-enclosed seas in Europe and the United States, and hypoxic extent and fisheries landings in five additional European systems (see supplementary material—Appendix). Estuaries and semi-enclosed seas analyzed span more than an order of magnitude in N loadings ($N \text{ km}^{-2} \text{ year}^{-1}$) and two orders of magnitude in surface area. For the 13 systems with N loading rates <18 tonnes $N \text{ km}^{-2} \text{ year}^{-1}$, fisheries landings (i.e., total biomass of landings of mobile species $\text{km}^{-2} \text{ year}^{-1}$) increased linearly with N loadings (equation: landings = $0.39 \times N$ loadings + 0.27; $R^2 = 0.87$; $P < 0.0001$) (Fig. 1b). Fisheries landings are not a perfect reflection of abundances and are affected by fishing regulations, historical overfishing, and economic and social influences. Nevertheless, fish production sets an upper bound on sustainable fisheries landings.

It is difficult to find compelling evidence for negative effects of hypoxia on fisheries for mobile

species, even in systems with extensive and persistent oxygen depletion, *if system-wide conditions and total landings are the focus of the analyses*. The relationship between fisheries landings and the spatial extent of hypoxia is positive (although we certainly do not imply causality) if systems with persistent anoxic or severely hypoxic basins (the Baltic Sea proper, and the Black Sea, and the Sea of Marmara) or widespread release of sulfides (the Azov Sea) are excluded (Fig. 1c; quadratic regression: $R^2 = 0.50$, $n = 19$; SigmaPlot 9.0). Extensive hypoxia or anoxia, low fisheries landings, and clearly demonstrated effects of oxygen depletion on individual species targeted by fisheries in the Baltic, Black, and Azov Seas (Nissling et al., 1994; Caddy, 2000; Debol'skaya et al., 2005; Oguz, 2005) would seem to suggest that low landings in these three systems are caused by hypoxia or co-occurring negative effects of eutrophication. However, the Baltic, Black, and Azov Seas have low N loading rates (Fig. 1b). The relationship between total fish landings and N loadings in these systems, and in systems with $\geq 10\%$ bottom area hypoxia, in general, is not different from that in estuaries and semi-enclosed seas with no or minimal hypoxia (Fig. 1b and regression of systems with N loadings of <18 tonnes $\text{km}^{-2} \text{ year}^{-1}$ presented above).

The two systems in our dataset with the highest N loading rates, and that fell on the declining section of the relationship between N loadings and fisheries landings, were Delaware and Galveston Bays—systems with no or minimal ($\ll 1\%$ total area) hypoxia. It is not clear whether landings in these systems are directly attributable to eutrophication and represent the declining productivity that Caddy (1993) predicted for highly enriched systems even though they lack severe oxygen depletion, or whether low fisheries landings reflect peculiarities of the systems or their fisheries. Four of the systems that fell on the declining slope of the curve relating N loadings and landings (in the order of increasing nitrogen loads: Corpus Christi Bay, the Neuse River, Delaware Bay, and Galveston Bay; Fig. 1b) are the smallest estuaries included in our analyses; small size may affect their propensity toward historical overfishing or other characteristics of their fisheries or fish production. These systems also lack purse-seine fisheries for Atlantic menhaden, which comprise $>50\%$ of landings of mobile species by weight in Chesapeake Bay.

The positive relationship between nutrient enrichment and fisheries yields measured at large spatial scales depends on an integrative, system-level view of the consequence of nutrient flow through marine food webs. Such an approach provides valuable information about the effects of nutrient enrichment on fisheries and provides a first-order estimate of the relationship between nutrients and fish production. However, calculations of system-wide landings involve spatial and temporal averaging by both researchers and the fish, themselves, that can mask habitat-, species-, and season-specific negative responses of fish and mobile macroinvertebrates to eutrophication.

In the majority of highly nutrient-enriched systems, eutrophication creates a spatial and temporal mosaic of habitats that vary in physiological suitability, species composition, spatial overlap of predators and prey, and prey abundance (Pihl et al., 1995; Breitburg, 2002; Craig & Crowder, 2005). High levels of nutrient enrichment can simultaneously stimulate prey production, creating areas of less favorable or completely unsuitable habitat by causing

low dissolved oxygen concentrations, increasing macroalgal growth, and leading to the decline or disappearance of submerged macrophytes. In most systems, low dissolved oxygen concentrations are spatially and temporally limited, developing seasonally, and primarily affecting sub-pycnocline waters.

As a result of this spatial matrix of prey-enhanced and degraded habitats, eutrophication, in general, and hypoxia, in particular, fundamentally alter the way mobile organisms utilize space and move through systems. The net effect of eutrophication on fish and mobile macroinvertebrates is dependent on both these altered movement patterns and the limitations to movement (Breitburg, 2002). Effects on fish and fisheries depend on the way both eutrophication and co-occurring stressors alter the spatial matrix, the way that individual species requirements intersect with those spatial effects, and the way that fisheries practices intersect with both physiological and behavioral responses to this spatial mosaic (Fig. 3).

Spatial averaging of fish abundance or landings statistics can either accentuate or mask negative effects of nutrient over-enrichment depending on the

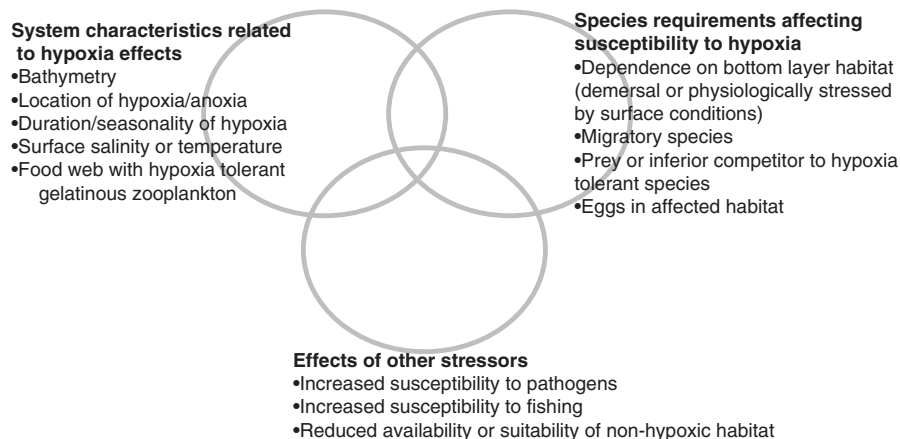


Fig. 3 Effects of nutrient enrichment on fish and fisheries depend on the intersection of habitat characteristics, species characteristics, and the co-occurrence of other stressors. System characteristics that modulate the effects of hypoxia include factors that influence the duration, timing, and location of hypoxia, the extent of shallow highly oxygenated waters, the suitability of highly oxygenated waters, and food web characteristics that may allow energy to be shunted into gelatinous zooplankton rather than fished species. Species' susceptibility to hypoxia is influenced by dependence on bottom layer habitat, migration patterns, whether the species is a prey or competitor of other species that are more tolerant of

low oxygen, and whether eggs are deposited into or sink into hypoxic bottom waters. Hypoxia fundamentally alters the movement of individuals through the system, so species characteristics that limit the ability to appropriately respond to hypoxia by altering movement and space-use patterns can be especially important. The presence of co-occurring stressors that are worsened by hypoxia, or whose importance increases because of species responses to hypoxia, can increase the likelihood of population or fisheries declines. Stressors that reduce the suitability of habitat that would otherwise be a refuge from hypoxia may lead to population declines

scale and location of areas averaged. Behavior-induced averaging of spatial patterns of habitat quality by fish can be achieved through foraging and habitat-use behaviors. Opportunistic and adaptive behaviors of mobile species can reduce the potential negative effects of eutrophication-degraded habitat patches (Pihl et al., 1992; Brandt & Mason, 1994) if the scale and speed of response exceeds that at which habitat suitability declines (Breitburg, 1992). Avoidance of low dissolved oxygen concentrations and preferential use of habitat with high prey abundances provide the potential for growth rates to more strongly reflect conditions in favorable habitat patches than in unfavorable areas, although density-dependent interactions (Eby et al., 2005; Eggleston et al., 2005; Craig et al., 2007) and limits to behavioral responses (Stierhoff et al., 2006, in press; Tyler & Targett, 2007) affect the balance between positive and negative effects of nutrient enrichment. Over time, growth may also integrate temporal patterns in the positive effects of increased prey and negative effects of physiological stress and spatially limited reductions in prey abundance.

Negative effects of nutrient enrichment on fish and fisheries primarily reflect species-specific characteristics and requirements, and spatial processes that affect the location and timing of oxygen depletion. This difference between ecosystem and behavioral/population perspectives is an important consideration in the development of a framework for understanding and managing eutrophication effects. Some species that have supported commercial, recreational, or subsistence fisheries are especially susceptible because of their behaviors, life history characteristics, or physiological tolerances. Sewage and nutrient discharge patterns can determine whether those species' characteristics result in population declines.

On a local scale (i.e., within a portion of an interconnected estuarine system rather than system wide), hypoxia can sharply reduce both fish abundances and landings. Descriptions of physical conditions and biota in river segments with anoxic or severely hypoxic waters virtually throughout the water column prior to modern sewage treatment suggest that the release of raw sewage into tributaries may create the most severe impacts on fish and fisheries. Low dissolved oxygen is thought to have blocked the migration of anadromous fishes and contributed to the decline of the American shad

(*Alosa sapidissima*) in the Delaware River Estuary before advanced sewage treatment was implemented in the city of Philadelphia (Weisberg et al., 1996, Albert, 1988). Release of raw sewage also led to widespread anoxia and loss of fish from large areas of both the Mersey (Jones, 2006) and Thames (Tinsley, 1998) Rivers prior to improved sewage treatment and reduced biological oxygen demand (BOD) of discharges. Entire year classes of smelt (*Osmerus eperlanus*) in the Elbe were lost in years when the timing and extent of low oxygen coincided with larval use of the affected habitat (Thiel et al., 1995). These cases are characterized by severe reductions in total fish biomass and fisheries at the local scale (i.e., within sub-estuaries rather than throughout the entire estuarine systems), and reductions in abundance and landings of severely affected species at a larger spatial scale.

Oxygen depletion, along with other negative effects of nutrient over-enrichment, can sometimes affect recruitment and cause extensive mortality in mainstem areas of estuaries and semi-enclosed seas as well. For example, in the Baltic Sea, Baltic cod (*Gadus morhua*) eggs sink into bottom waters with lethally low dissolved oxygen concentrations, and temporal variation in this species' recruitment has been linked to variation in the depth of hypoxic waters in deep Baltic basins (Köster et al., 2005). Hypoxia causes mortality of adult Norway lobsters (*Nephrops norvegicus*) and may be especially problematic for juveniles, which require higher dissolved oxygen concentrations for survival as well as to dig or search for burrows (Eriksson & Baden, 1997). Stimulation of macroalgae by high nutrient loadings can exacerbate problems of hypoxia by further depressing oxygen concentrations (Deegan, 2002), producing exudates that reduce survival and growth of larvae and small juveniles (Larson, 1997), and reducing the extent of preferred unvegetated habitat (Pihl et al., 2005). Under such conditions, decreasing the nutrient loading sufficiently to increase dissolved oxygen concentrations and reducing macroalgae may increase the production of affected species.

Since hypoxia primarily affects the bottom layer of the water column, demersal, epibenthic, and infaunal species tend to be more severely impacted than species that inhabit the upper water column (Caddy, 2000). Species composition of fish landings can change, and the economic value of fisheries (per unit

biomass) can decline with increases in the ratio of pelagic to demersal species. Hypoxia thus reinforces patterns created by fisheries removals in which groundfish stocks tend to be depleted before stocks of small pelagic species. Avoidance of oxygen concentrations that lead to mortality or reduced growth can reduce the potential negative effects of habitat degradation due to nutrient over-enrichment (Breitburg, 2002; Tyler & Targett, 2007). Even so, both decreased growth and reproductive impairment have been reported in the field at the local scale and in small and shallow systems (Wu et al., 2003; Stierhoff et al., in press). Decreases in demersal species and increases in pelagic planktivores caused by nutrient enrichment may, however, have important economic and social consequences even when they occur on a local rather than a system-wide scale if historical fishing grounds are affected, and the consequence is a spatial separation of high-density population centers and fishing communities.

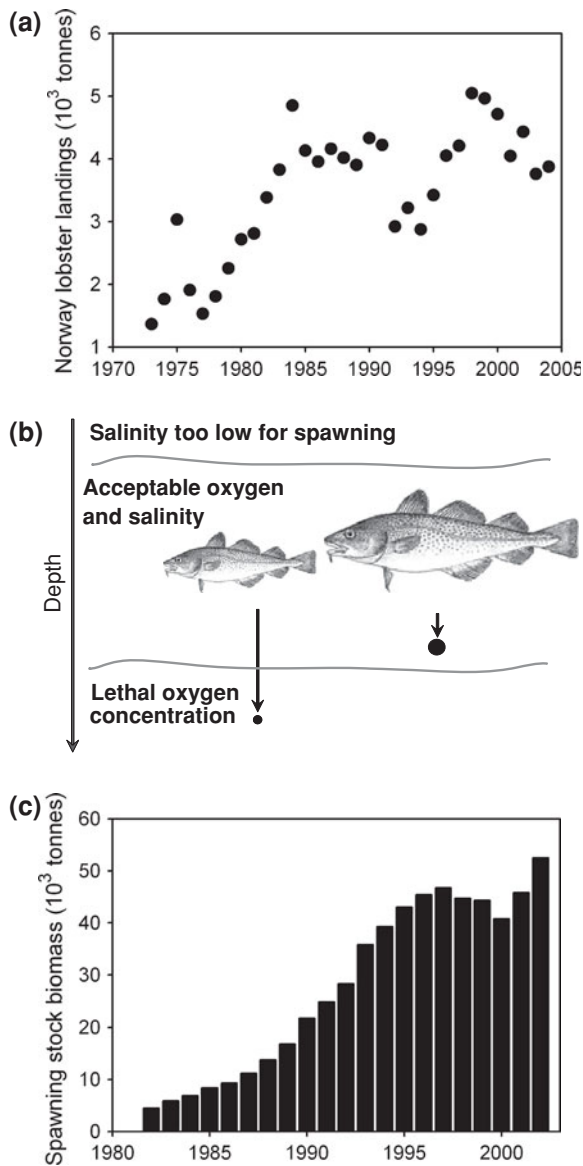
Sustainable fisheries landings and fisheries practices

Nutrient enrichment may buffer systems against fisheries removals by increasing fish growth, survival, and reproduction, thereby allowing for higher sustainable harvests. However, high landings can encourage fishing overcapitalization that can lead to overfishing, especially if nutrient loads are later reduced. The decline in anchovy (*Engraulis japonicus*) landings in the Seto Inland Sea that followed reductions in phosphorus loadings (Nagai, 2003) may have been the result of continued fishing pressure on a fish population with decreased anthropogenic nutrient-supplemented production.

Nutrient enrichment can also reduce sustainable harvests by reducing the growth, survival, and reproduction of targeted species where negative effects of eutrophication dominate (Fig. 2). Rates of fisheries removals that are sustainable in the absence of eutrophication-related reductions in production or survival may lead to overfishing and population declines in their presence. For example, macroalgal growth in nursery grounds of juvenile plaice (*Pleuronectes platessa*) in the Kattegat–Skagerrak may reduce recruitment to the fishery, and thus, slow the recovery of fish stocks even if fishing pressure is reduced or removed (Pihl et al., 2005). Eutrophication also can exacerbate effects of other stressors,

resulting in greater than additive cumulative effects of multiple stressors, and reducing sustainable rates of fishing mortality more than either stressor alone (Breitburg & Riedel, 2005). Although this article focuses on mobile species, one of the clearest examples of a eutrophication–disease–fishery interaction is for a sessile bivalve. The eastern oyster (*Crassostrea virginica*) has declined in many Atlantic coast estuaries in the United States as a result of both overfishing and disease (NRC, 2004). Sublethal hypoxia can compromise the immune response of adult oysters and greatly increase mortality caused by *Perkinsus marinus*, the protistan parasite that causes Dermo (Anderson et al., 1998). As a result, sustainable rates of oyster harvests are likely to be reduced in areas in which *P. marinus* is common, if nutrient enrichment leads to even moderate levels of hypoxia.

Behavioral avoidance of hypoxia can increase vulnerability to fishing gear, make fisheries more efficient, and result in increased catches. A well-documented example is the Norway lobster fishery in the Kattegat. Catch per unit effort peaked in 1982 as worsening hypoxia induced lobsters to leave their burrows, making them more accessible to capture (Baden et al., 1990). Fishers also adapt methods to local nutrient-related conditions, deploying gear in oxygenated refuge areas and along the edge of hypoxic zones where fish and mobile invertebrates aggregate. This practice may be widespread and has been noted in fisheries for crabs in the Neuse River Estuary (Selberg et al., 2001), flatfish in Osaka Bay (Jun Shoji, Hiroshima University, pers. comm.), and possibly for brown shrimp (*Farfantepenaeus aztecus*) (Craig & Crowder, 2005) and menhaden (*Brevoortia patronus*) (Smith, 2001) in the Gulf of Mexico. However, we could find no documented cases with strong evidence of population declines resulting from hypoxia-related increases in vulnerability to fishing practices or gear. Even landings of Norway lobster in the Kattegat–Skagerrak system have remained at near peak levels more than 20 years after hypoxia-related mortality was first identified (Fig. 4a), although this could reflect changes in fishing locations and increased fishing effort (L. Pihl, Göteborg University, pers. comm.). Nevertheless, hypoxia-induced increases in sensitivity to harvest might keep landings high even if populations are declining—a situation that could lead to delays in the response of management.



Fishing effects on species and system responses to nutrient enrichment

Although eutrophication can strongly affect sustainable fisheries yields, the large number of estuarine species that have experienced reduced stock sizes as a result of fisheries exploitation (e.g., Jackson et al., 2001; Lotze et al., 2006) indicate that fishing is a more frequent direct cause of finfish and shellfish population declines in estuaries than eutrophication. Fishing can be a large source of direct mortality, while growing evidence suggests that direct mortality of juveniles

and adults from hypoxia (i.e., fish kills), and other consequences of eutrophication (e.g., harmful algal blooms; HABs) is typically a relatively small proportion of total mortality. The primary effect of fishing is the removal of biomass, often of late juveniles or adults that have high reproductive value, and shifts in size structure to smaller body size. These effects may drive strong declines in abundance and, if they exceed the compensatory reserve of the population, decreases in population growth rates. In contrast, eutrophication can result in both increases and decreases in growth and survival over different spatial and temporal scales and life history stages, and it is the integration of these positive and negative effects that determines the net effect on population growth rates.

◀ **Fig. 4** Strong interactions between fisheries mortality and eutrophication occur because of effects of eutrophication on behaviors, effects of fisheries removals on population size structure, and because of the magnitude of mortality caused by fisheries exploitation. **a** Behavior: Norway lobster. Behavioral responses to hypoxia can increase the susceptibility of organisms to fishing gear, and at least initially result in increased landings. Catch per unit effort of Norway lobsters in the Kattegat along the coast of Sweden peaked in the mid-1980s, as worsening hypoxia induced lobsters to leave their burrows, making them more accessible to nets (Baden et al., 1990). Nevertheless, landings of Norway lobster in the Kattegat–Skagerrak area have remained high. **b** Size distributions: Baltic cod. Fisheries regulations can indirectly influence the susceptibility of cod eggs to hypoxia-induced mortality by influencing the size of spawning females in the population. Large females produce large eggs that are sufficiently buoyant to be retained in oxygenated mid-depth waters; smaller females produce small eggs that sink and perish (Vallin & Nissling, 2000). Cod image <http://stellwagen.noaa.gov/visit/welcome.html>. **c** Reduced fishing mortality: Striped bass. Decisive management action taken to protect spawning stock biomass of striped bass in Chesapeake Bay is often cited as a successful example of fisheries management. Stringent fishing regulations allowed the population to rebound even though eutrophication and its potential to negatively affect striped bass growth and habitat persisted

Fishing can, however, affect both system and species' sensitivities to nutrient enrichment. Slow-growing species, older and larger individuals, and top predators are often disproportionately impacted by fishing (Pauly et al., 1998; Jackson et al., 2001; Pitcher, 2001). Fisheries removal of piscivorous species can favor increases in short-lived mid-trophic level species that are more directly coupled, and likely to be more responsive, to nutrient enrichment (Oguz, 2005). Selective removal of species has the potential to simplify food webs, thereby increasing

responsiveness to top-down control (including fishing) (Hart, 2002). A mechanism has also been identified by which fishing can increase egg mortality. A downward shift in female size distribution caused by size-selective fisheries removals can increase hypoxia-related mortality of Baltic cod eggs by affecting egg size and buoyancy (Vallin & Nissling, 2000; Fig. 4b). The importance of this mechanism to other species is unknown.

Fishing can also reduce the ability of estuarine systems to absorb increases in pelagic primary production by selectively removing herbivorous species. Some of the most important fisheries in temperate estuaries target suspension feeders, whose combined removal can have a direct effect on phytoplankton prey. In the United States, menhaden (*Brevoortia tyrannus* and *B. patronus*) and bivalves such as oysters and clams are among the most important estuarine fisheries by weight and dollar value, respectively (Houde & Rutherford, 1993). Removal of suspension feeders from temperate estuaries is thought to have reduced the capacity of these systems to process nutrients (Newell, 1988, 2004) in much the same way that reduced populations of herbivorous fishes and sea urchins sensitize coral reefs to nutrient effects (Szmant, 2002), although the effect of any single suspension-feeding species will be limited by its spatial distribution and feeding ecology (e.g., Pomeroy et al., 2006; Fulford et al., 2007). Harvesting suspension-feeder biomass has also been suggested as a tool to remove the consumed excess primary production and buffer coastal systems against nutrient enrichment effects. Calculations suggest, however, that biomass removal of wild populations by fishing is not a large enough factor, *at least at sustainable removal levels*, to affect the severity of eutrophication (Boynton et al., 1995).

Effects of fisheries on eutrophication, and of eutrophication on fish yields, may be strongest in systems with extensive aquaculture production. Excess feed and feces associated with intensive cage culture of fish can cause localized hypoxia and harmful algal blooms, resulting in losses of both cultured and wild fish production (Rosenthal, 1985; Naylor et al., 2000). Bivalve aquaculture, in contrast, removes phytoplankton and can sometimes reduce negative effects of nutrient enrichment. Removal of farmed suspension-feeder biomass may have a greater potential than fisheries exploitation of wild

populations to reduce eutrophication because high rates of aquaculture removals need not cause the population declines and fisheries collapses characteristic of wild fish fisheries. Diversion of human sewage to aquaculture operations, as in the Kulti Estuary in India, can also simultaneously increase fish production and improve water quality. Aquaculture operations may, however, be particularly susceptible to mortality resulting from negative effects of eutrophication such as harmful algal blooms. *Chattonella* spp. blooms killed about 14 million yellowtail (*Seriola quinqueradiata*) in the eastern Seto Inland Sea in 1972, resulting in a financial loss equivalent to US\$ 106 million in 2004 dollars (Anderson et al., 2001).

Restoring and managing fishes in estuaries and semi-enclosed seas

Better coordinated and joint management of water quality and fisheries is important not only because the effects of fisheries removals and nutrient enrichment are tightly linked, but also because tools associated with the management of one stressor may help ameliorate problems caused by the other. Reducing fishing mortality can be a rapid, effective tool for increasing depleted populations even where eutrophication contributes to population decline. The most rapid route to the recovery of depressed populations is often through increased survival to reproductive sizes or ages and reduced mortality of reproductive individuals. Where fishing mortality contributes to low reproductive potential of the population, both results may be rapidly accomplished through appropriate fishing regulations if they are adopted and implemented in a timely manner.

Recent experience with striped bass (*Morone saxatilis*) management in Chesapeake Bay, USA, suggests that reducing fishing mortality can allow populations to recover even in systems where nutrient enrichment reduces the suitability and extent of habitat by increasing hypoxia (summarized from Richards & Rago, 1999; Fig. 4c). Both fisheries landings and numbers of juvenile striped bass declined during the 1970s and early 1980s. A number of factors—including low dissolved oxygen, contaminants, and acidic precipitation, as well as overfishing—were suggested as causes of reduced growth and survival. Nevertheless, management agencies

responded to declining striped bass numbers by taking the most direct and immediate action available—reducing fishing mortality. In 1981, the Atlantic States Marine Fisheries Commission (ASMFC) adopted the Interstate Fisheries Management Plan for the Striped Bass, which called for coast-wide size limits and spawning area closures; this was adopted by most states during 1981–1984. Maryland and Delaware declared moratoria on striped bass fishing and other jurisdictions imposed high minimum size limits in 1985; Virginia imposed a moratorium in 1989. Strong year classes, indicating high levels of successful reproduction, occurred in 1989 and 1993. In spite of the resumption of limited fishing in 1990, striped bass continued to rebound. The Chesapeake striped bass stock was declared fully recovered in 1995, and remains high today although water quality problems that potentially reduce striped bass growth and habitat quality persist. This example indicates the potential for reduced fishing mortality to allow fished stocks to increase even when multiple stressors affect the targeted population, and for rapid implementation of fisheries restrictions to be an effective management action while research is conducted to sort out the relative contribution of potential causes of an observed population decline.

The question of how to balance or coordinate nutrient and fisheries management to prevent further system degradation and to meet restoration goals is complicated by fundamental differences in nutrient and fisheries management, as well as the need to better predict outcomes of strategies for coordinated management of multiple stressors. Nutrient management is challenged with a spatial and functional mismatch between the source of the problem and its effects that is not typical in fisheries management. Individuals and businesses that generate nutrients are often separated from the water bodies those nutrients affect by meaningful distances. They often do not utilize, and are not financially dependent on, those water bodies. In contrast, fishers depend on bodies of water in which they fish, and are affected financially if targeted stocks are depleted. There are also large differences in costs to taxpayers of meeting nutrient and fisheries management goals. For example, the Chesapeake Bay Commission (2003) recently estimated the cost of water quality protection and restoration in the Bay at nearly US\$ 11.5 billion. Implementing fishing regulations can be far less expensive.

There can also be large differences between nutrient and fisheries management in the time scales between the identification of the problem, implementation of management actions, and system response. Regulations to reduce fishing can, in theory, be rapidly implemented while reductions of nitrogen require longer time periods, as sewage treatment plants are upgraded, riparian buffers are restored, and farming practices are changed. Less than 15 years elapsed between the initiation of fishery management planning and the full recovery of the Chesapeake Bay striped bass stock (Richards & Rago, 1999), as compared to a 45-year projected time frame for achieving water quality goals in the Chesapeake (Richard Batiuk, Chesapeake Bay Program, pers. comm.). In practice, however, there are many examples of long delays in implementing fisheries management strategies. The rapid recovery of striped bass in Chesapeake Bay required that fishing mortality be greatly reduced, while the population still had the reproductive capacity to rebound rapidly (Secor, 2000).

There has generally been an institutional disconnect between fisheries and water quality management despite their shared goals. In the United States, responsibility for water quality and fisheries management rests with different governmental agencies even though water quality criteria are set to protect fish populations and key habitats, and despite an increasing emphasis in fisheries management on habitat considerations and ecosystem-based management. The efficacy of separate management of nutrients and fisheries is compromised because each management branch (i.e., fisheries versus water quality) works within a shared landscape that is constantly changed by the other. Nutrient loading can affect sustainable yields of fisheries, and fishery removals and aquaculture can affect the ability of ecosystems to process anthropogenic nutrient loadings and can directly affect water quality. In addition, public and political support for nutrient management can be influenced by perceived effects of water quality conditions on fisheries and aquaculture.

New tools and approaches

Accelerating progress is being made toward the ultimate goal of ecosystem-level understanding and quantitative modeling (water quality to fish) that goes

beyond correlative approaches. While immediate management remedies to water quality and fisheries issues need not be based on complete mechanistic understanding, long-term sustainability requires the identification of the causes and interplay between water quality and fish community dynamics, and mechanistic understanding greatly aids in garnering stakeholder support for management actions. New methods for data collection and modeling are enabling increased understanding of nutrient and food-web dynamics in estuaries, and will lead to increased capabilities for quantifying the interactions between eutrophication and fisheries. These new data collection and modeling methods promise to meet three critical needs for improved understanding and management of systems stressed by both nutrient enrichment and fisheries exploitation: (1) the ability to scale up from effects measurable on a local or individual scale to population, fisheries, and whole-ecosystem responses in order to better understand the effects of eutrophication on fished populations and vulnerability to overfishing; (2) the ability to track active movement and transport of all life stages to evaluate environmental and fisheries risks; and (3) the ability to better predict outcomes of alternative management actions for species and systems exposed to both nutrient enrichment and fisheries exploitation. The citations below represent only a sampling of promising approaches.

Advances in water quality monitoring have refined our measurements and expanded the temporal and spatial scales of observations. We are improving our ability to make synoptic system-wide measurements of water quality conditions via remote sensing (Walker & Rabalais, 2006). Continuous monitoring of physicochemical conditions via ocean observing systems is being developed for many locations (Ocean US, 2002), and fishing effort can now be tracked via vessel monitoring systems (Gallaway et al., 2003; Murawski et al., 2005).

Improved techniques have refined growth and mortality rate estimates and are providing new information on fine-scale spatial distributions and behavior (Rose et al., 2001). Acoustic methods produce three-dimensional views of fish distributions that can be overlain onto fine-scale spatial maps of environmental variables (Brandt & Mason, 1994; Roy et al., 2004). Otolith ageing has provided a new level of resolution to our ability to determine causes

of fluctuations in year-class strength of finfish populations (Limburg et al., 1999). New methods for marking individual fish and microchemical analysis of skeletal material have refined estimates of mortality and movement (Secor et al., 1995), and ultrasonic telemetry now allows for almost continuous tracking of the movements of individuals through time and space (Szedlmayer, 1997; Bell et al., 2003). Molecular and genetic-based analysis provides information on the degree of mixing of populations (Carvalho & Hauser, 1994; Manel et al., 2003). We can monitor the physiology, behavior, growth, and energetic status of free-ranging animals on increasingly fine temporal and spatial scales through a variety of methods (Cooke et al., 2004; Makris et al., 2006), including RNA:DNA ratios (Stierhoff et al., in press) and implantable sensors that directly detect oxygen concentrations encountered (Svendsen et al., 2006).

Theoretical advances have occurred in our conceptual thinking about ecosystems and in the resolution and melding of water quality and fish models. Shifts in our conceptual view of estuarine ecosystems include the recognition of the importance of the microbial food web and sediments to nutrient recycling (Cornwell et al., 1999; Cloern, 2001), how indirect effects that are not simply bottom-up or top-down can affect ecosystems (Cury et al., 2000), and the increasing understanding of how regional and global climate patterns affect local conditions (Hoffman & Powell, 1998). Two seemingly large shifts in our theoretical basis for viewing ecosystems have been the shift from equilibrium analysis to dynamic and stochastic analysis (Turchin, 1995), and viewing ecosystems as complex systems (Bascompte & Sole, 1995). The shift from equilibrium to dynamic systems began several decades ago, while the complexity theory view is just beginning to enter the mainstream thinking. Complexity theory suggests that the non-linear responses of systems to perturbations and change, which keep on surprising us (Paine et al., 1998), are in fact expected responses from a complex system and that local small-scale interactions lead to many of the macroscopic properties we attribute to communities and ecosystems.

A broader array of statistical and analytical methods is becoming available to researchers (Venables & Dichmont, 2004; Hobbs & Hillborn, 2006). Numerical models of water quality, hydrodynamics, and lower trophic-level food webs have advanced

dramatically and are being developed at a resolution not imagined computationally possible a few years ago (e.g., McGillicuddy et al., 2003). Such detail is becoming important to capture the complicated hydrodynamics and nutrient and plankton dynamics typical of coastal ecosystems. At the same time, fish models have been progressing with finer spatial resolution and the tracking of individuals (DeAngelis & Mooij, 2005). Individual-based modeling nicely accommodates the spatial mosaic imposed by variation in water quality and habitat. The mass-balance trophic network model, Ecopath with Ecosim, can be used to examine the relationship between primary production and fisheries production (Christensen and Walters, 2004), although it does not easily accommodate the effects of hypoxia on foraging. Water quality models that start at the base of the food web and fish models that start near the top of the food web are slowly but steadily converging (Runge et al., 2004; Rose et al., 2008). In addition, it will become increasingly important to address processes not typical for separate applications of water quality and fisheries models, such as the movement of adult fish within a high-resolution three-dimensional unstructured grid. Just over the horizon are models that include humans as fully dynamic components (Massey et al., 2006).

Conclusions

Estuaries continue to provide valued ecosystem services such as nutrient processing and protein production even when they are challenged by moderate levels of nutrient additions and fisheries removals. Even in severely degraded systems, reductions in nutrient loading lead to improvements in water quality and increases in fish populations negatively affected by nutrient enrichment (Albert, 1988; Thomas, 1998; Mee et al., 2005; Jones, 2006). Similarly, reductions in fishing mortality can lead to increased populations where suitable habitat and reproductive potential in the population remain sufficient (Richards & Rago, 1999).

This resilience of estuaries may result from their basic nature. Estuaries are spatially heterogeneous, open systems that exhibit dynamic within-year and among-year variability. Nutrients are flushed out of the system, and migratory species are replenished

from broadly distributed coastal populations. In the vast majority of estuaries, hypoxia is disrupted annually or more frequently by seasonal destratification and wind mixing of the water column. Estuarine organisms are adapted to these dynamic conditions, are generally plastic in their tolerances and behaviors, and typically have high egg production rates that facilitate population increase under favorable conditions. Resilience of estuaries may be facilitated by limits on the magnitude of fishing and eutrophication effects on fish and mobile invertebrate species; although there are exceptions (e.g., sturgeons: Niklitschek & Secor, 2005), neither stressor typically results in complete extirpations of these species throughout interconnected estuarine systems. More closed systems, including some semi-enclosed seas included in our analysis and discussion, lack some of the properties that impart resilience in estuaries connected to oceans or other large water bodies.

Of particular concern are co-occurring stressors, or levels of fisheries removals and nutrient loadings that lead to threshold responses that are resistant to remediation through fisheries and nutrient management (Scheffer et al., 2001; Breitburg & Riedel, 2005). These threshold responses may be particularly likely where multiple stressors reinforce the direction of ecosystem change and reduce the likelihood that a system can retrace the same trajectory during restoration as during degradation. Since eutrophication and fisheries exploitation co-occur, there is the potential for eutrophication and fishing to interact, each affecting the way populations and estuarine ecosystems respond to the other stressor and to management actions. Parallels between agricultural systems and nutrient-enriched estuaries may include reduced stability and heightened susceptibility to natural and anthropogenic stressors. Determining the potential for nutrient enrichment to affect the susceptibility of species and systems to overfishing, the spatial dislocation of fisheries and fishing communities, and the economic value of fisheries is critically important.

We raise the issue of cumulative effects of eutrophication and fisheries not only because of the increased understanding of the potential interactions among these stressors, but also because we now have measurement and modeling techniques to better analyze their individual and simultaneous effects on coastal systems. Joint or coordinated management of

nutrients and fisheries is warranted by the intertwined effects of these two stressors as well as because we are well along the way to developing the means to improve the management of both water quality and fisheries by managing them jointly.

Acknowledgments This manuscript is the result of a workshop held in 2005 at the Smithsonian Environmental Research Center and a keynote address by D. Breitburg at the Eutro2006 Symposium in Denmark. We thank NOAA-Center for Sponsored Coastal Ocean Research and the Smithsonian Marine Science Network for workshop funding, and the organizers of Eutro2006 for the wide-range sharing of ideas they fostered. We would especially like to thank L. Pihl, S. Hannsen, and S. Baden for sharing their insight on the Baltic, Kattegat, and Skagerrak systems, D. Boesch for providing raw data for Fig. 1a, and L. Davias and D. Hondorp for help in identifying and analyzing data on nutrient loadings, the spatial extent of hypoxia, and fisheries landings. We also thank many persons listed in the supplementary material—Appendix—who provided data or access to data summarized in Fig. 1b and c.

References

- Albert, R. C., 1988. The historical context of water quality management for the Delaware Estuary. *Estuaries* 11: 99–107.
- Anderson, R. S., L. L. Brubacher, L. M. Ragone Calvo, E. M. Bureson & M. A. Unger, 1998. Effect of in vitro exposure to tributyltin and hypoxia on the progression of *Perkinsus marinus* infections and host defense mechanism in oysters, *Crassostrea virginica* (Gmelin). *Journal of Fish Diseases* 21: 371–380.
- Anderson, D. M., P. Andersen & V. M. Bricelj, 2001. Monitoring and management strategies for harmful algal blooms in coastal waters, APEC #201-MR-01.1. Asia Pacific Economic Program, Singapore, and Intergovernmental Oceanographic Commission Technical Series No. 59.
- Baden, S. P., L. Pihl & R. Rosenberg, 1990. Effects of oxygen depletion on the ecology, blood physiology and fishery of the Norway lobster (*Nephrops norvegicus* L.). *Marine Ecology Progress Series* 67: 141–155.
- Barrett, J. H., A. M. Locker & C. M. Roberts, 2004. The origins of intensive marine fishing in Medieval Europe: the English evidence. *Proceedings of the Royal Society of London* 271: 2417–2421.
- Bascompte, J. & R. V. Sole, 1995. Rethinking complexity: modeling spatio-temporal dynamics in ecology. *Trends in Ecology & Evolution* 10: 361–366.
- Bell, G. W., D. B. Eggseton & T. G. Wolcott, 2003. Behavioral responses of free-ranging blue crabs to episodic hypoxia. II. Feeding. *Marine Ecology Progress Series* 259: 227–235.
- Birkeland, C. & P. K. Dayton, 2005. The importance in fishery management of leaving the big ones. *Trends in Ecology & Evolution* 20: 356–358.
- Boesch, D. R., 2002. Challenges and opportunities for science in reducing nutrient over-enrichment of coastal ecosystems. *Estuaries* 25: 886–900.
- Boynton, W. R., J. J. Garber, R. Summers & W. M. Kemp, 1995. Inputs, transformations and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18: 285–314.
- Brandt, S. B. & D. M. Mason, 1994. Landscape approaches for assessing spatial patterns in fish foraging and growth. In Stouder, D. J., K. L. Fresh & R. J. Feller (eds), *Theory and Application of Fish Feeding Ecology*. University of South Carolina Press, Columbia, SC: 211–240.
- Breitburg, D. L., 1992. Episodic hypoxia in the Chesapeake Bay: interacting effects of recruitment, behavior and a physical disturbance. *Ecological Monographs* 62: 525–546.
- Breitburg, D. L., 2002. Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries. *Estuaries* 25: 767–781.
- Breitburg, D. L. & G. F. Riedel, 2005. Multiple stressors in marine systems. In Norse, E. A. & L. B. Crowder (eds), *Marine Conservation Biology: The Science of Maintaining the Sea's Biodiversity*. Island Press, Washington, DC: 167–182.
- Bricker, S. B., C. G. Clement, D. E. Pirhalla, S. P. Orlando & D. R. G. Farrow, 1999. National Estuarine Eutrophication Assessment. Effects of Nutrient Enrichment in the Nation's Estuaries. National Ocean Service, Special Projects Office and National Centers for Coastal Ocean Science, Silver Spring, MD.
- Caddy, J. F., 1993. Toward a comparative evaluation of human impacts on fishery ecosystems of enclosed and semi-enclosed seas. *Reviews in Fisheries Science* 1: 57–95.
- Caddy, J. F., 2000. Marine catchment basin effects versus impacts of fisheries on semi-enclosed seas. *ICES Journal of Marine Science* 57: 628–640.
- Carvalho, G. R. & L. Hauser, 1994. Molecular genetics and the stock concept in fisheries. *Reviews in Fish Biology and Fisheries* 4: 326–350.
- Chesapeake Bay Commission, 2003. *The Cost of a Clean Bay. Assessing Funding Needs throughout the Watershed*. Chesapeake Bay Commission, Annapolis, MD.
- Christensen, V. & C. J. Walters, 2004. Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling* 172: 109–139.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Cooke, S. J., S. G. Hinch, M. Wikelski, R. D. Andrews, L. J. Kuchel, T. G. Wolcott & P. J. Butler, 2004. Biotelemetry: a mechanistic approach to ecology. *Trends in Ecology & Evolution* 19: 334–343.
- Cornwell, J. C., W. M. Kemp & T. M. Kana, 1999. Denitrification in coastal ecosystems: methods, environmental controls, and ecosystems level controls, a review. *Aquatic Ecology* 33: 41–54.
- Craig, J. K. & L. B. Crowder, 2005. Hypoxia-induced habitat shifts and energetic consequences in Atlantic croaker and brown shrimp on the Gulf of Mexico shelf. *Marine Ecology Progress Series* 294: 79–94.
- Craig, J. K., J. A. Rice, L. B. Crowder & D. A. Nadeau, 2007. Density-dependent growth and survival in juvenile estuarine fishes: an experimental approach with spot (*Leiostomus xanthurus*). *Marine Ecology Progress Series* 343: 251–262.

- Cury, P., A. Bakun, R. J. M. Crawford, A. Jarre, R. A. Quinones, L. J. Shannon & H. M. Verheye, 2000. Small pelagics in upwelling systems: patterns of interactions and structural changes in 'wasp-waist' ecosystems. *ICES Journal of Marine Science* 57: 603–618.
- DeAngelis, D. L. & W. M. Mooij, 2005. Individual-based modeling of ecological and evolutionary processes. *Annual Review of Ecology, Evolution & Systematics* 36: 147–168.
- Debol'skaya, E. I., E. I. Yakushev & A. I. Sukhinov, 2005. Formation of fish kills and anaerobic conditions in the Sea of Azov. *Water Resources* 32: 151–162.
- Deegan, L. A., 2002. Lessons learned: the effects of nutrient enrichment on the support of nekton by seagrass and salt marsh ecosystems. *Estuaries* 25: 727–742.
- de Leiva Moreno, J. I., V. N. Agostini, J. F. Caddy & F. Carocci, 2000. Is the pelagic-demersal ratio from fishery landings a useful proxy for nutrient availability? A preliminary data exploration for the semi-enclosed seas around Europe. *ICES Journal of Marine Science* 57: 1091–1102.
- Diaz, R. J., 2001. Overview of hypoxia around the world. *Journal of Environmental Quality* 30: 275–281.
- Diaz, R. J. & R. Rosenberg, 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology Annual Review* 33: 245–303.
- Diaz, R. J. & R. Rosenberg, 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321: 926–929.
- Eby, L. A., L. B. Crowder, C. M. McClellan, C. H. Peterson & M. J. Powers, 2005. Habitat degradation from intermittent hypoxia: impacts on demersal fishes. *Marine Ecology Progress Series* 291: 249–262.
- Eggleston, D. B., G. W. Bell & A. D. Amavisca, 2005. Interactive effects of episodic hypoxia and cannibalism on juvenile blue crab mortality. *Journal of Experimental Marine Biology and Ecology* 325: 18–26.
- Eriksson, S. P. & S. P. Baden, 1997. Behaviour and tolerance to hypoxia in juvenile Norway lobster (*Nephrops norvegicus*) of different ages. *Marine Biology* 128: 49–54.
- Fulford, R. S., D. L. Breitburg, R. I. E. Newell, W. M. Kemp & M. Luckenbach, 2007. Effects of oyster population restoration strategies on phytoplankton biomass in Chesapeake Bay: a flexible modeling approach. *Marine Ecology Progress Series* 336: 43–61.
- Gallaway, B. J., J. G. Cole & L. R. Martin, 2003. An evaluation of an electronic logbook as a more accurate method of estimating spatial patterns of trawling effort and bycatch in the Gulf of Mexico shrimp fishery. *North American Journal of Fisheries Management* 23: 787–809.
- Gatto, M., 1991. Some remarks on models of plankton densities in lakes. *American Naturalist* 137: 264–267.
- Ginzburg, L. & H. Akçakaya, 1992. Consequences of ratio-dependent predation for steady state properties of ecosystems. *Ecology* 73: 1536–1543.
- Hart, D. R., 2002. Intraguild predation, invertebrate predators, and trophic cascades in lake food webs. *Journal of Theoretical Biology* 218: 111–128.
- Hobbs, N. T. & R. Hilborn, 2006. Deepening ecological insight using contemporary statistics. *Ecological Applications* 16: 3–4.
- Hoffman, E. E. & T. M. Powell, 1998. Environmental variability effects on marine fisheries: four case histories. *Ecological Applications* 8: S23–S32.
- Houde, E. D. & E. S. Rutherford, 1993. Recent trends in estuarine fisheries: predictions of fish production and yield. *Estuaries* 16: 161–176.
- Jackson, J. B. C., M. X. Kirby, W. H. Berger et al., 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629–638.
- Jones, P. D., 2006. Water quality and fisheries in the Mersey estuary, England: a historical perspective. *Marine Pollution Bulletin* 53: 144–154.
- Kennish, M. J., 2002. Environmental threats and environmental future of estuaries. *Environmental Conservation* 29: 78–107.
- Kirby, M. X., 2004. Fishing down the coast: historical expansion and collapse of oyster fisheries along continental margins. *Proceedings of the National Academy of Science* 101: 13096–13099.
- Köster, F. W., C. Möllmann, H.-H. Hinrichsen, J. Tomkiewicz, K. Wieland, G. Kraus, R. Voss, B. R. MacKenzie, D. Schnack, A. Makarchouk, M. Plikshs & J. E. Beyer, 2005. Baltic cod recruitment—the impact of climate variability on key processes. *ICES Journal of Marine Science* 62: 1408–1425.
- Larson, F., 1997. Survival and growth of plaice (*Pleuronectes platessa*) larvae and juveniles in mats of *Enteromorpha* sp. The effects of algal exudates and nocturnal hypoxia. M.S. Thesis. Göteborg University, Sweden.
- Limburg, K. E., M. L. Pace & K. K. Arend, 1999. Growth, mortality, and recruitment of larval *Morone* spp. in relation to food availability and temperatures in the Hudson River. *Fishery Bulletin* 97: 80–91.
- Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson & J. B. C. Jackson, 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 313: 1806–1809.
- Makris, N. C., P. Ratilal, D. T. Symonds, S. Jagannathan, S. Lee & R. W. Nero, 2006. Fish population and behavior revealed by instantaneous continental shelf-scale imaging. *Science* 311: 660–663.
- Manel, S., M. K. Schwartz, G. Luikart & P. Taberlet, 2003. Landscape genetics: combining landscape ecology and population genetics. *Trends in Ecology & Evolution* 18: 189–197.
- Massey, D. M., S. C. Newbold & B. Gentner, 2006. Valuing water quality changes using a bioeconomic model of a coastal recreational fishery. *Journal of Environmental Economics and Management* 52: 482–500.
- McCann, K. S., A. Hastings & D. R. Strong, 1998. Trophic cascades and trophic trickles in pelagic food webs. *Proceedings of the Royal Society of London B* 265: 205–209.
- McCauly, E., W. M. Murdoch & S. Watson, 1988. Simple models and variation in plankton densities among lakes. *American Naturalist* 132: 382–403.
- McGillicuddy, D. J., L. A. Anderson, S. C. Doney & M. E. Maltrud, 2003. Eddy-driven sources and sinks of nutrients in the upper ocean: results from a 0.1° resolution model of the North Atlantic. *Global Biogeochemical Cycles* 17: 1–20.

- Mee, L. D., J. Friedrich & M. T. Gomoiu, 2005. Restoring the Black Sea in times of uncertainty. *Oceanography* 18: 100–111.
- Micheli, F., 1999. Eutrophication, fisheries, and consumer-resource dynamics in marine pelagic ecosystems. *Science* 285: 1396–1398.
- Murawski, S. A., S. E. Wigley, M. J. Fogarty, P. J. Rago & D. G. Mountain, 2005. Effort distribution and catch patterns adjacent to temperate MPAs. *ICES Journal of Marine Science* 62: 1150–1167.
- Nagai, T., 2003. Recovery of fish stocks in the Seto Inland Sea. *Marine Pollution Bulletin* 47: 126–131.
- Naylor, R. L., R. J. Goldburd, J. H. Primavera, N. Kautsky, M. C. M. Beveridge, J. Clay, C. Folke, J. Lubchenco, H. Mooney & M. Troell, 2000. Effect of aquaculture on world fish supplies. *Nature* 405: 1017–1024.
- Newell, R. I. E., 1988. Ecological changes in Chesapeake Bay: are they the result of overharvesting the American Oyster, *Crassostrea virginica*? Understanding the estuary: advances in Chesapeake Bay research. Chesapeake Research Consortium: 536–546.
- Newell, R. I. E., 2004. Ecosystem influences of natural and cultivated populations of suspension-feeding bivalve molluscs: a review. *Journal of Shellfish Research* 23: 51–61.
- Niklitschek, E. J. & D. H. Secor, 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. *Estuarine and Coastal Shelf Science* 64: 135–148.
- Nissling, A., H. Kryvi & L. Vallin, 1994. Variation in egg buoyancy of Baltic cod *Gadus morhua* and its implications for egg survival in prevailing conditions in the Baltic Sea. *Marine Ecology Progress Series* 110: 67–74.
- Nixon, S. W., 1988. Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnology and Oceanography* 33: 1005–1025.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes and future concerns. *Ophelia* 41: 199–219.
- Nixon, S. W. & B. A. Buckley, 2002. 'A strikingly rich zone'—nutrient enrichment and secondary production in coastal marine ecosystems. *Estuaries* 25: 782–796.
- NRC, 2000. Clean Coastal Waters—Understanding and Reducing the Effects of Nutrient Pollution. National Research Council. National Academy Press, Washington, DC.
- NRC, 2004. Nonnative Oysters in the Chesapeake Bay. National Research Council. National Academy Press, Washington, DC.
- Ocean US, 2002. Building Consensus: Toward an Integrated and Sustained Ocean Observing System (IOOS). Ocean US, Arlington, VA.
- Oczkowski, A. & S. Nixon, 2008. Nutrient over-enrichment and the rise and fall of a coastal fishery; a review of data from the Nile Delta, Egypt. *Estuarine, Coastal and Shelf Science* 77: 309–319.
- Oguz, T., 2005. Long-term impacts of anthropogenic forcing on the Black Sea ecosystem. *Oceanography* 18: 112–121.
- Paine, R. T., M. J. Tegner & E. A. Johnson, 1998. Compounded perturbations yield ecological surprises. *Ecosystems* 1: 535–545.
- Pauly, D., V. Christensen, J. Dalsgaard, R. Froese & F. Torres Jr., 1998. Fishing down marine food webs. *Science* 279: 860–863.
- Peterson, C. H., H. C. Summerson, E. Thomson, H. S. Lenihan, J. Grabowski, L. Manning, F. Micheli & G. Johnson, 2000. Synthesis of linkages between benthic and fish communities as a key to protecting essential fish habitat. *Bulletin of Marine Science* 66: 759–774.
- Pihl, L., S. P. Baden, R. J. Diaz & L. C. Schaffner, 1992. Hypoxia-induced structural changes in the diet of bottom-feeding fish and crustacean. *Marine Biology* 112: 349–361.
- Pihl, L., I. Isaksson, H. Wennhage & P. O. Moksnes, 1995. Recent increase of filamentous algae in shallow Swedish bays: effects on the community structure of epibenthic fauna and fish. *Aquatic Ecology* 29: 349–358.
- Pihl, L., J. Modin & H. Wennhage, 2005. Relating plaice (*Pleuronectes platessa*) recruitment to deteriorating habitat quality: effects of macroalgal blooms in coastal nursery grounds. *Canadian Journal of Fisheries and Aquatic Science* 62: 1184–1193.
- Pitcher, T. J., 2001. Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. *Ecological Applications* 11: 601–617.
- Pomeroy, L. R., C. F. D'Elia & L. C. Schaffner, 2006. Limits to top-down control on phytoplankton by oysters in Chesapeake Bay. *Marine Ecology Progress Series* 325: 301–309.
- Rabalais, N. N. & R. E. Turner, 2001. Coastal hypoxia: Consequences for living resources and ecosystems. American Geophysical Union, Washington, DC.
- Richards, P. R. & P. J. Rago, 1999. A case history of effective fishery management: Chesapeake Bay striped bass. *North American Journal of Fisheries Management* 19: 356–375.
- Rose, K. A., J. H. Cowan, K. O. Winemiller, R. A. Myers & R. Hilborn, 2001. Compensatory density-dependence in fish populations: importance, controversy, understanding, and prognosis. *Fish and Fisheries* 2: 293–327.
- Rose, K. A., B. A. Megrey, D. Hay, F. Werner & J. Schweigert, 2008. Climate regime effects on Pacific Herring growth using coupled nutrient-phytoplankton-zooplankton and bioenergetics models. *Transactions of the American Fisheries Society* 137: 278–297.
- Rosenthal, H., 1985. Constraints and perspectives in aquaculture development. *GeoJournal* 10: 305–324.
- Roy, D., G. D. Haffner & S. B. Brandt, 2004. Estimating fish production potentials using a temporally explicit model. *Ecological Modelling* 173: 241–257.
- Runge, J. A., P. J. S. Franks, W. C. Gentleman, B. A. Megrey, K. A. Rose, F. E. Werner & B. Zakardjian, 2004. Diagnosis and prediction of variability in secondary production and fish recruitment processes: developments in physical-biological modelling. In Robinson, A. R. & K. Brink (eds), *The Global Coastal Ocean: Multi-Scale Interdisciplinary Processes*, Vol. 13, The Sea. Harvard University Press, Cambridge, MA.
- Scheffer, M., S. Carpenter, J. A. Foley, C. Folke & B. Walker, 2001. Catastrophic shifts in ecosystems. *Nature* 413: 591–596.
- Secor, D. H., 2000. Longevity and resilience of Chesapeake Bay striped bass. *ICES Journal of Marine Science* 57: 808–815.

- Secor, D. H. & J. R. Waldman, 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. *American Fisheries Society Symposium* 23: 203–216.
- Secor, D. H., E. D. Houde & D. M. Montelone, 1995. A mark-release experiment on larval striped bass *Morone saxatilis* in a Chesapeake Bay tributary. *ICES Journal of Marine Science* 52: 87–101.
- Seitzinger, S. P., C. Kroeze, A. F. Bouwman, N. Caraco, F. Dentener & R. V. Styles, 2002. Global patterns of dissolved inorganic and particulate nitrogen inputs to coastal systems: recent conditions and future projections. *Estuaries* 25: 640–655.
- Selberg, C. D., L. A. Eby & L. B. Crowder, 2001. Hypoxia in the Neuse River Estuary: responses of blue crabs and crabbers. *North American Journal of Fisheries Management* 21: 358–366.
- Smith, J. W., 2001. Distribution of catch in the Gulf menhaden, *Brevoortia patronus*, purse seine fishery in the northern Gulf of Mexico from logbook information: are there relationships to the hypoxic zone? In Rabalais, N. N. & R. E. Turner (eds), *Coastal Hypoxia: Consequences for Living Resources and Ecosystems*. *Coastal and Estuarine Studies* 58, American Geophysical Union: 311–320.
- Stierhoff, K. L., T. E. Targett & K. Miller, 2006. Ecophysiological responses of juvenile summer and winter flounder to hypoxia: experimental and modeling analyses of effects on estuarine nursery quality. *Marine Ecology Progress Series* 325: 255–266.
- Stierhoff, K. L., T. E. Targett & J. H. Power, in press. Hypoxia-induced growth limitation of juvenile fishes in an estuarine nursery: assessment of small-scale temporal dynamics using RNA:DNA. *Canadian Journal of Fisheries and Aquatic Science*.
- Svendsen, J., K. Aarestrup, J. F. Steffensen & J. Herskin, 2006. Use of a novel acoustic dissolved oxygen transmitter for fish telemetry. *Marine Technology Society Journal* 40: 103–108.
- Szedlmayer, S. T., 1997. Ultrasonic telemetry of red snapper, *Lutjanus campechanus*, at artificial reef sites in the northeast Gulf of Mexico. *Copeia* 4: 846–850.
- Szmant, A. E., 2002. Nutrient enrichment on coral reefs: is it a major cause of coral reef decline? *Estuaries* 25: 743–766.
- Thiel, R., A. Sepulveda, R. Kafemann & W. Nellen, 1995. Environmental factors as forces structuring the fish community of the Elbe Estuary. *Journal of Fish Biology* 46: 47–69.
- Thomas, M., 1998. Temporal changes in the movements and abundance of Thames estuary fish populations. In Attrill, M. J. (ed.), *A Rehabilitated Estuarine Ecosystem*. Kluwer Academic Publishers, Dordrecht: 115–139.
- Tinsley, D., 1998. The Thames estuary: a history of the impact of humans on the environment and a description of the current approach to environmental management. In Attrill, M. J. (ed.), *A Rehabilitated Estuarine Ecosystem*. Kluwer Academic Publishers, Dordrecht: 5–26.
- Turchin, P., 1995. Population regulation: old arguments and a new synthesis. In Cappuccino, N. & P. W. Price (eds), *Population Dynamics: New Approaches and Synthesis*. Academic Press, New York: 19–40.
- Tyler, R. M. & T. E. Targett, 2007. Juvenile weakfish *Cynoscion regalis* distribution in relation to diel-cycling dissolved oxygen in an estuarine tributary. *Marine Ecology Progress Series* 333: 257–269.
- Vallin, L. & A. Nissling, 2000. Maternal effects on eggs size and egg buoyancy of Baltic cod, *Gadus morhua*—implications for stock structure effects on recruitment. *Fisheries Research* 49: 21–37.
- Venables, W. M. & C. M. Dichmont, 2004. GLMS, GAMS, and GLMMs: an overview of theory for applications in fisheries research. *Fisheries Research* 70: 319–337.
- Walker, N. D. & N. N. Rabalais, 2006. Relationships among satellite chlorophyll a, river inputs and hypoxia on the Louisiana continental shelf, Gulf of Mexico. *Estuaries and Coasts* 29: 1081–1093.
- Weisberg, S. B., H. T. Wilson, P. Himchak, T. Baum & R. Allen, 1996. Temporal trends in abundance of fish in the tidal Delaware River. *Estuaries* 19: 723–729.
- Wu, R. S., B. S. Zhou, D. J. Randall, N. Y. S. Woo & P. K. S. Lam, 2003. Aquatic hypoxia is an endocrine disrupter and impairs fish reproduction. *Environmental Science and Technology* 37: 1137–1141.

Assessment of the eutrophication status of transitional, coastal and marine waters within OSPAR

Ulrich Claussen · Wanda Zevenboom ·
Uwe Brockmann · Dilek Topcu · Peter Bot

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 49–58.
DOI: 10.1007/s10750-009-9763-3 © Springer Science+Business Media B.V. 2009

Abstract Eutrophication (nutrient enrichment and subsequent processes) and its adverse ecosystem effects have been discussed as main issues over the last 20 years in international conferences and conventions for the protection of the marine environment such

as the North Sea Conferences and the 1992 OSPAR Convention (OSPAR; which combined and updated the 1972 Oslo Convention on dumping waste at the sea and the 1974 Paris Convention on land-based sources of marine pollution). OSPAR committed itself to reduce phosphorus and nitrogen inputs (in the order of 50% compared with 1985) into the marine areas and ‘to combat eutrophication to achieve, by the year 2010, a healthy marine environment where eutrophication does not occur’. Within OSPAR, the Comprehensive Procedure (COMPP) has been developed and used to assess the eutrophication status of the OSPAR maritime area in an harmonised way. This is based on classification in terms of the following types of areas Non-Problem Areas (no effects), Potential Problem Areas (not enough data to assess effects) and Problem Areas (effects due to elevated nutrients and/or due to transboundary transport from adjacent areas). The COMPP consists of a set of harmonised assessment criteria with their area-specific assessment levels and an integrated area classification approach. The criteria cover all aspects of nutrient enrichment (nutrient inputs, concentrations and ratios) as well as possible direct effects (e.g. increased levels of nuisance and/or toxic phytoplankton species, shifts and/or losses of submerged aquatic vegetation) and indirect effects (e.g. oxygen deficiency, changes and/or death of benthos, death of fish, algal toxins). The COMPP also includes supporting environmental factors. It takes account of synergies and harmonisation with the EC Water Framework Directive, and has formed a major

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

U. Claussen (✉)
German Federal Environmental Agency
(Umweltbundesamt), Wörlitzer Platz 1,
06844 Dessau, Germany
e-mail: ulrich.claussen@uba.de

W. Zevenboom
Ministry of Transport, Public Works and Water
Management, RWS-North Sea Directorate Water and
Shipping, 2280 HV Rijswijk, The Netherlands
e-mail: wanda.zevenboom@rws.nl

U. Brockmann · D. Topcu
Institute for Biogeochemistry and Marine Chemistry,
Hamburg University, Martin-Luther-King-Platz 6,
20146 Hamburg, Germany

U. Brockmann
e-mail: brockmann@uni-hamburg.de

P. Bot
Rijkswaterstaat Waterdienst, Postbus 17,
8200 AP Lelystad, The Netherlands
e-mail: peter.bot@rws.nl

basis for the EC eutrophication guidance. Recently, additional components, such as total nitrogen, total phosphorus and transboundary transports have been included in the assessment of, e.g. the German Bight. The second application of the COMPP resulting in an update of the eutrophication status of the OSPAR maritime area will be finalised in 2008, and will include the agreed integrated set of Ecological Quality Objectives (EcoQOs) with respect to eutrophication.

Keywords Eutrophication · Assessment · OSPAR · Comprehensive procedure · German Bight · North Sea

Introduction

Assessments of the eutrophication status of transitional, coastal and marine waters within OSPAR are performed for the Northeast Atlantic, including the Greater North Sea, the Celtic Sea, the Bay of Biscay and the Iberian Coast. A similar procedure is applied in the Baltic Sea by HELCOM.

The aim of the eutrophication strategy of OSPAR is to combat eutrophication in order to achieve and maintain a healthy marine environment where eutrophication, as anthropogenically caused nutrient enrichment and succeeding effects, does not occur past 2010.

For this reason (i) repeated assessments are performed to characterise the regional eutrophication status by the so called ‘Common Procedure for the Identification of the Eutrophication Status of the Maritime Area of the Oslo and Paris Conventions’, an internationally harmonised classification procedure and (ii) OSPAR has, as a first step, initiated measures to reduce the nutrient inputs to parts of the OSPAR Convention area by 50% compared to 1985 within 10 years where these inputs are likely, directly or indirectly, to contribute to inputs into eutrophication problem areas.

The COMP has been developed within OSPAR to allow the characterisation of the maritime areas with regard to their eutrophication status on a common basis. It includes (i) a screening procedure for mainly offshore water where anthropogenically induced eutrophication effects are obviously absent and (ii) the Comprehensive Procedures (COMPP) to be applied in more affected coastal waters. Both

procedures were first applied in 2002 for the estuarine and marine waters of the OSPAR Contracting Parties.

A guideline for a harmonised holistic eutrophication assessment has been developed for COMPP, considering synergies and harmonisation with the EC Water Framework Directive (WFD), including a scheme of the main cause/effect relationships in the eutrophication process.

The main parts of this procedure and some selected examples are presented here briefly, considering that mainly the OSPAR Contracting Parties have free access to the OSPAR documents, including extended descriptions of COMPP and its applications.

Interaction of eutrophication processes

The common procedure of OSPAR includes the main parameters involved in eutrophication processes. They are roughly differentiated into four categories (Fig. 1):

- Category I. nutrient enrichment,
- Category II. direct effects (e.g. algal blooms),
- Category III. indirect effects (e.g. oxygen deficiency) and
- Category IV. other effects (e.g. algae toxins).

The parameters are arranged according to their causal relationships. These are illustrated in an ecosystem flow diagram, which simplifies the possible interactions. Other flow diagrams, developed for the management of aquatic ecosystems, may be focussed on estuaries (De Jonge & Elliott, 2001, Bricker et al., 2003) or more complex systems and may also include socioeconomic responses as well as management options (Lundberg, 2005). The OSPAR conceptual framework focussing on the main eutrophication processes has also been transferred to activities in the implementation of the EC Water Framework Directive (WFD).

The nutrient concentrations and ratios are controlled by the different inputs, imports and exports and losses (e.g. denitrification), followed by the nutrient uptake by phytoplankton and macrophytes as direct effects. Transboundary fluxes include imports and exports as well which are controlling all pelagic parameters in open coastal waters in addition to nutrients from other sources.

All processes are influenced by environmental factors, such as stratification, residence time of water

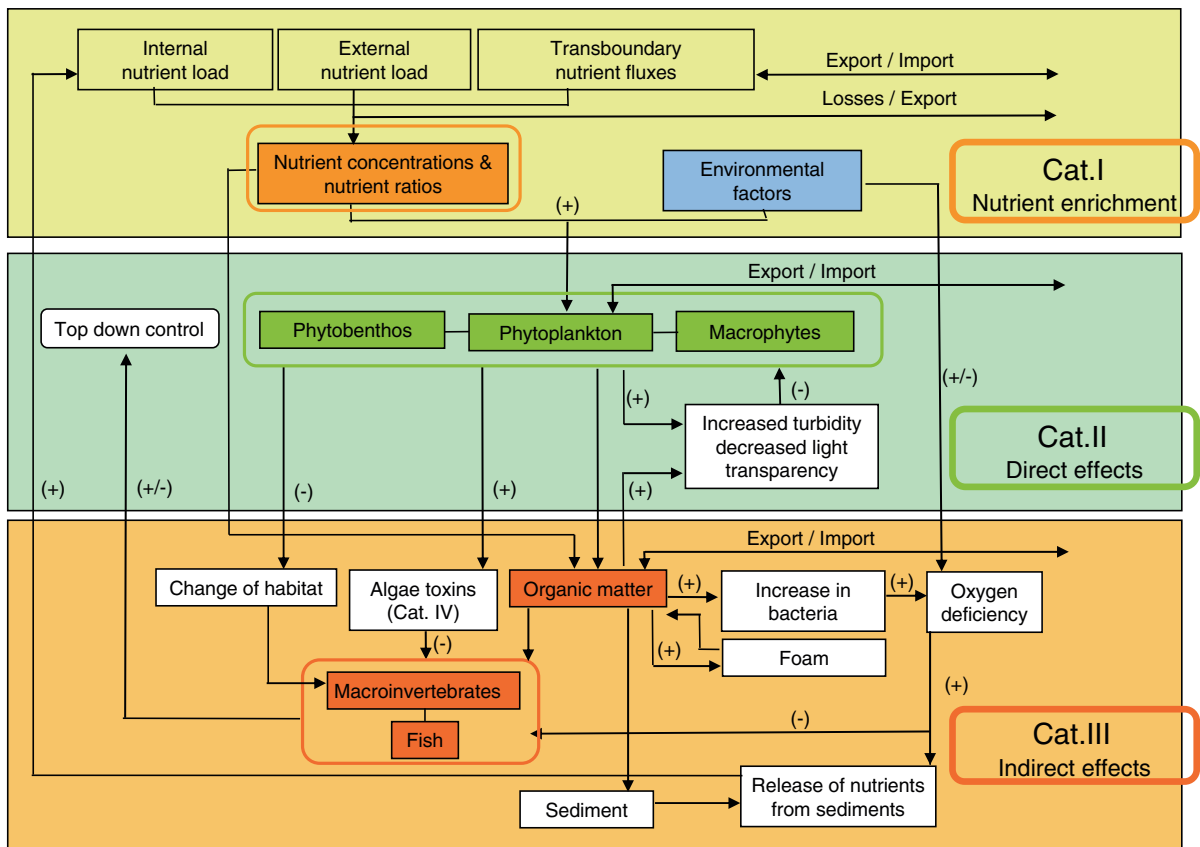


Fig. 1 Interaction of eutrophication processes (Cat. = categories of the eutrophication processes, (+) = enhancement of process, (-) = inhibition of process)

masses and turbulence, forcing the resuspension of sediments and affecting the light climate which is also influenced by phytoplankton standing crops.

The indirect effects include the production of organic matter from different sources. This material serves as food for the zoobenthos and may give rise to oxygen depletion in estuaries or the bottom water of stratified areas, after accumulation and decomposition, possibly causing death of zoobenthos and fish.

All key parameters combined by these processes, have been involved in the assessment of OSPAR (Fig. 2): river inputs, winter dissolved inorganic nitrogen (DIN) and phosphate concentrations (DIP), chlorophyll *a* of phytoplankton species, shifts of macrophytes and, in relation to nutrient enrichment: organic matter, oxygen deficiencies and death of animals, caused by low oxygen levels or toxic phytoplankton species. Finally, the changes of the ecosystem structure are assessed within specified

areas, which are characterised by salinity and specific environmental conditions, such as stratified and/or sedimentation areas.

Basic eutrophication assessment parameters are winter DIN and DIP concentrations. This season has been selected because during winter biologic activity (e.g. phytoplankton growth) is low. Therefore, highest nutrient concentrations can be expected and observed at that time of the year. This amount of nutrients is potentially available for the phytoplankton spring bloom. However, the winter nutrient concentrations at the specific monitoring site have mostly disappeared when eutrophication effects are observed in a specific area. For this reason, new assessment parameters have recently been introduced: total nitrogen (TN) and total phosphorus (TP) as annual means, thus combining seasonal processes to a large extent and allowing the combination of natural background concentrations from fresh water and sea water (Fig. 2).

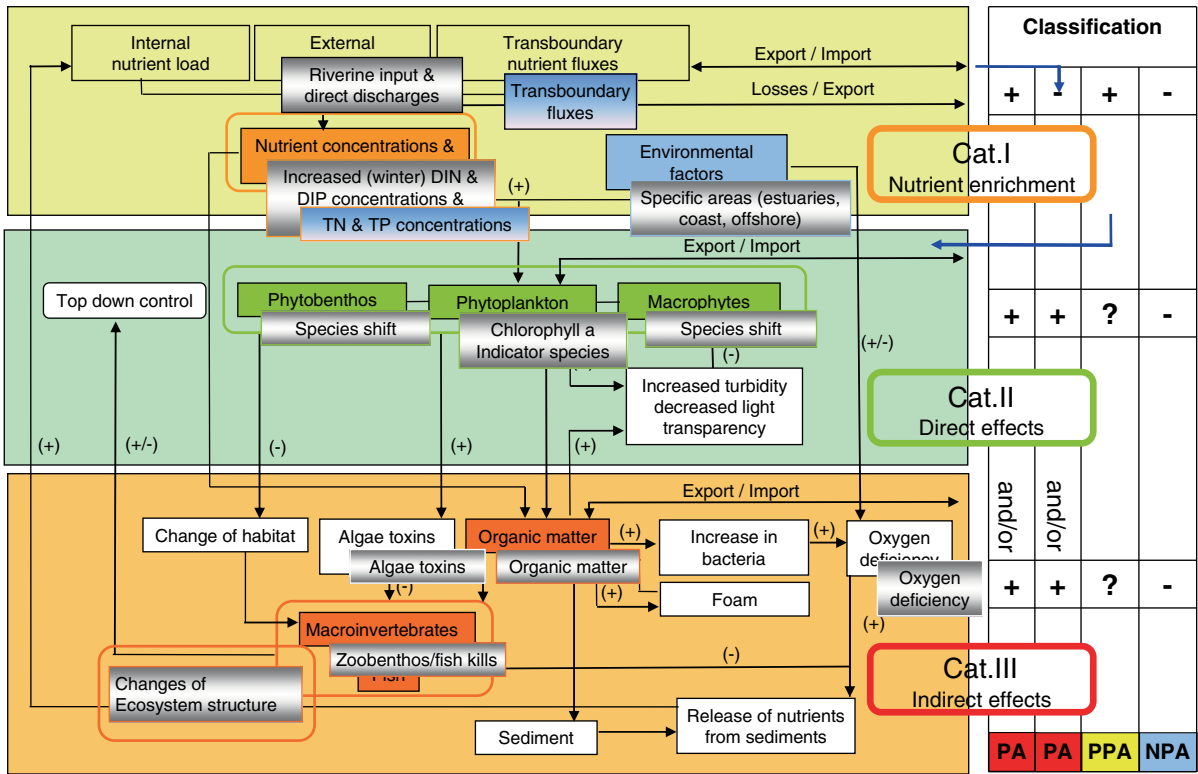


Fig. 2 Interaction of eutrophication processes (background), assessed parameters (in grey) and classification: “+” = above elevated level, “-” = below elevated level, “?” = insufficient

data; PA = Problem Area, PPA = Potential Problem Area, NPA = Non Problem Area

Since the nutrient gradients in the coastal water are not only influenced by river discharges and atmospheric deposition, but are mainly controlled by transboundary imports and exports, these transboundary transports have been added to COMPP as well.

Silicate concentrations will also be assessed in future, as background information for the possible shift in phytoplankton groups (from diatoms to flagellates) because many harmful species are flagellates and there are some indications for an increase of harmful blooms and flagellates abundances in connection with eutrophication (Radach et al., 1990, Cloern, 2001).

Assessments and measures

The anchor of the OSPAR eutrophication assessment is the definition of natural background concentrations (or levels) which serve as reference levels. Their derivation follows similar rules as for the WFD. If,

for a specific parameter, 50% of the area-specific natural background concentrations or conditions are surpassed the area is classified as “Problem Area” (PA), indicated by a [+]. No effects meaning a derivation <50% are classified as “Non-Problem Area” (NPA) by a [-]. Within the four categories, the “one out—all out” principle is applied. If significant eutrophication effects are observed, a [+] will be scored for category II or III, resulting in a classification as “Problem Area”.

If there are effects but no elevated nutrient concentrations, transboundary imports may be assumed. Areas without visible effects but with elevated nutrient concentrations are classified as “Potential Problem Areas” (PPA) because monitoring may have failed to detect effects, e.g. due to complex hydrodynamic conditions. However, nutrients may be exported from these areas causing transboundary effects in adjacent regions. PPA are indicated by “?”.

These classifications have been applied for the OSPAR maritime area by the Contracting Parties,

resulting in national reports on more or less extended “Problem and Potential Problem Areas”. Based on the national information, OSPAR produced an “Integrated Report 2003 on the Eutrophication Status of the OSPAR Maritime Area based upon the First Application of the Comprehensive Procedure”. Figure 3 shows the eutrophication status for the Greater North Sea for that assessment. Whereas along the continental coast from Belgium to Denmark large areas are classified as Problem Areas, along the coasts of the United Kingdom only estuaries are classified as Problem Areas. Along the steep Norwegian Skagerrak coast offshore areas are mainly classified as Potential Problem Area, contrary to the Danish coast and the Swedish Kattegat coast and despite the fact of frequent occurrence of harmful blooms along the Norwegian Coastal Current. These classifications, compiled by the OSPAR secretariat, are based on national reports.

As an example for one of the newly introduced parameters, an assessment of Total Nitrogen (TN) (means calculated from all seasons) is compared at different scales (Fig. 4). Determination of total nutrient concentrations includes all nutrient fractions

(inorganic dissolved, organic dissolved and particulate) and is less affected by seasonal nutrient conversion processes than the inorganic nutrients which are seasonally replaced by organic compounds (Butler et al., 1979). Annual means of TN can be assessed through all seasons, and thereby bridging the seasonal and regional developments. Additionally, significant correlations between TN and chlorophyll have been found in many areas (Smith, 2006) indicating quantitative relationships between these parameters. However, it has to be acknowledged that TN is a parameter that can be used on a voluntary basis to obtain an improved insight into its possible value for the eutrophication assessment. For TN to be included into the core set of harmonised OSPAR COMPP parameters further work is necessary.

For TN in coastal waters, natural background concentrations have been derived by modelling based on literature data for pristine German rivers (Behrendt et al., 2003) and extrapolated historical coastal concentrations (Van Raaphorst et al., 2000) for the German Bight. Differences between recent concentrations and the natural background were calculated as percentage of reference values. Regions, where

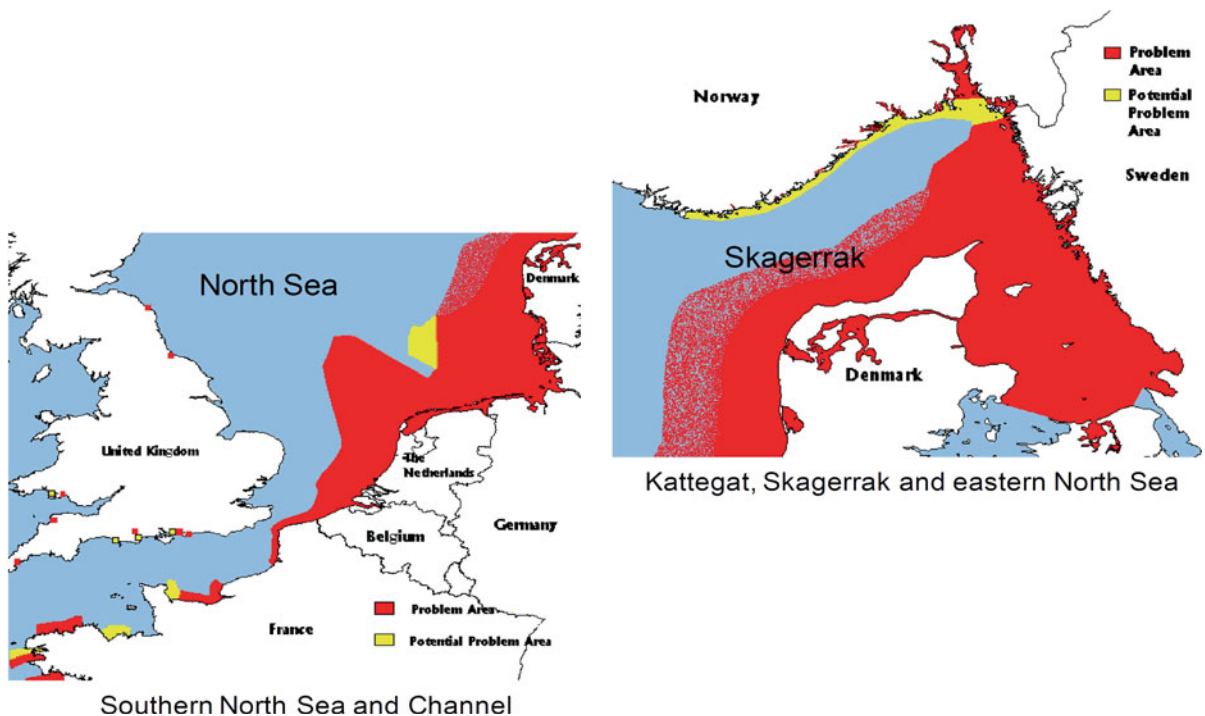


Fig. 3 Eutrophication status in 2002 of selected OSPAR areas

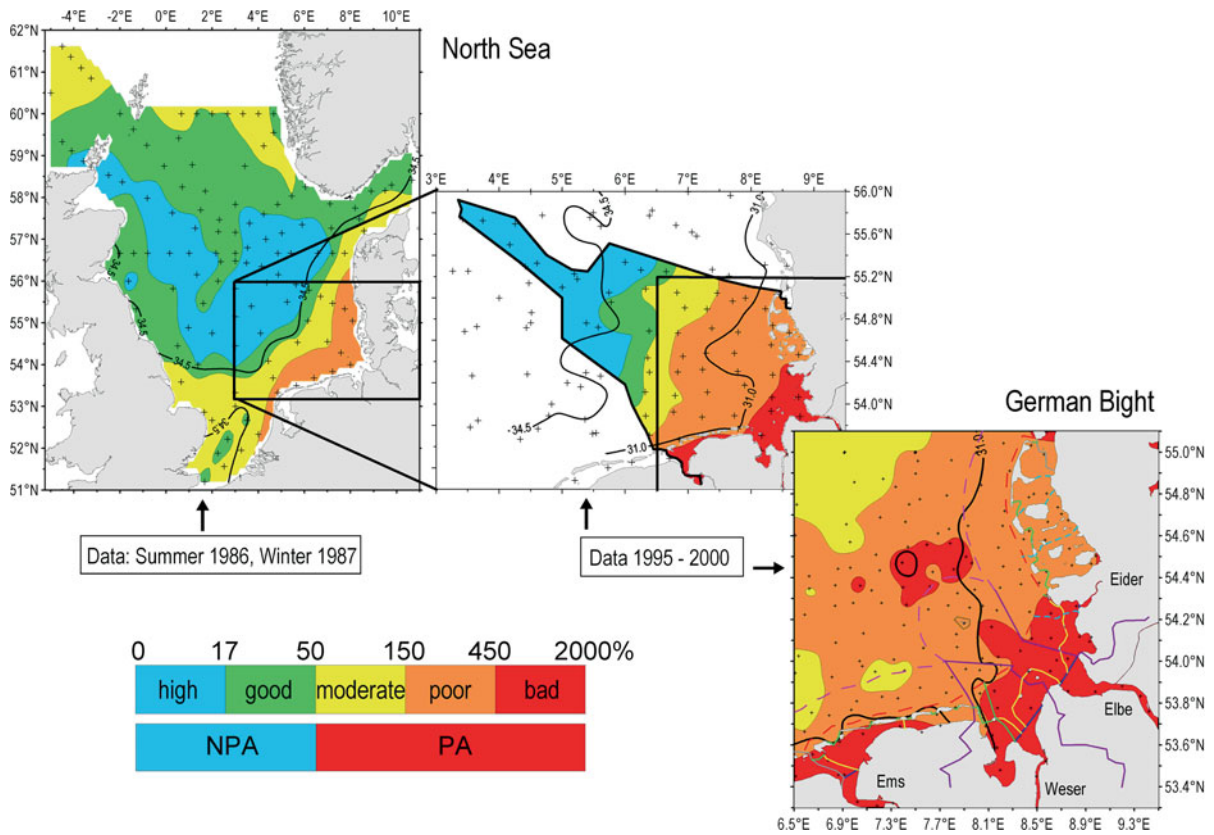


Fig. 4 Differences between pristine and recent TN in % of pristine values, which were developed for the German Bight and transferred to the whole North Sea as a test

150% of reference conditions (=100%) were surpassed, were classified as Problem Areas. This was the case for the whole German Bight, the Continental Coastal Water, loaded with about 20–30 μM TN, as well as small strips of Norwegian Coastal Water. The German approach was extended to the whole North Sea as a test, showing also the high nutrient loads of the Atlantic inflow affecting the shallow North Sea which is a sink for nitrogen (Seitzinger & Giblin, 1996). Due to the lack of data it was not possible to assess the same time period for all areas.

The open coastal waters receive and export most of the nutrients by transboundary transports. This process is also relevant for the WFD-areas because particulate material will be trapped partly in estuaries and tidal flats by the estuarine circulation and asymmetric tides.

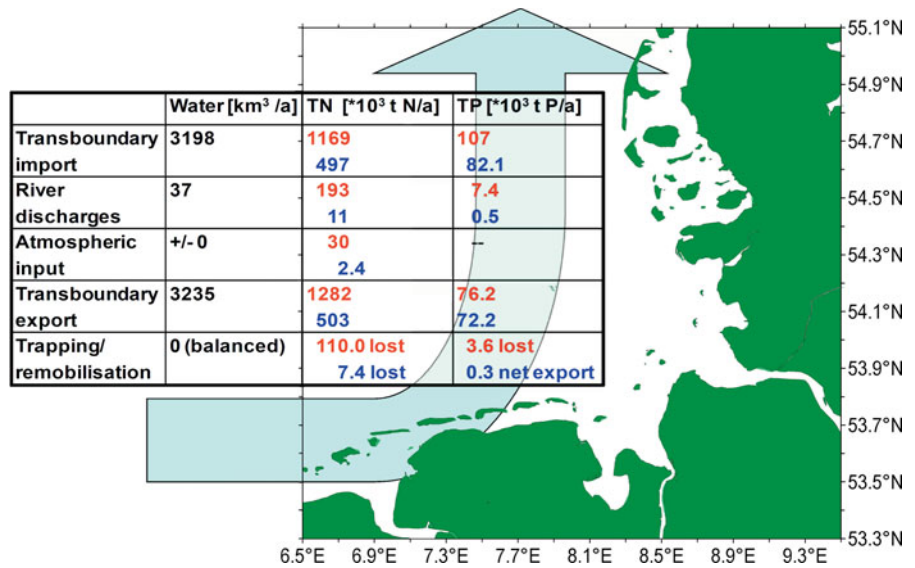
Based on the simple assumption of a constant coastal current transport (Mittelstaedt et al., 1983), imports and exports of the German Bight were calculated from recent (1995–2000) mean and pristine

offshore concentrations (TN: 11 μM , TP: 0.72 μM) (Fig. 5). The borders for the in- and outflow were closed to get a balanced budget of water masses. Recent data, necessary for budget calculations, are very scarce due to missing seasonal representative sampling and rare complete analyses of TN and TP. The river Rhine input is included within the import.

The transported water masses through the German Bight area, containing 476 km^3 water, were balanced by different in- and outflow velocities through the defined boundaries with their different cross-section sizes (west: 5.59 km^2 ; north: 3.42 km^2): 1.8 cm/s inflow, 3 cm/s outflow (946 km^3/y). Across the north-western edge the entering water will leave the area soon. The water masses will be exchanged about 7 times/year.

The atmospheric input into the area of the German Bight contributes with about 30,000 t/y nitrogen to the budget (Bartnicki & Fagerli, 2003, Rendell et al., 1993). Historically, 1 kg N/ha deposition is assumed (Nixon, 1997). About $110 \cdot 10^3$ t N/year are lost in

Fig. 5 Annual budget of TN and TP in the German Bight for pristine and mean recent (1995–2000) concentrations. The *arrow* indicates the residual current. The borders for the budget correspond to those of the map



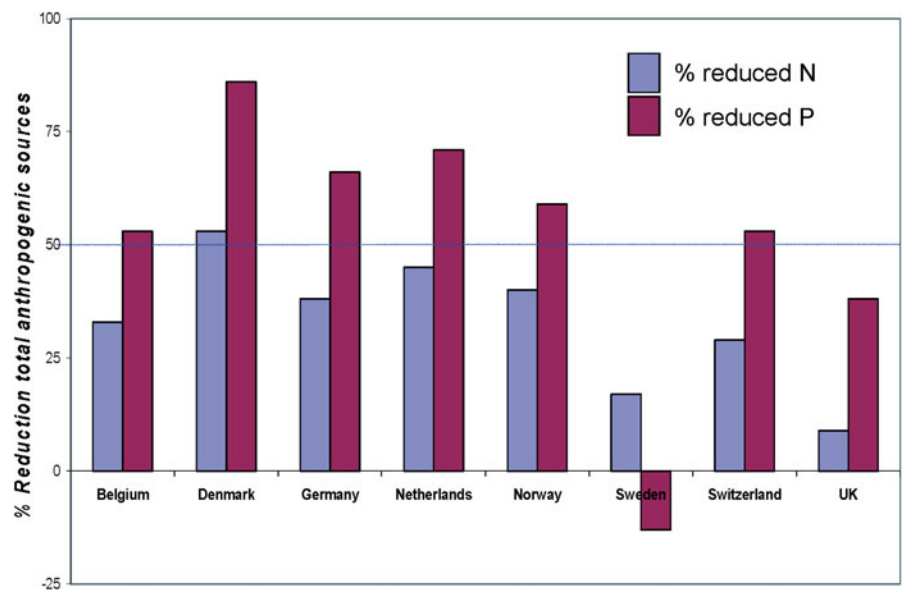
the German Bight, corresponding to 36 $\mu\text{M}/\text{m}^2 \text{ h}$ which is in the range of estimated denitrification rates (Lohse et al., 1996). Also 3,600 t/y phosphorus are lost, which is equivalent to 4.7 $\mu\text{M}/\text{m}^2 \text{ y}$ for the whole area. However, most of the particulate fraction was probably trapped in the Wadden Sea and estuaries due to the estuarine circulation (Postma, 1984). It must be assumed that also nitrogen was partly trapped in the sediments, at least transitionally.

Beddig et al. (1997) assumed transboundary transports of about $900\text{--}1,000 \times 10^3 \text{ t N/year}$ within

a nitrogen budget of the German Bight in 1990 and 1991, which is in the same range of the presented recent calculation.

Reacting to the observed eutrophication problems, the nutrient discharges from anthropogenic sources have been reduced significantly within the OSPAR Convention Area, stimulated by the agreement to reduce the loads of N and P by 50% compared to 1985 discharges. The achieved reductions between 1985 and 2003 show that this goal was generally met for phosphorus discharges (Fig. 6). For Germany the

Fig. 6 Total percentage reductions of anthropogenic discharges between 1985 and 2003



data are limited to 2000, and UK estimates for inputs from aquacultures are not included. For nitrogen discharges significant reduction could also be achieved in the order of 10–50%. However, the river discharges are very variable, especially for nitrogen, dependent on the amount of freshwater run-off. Therefore, long time periods and concentrations should be considered for trend analyses.

Towards a pan-European assessment strategy

The second application of the COMPP will result in an updated assessment of the eutrophication status of the OSPAR maritime area in 2008 and will include the agreed integrated set of Ecological Quality Objectives (EcoQOs) with respect to eutrophication.

Many of the parameters assessed by OSPAR are also considered within the WFD, which can be seen by

comparison (Fig. 7). These especially are the biological quality elements phyto-benthos, phytoplankton, macrophytes and macrozoobenthos, supplemented by assessment-supporting physico-chemical parameters like nutrients, oxygen conditions and transparency.

These synergies have been addressed by OSPAR including the “translation” of thresholds (Table 1). Non Problem Area (NPA) is corresponding to high and good scores, Problem Area (PA) to moderate and worse scores. However, the appropriate OSPAR threshold between NPA and PA, allowing a surplus of 50% of natural background concentrations, can lead to misclassifications according to the WFD which defines the “good status” by only “slight differences” from “high status” (=reference).

First steps of harmonisation between the different assessment methods have already been realised, by utilisation of central COMPP parts. An interim document “Towards a guidance document on eutrophication

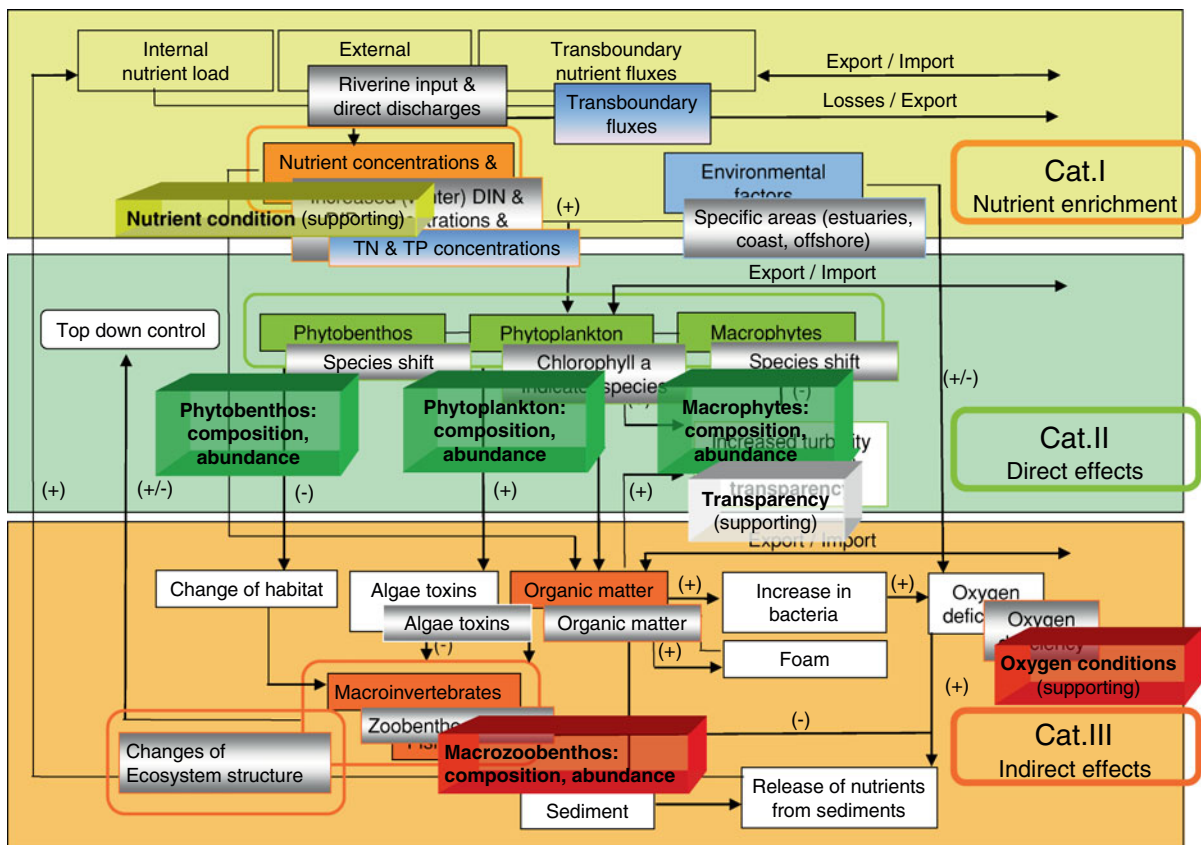


Fig. 7 Interaction of eutrophication processes: assessed OSPAR and WFD parameters. Background: interaction of eutrophication processes (Fig. 1); Second level: assessed parameters (Fig. 2), Surface level (in bold, 3-D-boxes): assessed WFD parameters

Table 1 Relationship between OSPAR COMPP, EcoQOs and WFD classification

| | | | | | | |
|-------------|---|------------------|------------------------|--------------|--------------|-----|
| OSPAR COMPP | Further Application | Non-Problem Area | | Problem Area | | |
| | Initial Application | Non-Problem Area | Potential Problem Area | | Problem Area | |
| WFD | | High | Good | Moderate | Poor | Bad |
| | | OSPAR Background | ↑ | | | |
| | | | | | | |
| EcoQOs | % Ecological Quality Objectives Eutrophication % OSPAR Assessment Level (reflecting natural variability and (slight) disturbance (OSPAR Background + 50 %)) | | | | | |

assessment in the context of European water policies” has been developed as part of the implementation of the EC Water Framework Directive. It is based on work of a “eutrophication steering group” led by DG Environment and on two workshops with experts for all water categories (lakes, rivers, transitional and coastal water) from all EU Member States and the Joint Research Centre. This guidance document includes a conceptual framework for eutrophication assessment based on the OSPAR conceptual framework but adapted to a pan-European perspective. In this document the different relevant European directives are considered, such as the Urban Waste Water Treatment Directive, the Nitrates Directives, and especially the Water Framework Directive. Key terms are compared, as well as the possible assessment results for waters responding to nutrient enrichment.

However, in comparison to OSPAR only the biological quality elements are considered with the same meaning. Nutrients, the main origin of anthropogenic eutrophication processes, which are fully assessed by OSPAR, are only considered as supporting parameters for the WFD assessments.

In addition to the European assessment methods, an assessment method has also been developed and broadly applied for estuaries in the United States (Bricker et al., 2003) which has many similarities to the methods of OSPAR and the WFD. Therefore, other developments and experiences should also be considered for the further development of a common assessment strategy. This integration of different assessment methodologies should be continued by harmonisation of:

- Area specification and typing (natural interfering processes),
- Determination of quantitative causal relationships for specific regions and tests of generally applicable correlations,
- References and thresholds,
- Supplementary parameters,
- Monitoring resolutions and requirements of precision,
- Development of classification tools,
- Reporting formats and time periods,
- Modelling of transboundary transports, residence times and reduction scenarios,
- Most effective reduction measures.

Climate changes, causing, e.g. longer seasonal stratification, may increase eutrophication effects, e.g. by extending oxygen depletion in enclosed bottom waters or supporting seasonal dominance of dinoflagellates. OSPAR is addressing these processes by assessment of combined effects of nutrient offshore transports, riverine discharges, atmospheric deposition, trapping and conversions processes, considering influences of hydrodynamic factors and their possible changes.

Improvements of transitional and coastal waters will result from implementation of the WFD, but they cannot be properly managed without considering imports of particulate matter by estuarine circulation and asymmetric tides. These are affected by transboundary fluxes of nutrients and organic matter. The complex eutrophication processes at the sea/shore transition zone are addressed by OSPAR, supplementing WFD and the new European Marine Strategy Framework Directive.

Acknowledgements The material for the Figs. 1, 2, 3 and 6 was provided by the OSPAR secretariat, London 2006. We are grateful to the technical assistance given by Monika Schütt and acknowledge the improvement of the manuscript by E. Lysiak-Pastuszek and an anonymous expert.

References

- Bartnicki, J. & H. Fagerli, 2003. Atmospheric supply of nitrogen to the OSPAR Convention Waters, EMEP MSC-W Report for OSPAR, EMEP-MSW Note 4/2003, Summary Report for UBA, Oslo, ISSN 0332-9879: 42 pp.
- Beddig, S., U. Brockmann, W. Dannecker, D. Körner, T. Pohlmann, W. Puls, G. Radach, A. Rebers, H.-J. Rick, M. Schatzmann, H. Schlünzen & M. Schulz, 1997. Nitrogen fluxes in the German Bight. *Marine Pollution Bulletin* 34: 382–394.
- Behrendt, H., M. Bach, R. Kunkel, D. Opitz, W.-G. Pagenkopf, G. Scholz & F. Wendland, 2003. Internationale Harmonisierung der Quantifizierung von Nährstoffeinträgen aus diffusen und punktuellen Quellen in die Oberflächengewässer Deutschlands. UBA-FB 000446, Texte 82/03: 201 pp.
- Bricker, S. B., J. G. Ferreira & T. Simas, 2003. An integrated methodology for assessment of estuarine trophic status. *Ecology Modelling* 169: 39–60.
- Butler, E. I., S. Knox & M. I. Liddicoat, 1979. The relationship between inorganic and organic nutrients in sea water. *Journal of the Marine Biological Association of the United Kingdom* 59: 239–250.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- De Jonge, V. N. & M. Elliott, 2001. Eutrophication. In Steele, J., S. Thorpe & K. Turekian (eds), *Encyclopedia of Ocean Sciences*. Elsevier, Amsterdam: 852–870.
- Lohse, L., R. Kloosterhuis, W. van Raaphorst & W. Helder, 1996. Denitrification rates in continental shelf sediments of the North Sea: acetylene block technique versus isotope pairing. *Marine Ecology Progress Series* 132: 169–179.
- Lundberg, C., 2005. Conceptualizing the Baltic Sea Ecosystem: an interdisciplinary tool for environmental decision making. *Ambio* 34: 433–439.
- Mittelstaedt, E., W. Lange, C. Brockmann & K. C. Soetje, 1983. Die Strömungen in der Deutschen Bucht. BSH Map, Nr. 2347: 141 pp.
- Nixon, S. W., 1997. Prehistoric nutrient inputs and productivity in Narragansett Bay. *Estuaries* 20: 253–261.
- Postma, H., 1984. Introduction to the symposium on organic matter in the Wadden Sea. Netherlands Institute for Sea Research Publication Series 10: 15–22.
- Radach, G., J. Berg & E. Hagmeier, 1990. Long-term changes of the annual cycles of meteorological, hydrographic, nutrient and phytoplankton time series at Helgoland and at LV ELBE 1 in the German Bight. *Continental Shelf Research* 10: 305–328.
- Rendell, A. R., C. J. Ottley, T. D. Jickells & R. M. Harrison, 1993. The atmospheric input of nitrogen species to the North Sea. *Tellus* 45B: 53–63.
- Seitzinger, S. P. & A. E. Giblin, 1996. Estimating denitrification in North Atlantic continental shelf sediments. In Howarth, R. W. (ed.), *Nitrogen Cycling in the North Atlantic Ocean and its Watersheds*. Kluwer Academic Publishers, Dordrecht: 235–260.
- Smith, V. H., 2006. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51: 377–384.
- Van Raaphorst, W., V. N. de Jonge, D. Dijkhuizen & B. Frederiks, 2000. Natural background concentrations of phosphorus and nitrogen in the Dutch Wadden Sea. RIKZ/2000.013, The Hague: 53 pp.

Can the EU agri-environmental aid program be extended into the coastal zone to combat eutrophication?

Odd Lindahl · Sven Kollberg

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 59–64.
DOI: 10.1007/s10750-009-9771-3 © Springer Science+Business Media B.V. 2009

Abstract Eutrophication of coastal waters is a serious environmental problem with high costs for society globally. This is a development which demands immediate environmental action along many coastal sites. Since the 1980s, mussel farming has been recognized by Swedish environmental authorities as a possible measure to improve coastal water quality. Concepts and management strategies on how to increase mussel farming and thus combat coastal marine eutrophication has recently been developed in Sweden. The main principle of this development has been the implementation of nutrient trading as a management tool. This imposes demands on those who emit the pollution through the establishment of emission quotas, which are traded and bought by the emitter. The seller is a nutrient harvesting enterprise, e.g., a mussel farmer. This principle is particularly straightforward when the nutrients are discharged from a point source. When examining the nutrient supply from all diffuse sources, the situation is more complex. However, since the major part of the nutrient

supply to coastal waters in many areas of Europe has its origin in agricultural operations, we suggest that the EU agro-environmental aid program could be extended into the coastal zone in order to combat eutrophication. In practice, this should involve support paid to mussel farming enterprises through their harvest of mussels (and thus their harvest of nutrients) in the same way as support is paid to agricultural farmers for operations that reduce nutrient leakage from their farmland. This is a simple, cost-effective and straightforward way of improving coastal water quality at many coastal sites that will, at the same time, provide coastal jobs. However, this eutrophication combat method depends on the EU agro-environmental aid program being extended beyond the shoreline.

Keywords Eutrophication · Mussel farming · Nutrient trading · EU agri-environmental aid · Coastal water quality

Introduction

Eutrophication of coastal waters is causing anoxic bottom conditions and the formation of algal mats in shallow bays (Diaz & Rosenberg, 1995; Cloern, 2001). Several international agreements (e.g., Hel-Com and OSPAR) include goals to reduce the supply of anthropogenic nitrogen and phosphorus to the sea, but these have so far not been met. The common blue mussel (*Mytilus edulis*) could be used to reduce

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

O. Lindahl (✉) · S. Kollberg
Kristineberg Marine Research Station, Kristineberg 566,
450 34 Fiskebäckskil, Sweden
e-mail: odd.lindahl@kva.se

nutrients in the coastal zone since mussels feed mainly on phytoplankton which in turn act as nutrient sinks. Thus, harvest of mussels will decrease the nutrient level in the coastal zone. One kilogram of live mussels can remove 8.5–12 g of nitrogen, 0.6–0.8 g of phosphorous, and about 40–50 g of carbon (Lutz, 1980; Petersen & Loo, 2004; Syversen, personal communication).

Over the last 10 years, much of the research performed on the Swedish West Coast concerning mussel farming has focused on its positive environmental aspects. It became clear that society was lacking a direct tool to encourage and support this farming. For this reason, Lindahl et al. (2005) suggested the introduction of nutrient emission trading as a compensation measure where the mussel farm enterprise would be paid for the ecosystem service provided. Another possibility is to exploit an already existing system for environmental aid, e.g., the existing EU-program within the agricultural sector (EEC 2078/92 and 1257/1999), to apply it for organisms grown in the coastal zone. In the agricultural environmental aid programs, support is currently given only for the establishment of wetlands, spring cultivation, catch crops, and so forth in order to decrease nutrient release from farmland into the environment.

We postulate that large scale mussel farming is a realistic and cost-effective method to decrease the negative effects of eutrophication. At the same time, healthy marine food is produced from a low level of the food chain, nutrients are recycled from sea to land and new jobs are provided. The potential of the ecological and environmental benefits of mussel farming on improving coastal water quality are scientifically well known, as pointed out by, e.g., Ryther et al. (1972), Haamer et al. (1999), Edebo et al. (2000), Newell (2004), and Lindahl et al. (2005).

Mussel farming as an environmental measure—the Swedish experience

Long-line farming

There are numerous sites along the Swedish west coast which are suitable for mussel farming, according to the criteria: access to a water area in competition with other coastal activities, reasonable wind protection, water depths between 6 and 25 m,

and average current speeds of more than 5 cm s⁻¹. Obtaining a farm license is normally not a problem.

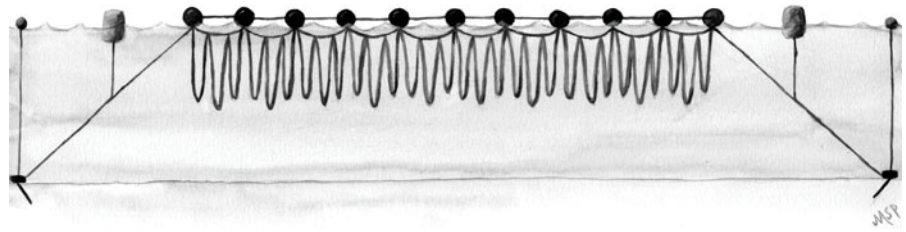
In Sweden, long-line farming is the most common method of mussel farming. The blue mussel (*Mytilus edulis*) is grown on vertical suspenders attached to horizontal long-lines (Fig. 1). On the Swedish west coast, about 300 tons of mussels may be produced per hectare of sea surface in 12–18 months. Each hectare of mussel farming needs between 25 and 15 hectares of phytoplankton for mussel food, depending on how fast the mussels grow. This calculation has been made by using the long-term mean from 1985 to 2006 of the annual primary phytoplankton production of 243 gC m⁻² year⁻¹ (Lindahl, 2007), a carbon content of 4% in the live mussel and a gross growth efficiency of 0.2 (Riisgård & Randløv, 1981). A similar calculation for the Baltic area showed that 7.5 hectares of food is needed for each hectare of farm area according to on-going mussel farm trials. This estimate was based on a production of 120–180 tons per hectare of long-line farmed mussels grown over 2–3 years (Lindahl & Kollberg, unpublished data) and a phytoplankton production of 160 gC m⁻² year⁻¹ (Elmgren, 1984). It was assumed that the carbon content and gross growth efficiency was the same as above.

One often-discussed drawback of long-line mussel farming is the bio-sedimentation below the farms. The negative effects depend on biological and technical farming conditions in relation to bottom water exchange at the site. It is very important that the sediment surface never becomes anoxic in order to maintain the nitrification and denitrification processes (Newell, 2004). Oxidic bottom conditions are maintained through natural bottom water renewal, which supplies the sediment surface at the site with enough oxygen. Best practices when running a mussel farm should include monitoring the sediment below the farm and managing the farming according to how the conditions develop. If conditions deteriorate, it is comparatively easy to move a long-line system since it has only one anchor at each end.

First example of nutrient trading

In order to combat eutrophication of the Swedish west coast, it was suggested that the volume of farmed mussels should be raised significantly (Lindahl et al., 2005). Swedish mussel farming did not

Fig. 1 Schematic drawing of long-line mussel farm



expand at all for many years until the community board of Lysekil was permitted (Fig. 2), as a trial between 2005 and 2011, to continue to emit nitrogen from the sewage plant, presupposing that the same amount of nitrogen was “harvested” and brought ashore by 3,900 tons of farmed blue mussels. The cost of 150,000 € for the Lysekil community was far below the price for nitrogen removal in the sewage plant. This payment goes to a mussel farming enterprise, which has been contracted for the removal of 39 tons of nitrogen from the recipient. This was estimated to correspond to 100% nitrogen treatment of the emission from the sewage treatment plant. Further, 3.6 tons of phosphorus are also removed annually from the recipient through the mussel harvest, which also could be traded (Lindahl et al., 2005).

The Lysekil case is a good example of the principle of trading nutrient emissions from a point source. However, the authors want to point out that mussel farming as a nutrient compensation measure should be reserved for diffuse emissions into the coastal zone coming from, for example, agriculture and atmosphere, because for these there are few other effective options available. This is important especially as the diffuse emissions into the coastal waters make up more than 80% of the total of which roughly half is coming from agriculture operations (Anon., 2001).

The EU agro-environmental aid program and mussel farming

Further growth of the mussel industry is still desired by the Swedish society. The authors therefore suggest that diffuse nutrient emissions into the coastal zone, e.g., from agriculture, should be traded and possible to compensate by mussel farming (Fig. 3). In the agricultural sector a legal system, the EU agro-environmental

aid program, already exists to promote environmentally healthy methods of decreasing nutrient release from farmland into the environment. The legal framework is quite complex, but some of the measures are easily comparable with mussel farming.

Mussels can be considered as a type of catch crop. In the aid program, catch crops are compensated for at 100 € per hectare. The annual compensation for spring cultivation of a catch crop was in 2006 45 € per hectare. In Sweden, catch crops are farmed on 180,000 hectare of farmland, and spring cultivation is used on 90,000 hectare. The total annual cost for these measures is 22 million €. The Swedish Commission for the Environment of the Seas (Anon., 2003) has calculated that catch crops and spring cultivation together decrease nitrogen release by 2,000 tons. This gives a price of 11 € per kg of retained nitrogen. If the mussels were to be compensated according to the same price for retained nitrogen, it means an environmental subsidy of about 0.11 € per kg of live mussels. This is roughly 25% of what a mussel farm enterprise needs as gross income for harvested mussels (as estimated by the authors).

Another method to decrease nitrogen emissions is the establishment of wetlands, which reduces nitrogen by an average of 120 kg per hectare. Contracts for maintenance are set up for a 20-year period, in 2006 allocated as a yearly compensation of 325 € per hectare, equivalent to 2.7 € per kg of nitrogen. The construction costs could be compensated for by 90% at the most and with an upper limit of 11,000 € per hectare. One hectare of wetland (=120 kg N) would be equivalent to approximately 12 tons of mussels. A long-line mussel farm covering 1 hectare will cost about 46,000 € and produce 300 ton of mussels per 12–18 months—in practice, 150 tons per year. The harvest of these will therefore, with a focus on nitrogen, correspond to about 12.5 hectares of wetlands. If the mussel farm construction costs were

Fig. 2 Map of north-west Europe and the Swedish west coast

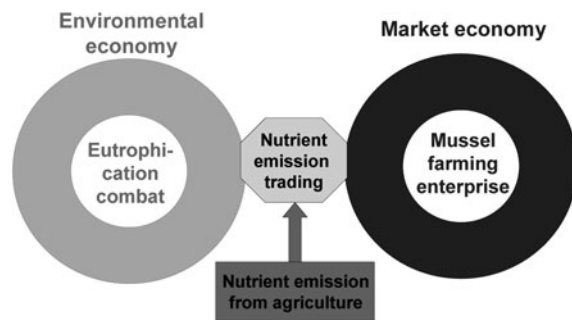
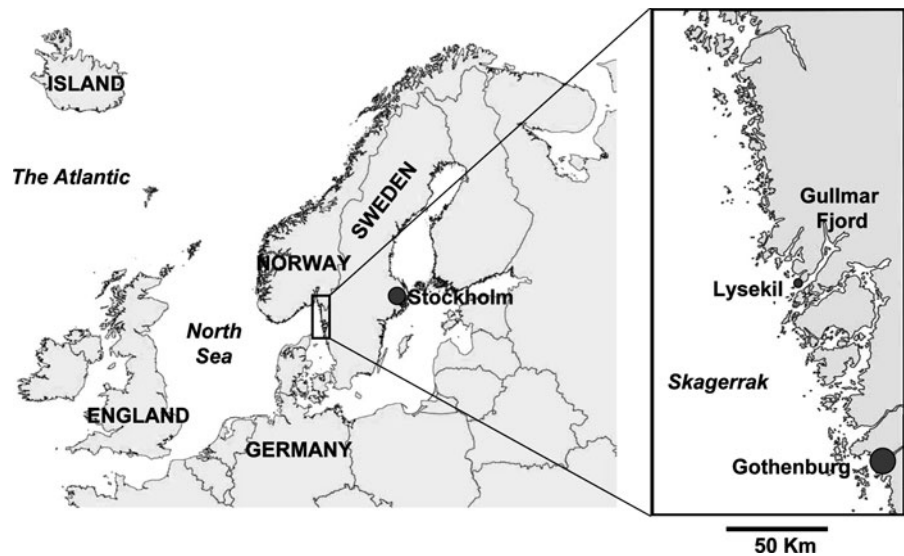


Fig. 3 The principle of nutrient trading, which connects the environmental and the market economies

compensated for by 41,400 € (90% of 46,000 €), it is far below the maximum limit of 137,500 € for 12.5 hectare of wetland construction.

A full comparison should include a comparison of the low running costs of wetlands versus the much higher costs for mussel farming. Harvesting costs alone amount to 0.10–0.15 € per kg. On the other hand, a wetland does not produce a commodity of commercial value, while the first-hand value of mussels is between 0.3 and 0.5 € per kg.

During the years 1998–2001, 25 hectares of wetlands were established on the Swedish west coast. The investment costs seen over a 20-year period were calculated to be 3,400 € per hectare per year. With a calculated reduction of 120 kg of nitrogen per hectare per year, the costs per kg of reduced nitrogen would be 28 €. This cost transformed to nitrogen uptake by mussels corresponds to about 0.23 € per kg.

Another option to finance the trading of nutrient emissions from agriculture would be to use the environmental tax paid on fertilizers. This Swedish tax has been introduced to decrease the elevated levels of fertilizer spread in the fields. In the opinion of the authors, this tax could have a double effect if some of it were used to further decrease nutrient levels in the sea through subsidizing extended mussel culture. This could, at least partly, pay for the nutrient removal service mussel farming provides.

When comparing the uptake of nutrients by catch crops and wetlands to mussels, the temporal aspect is important. In agriculture and for wetlands the uptake is far higher when temperatures are high (during summer). The primary production in the sea and also the mussel filtration capacity is much less temperature-dependent (Loo & Rosenberg, 1983). The phytoplankton growth period is more light-dependent and may last from March to October (Lindahl, 1995). This means that nutrient emissions during 8 months of the year are more or less rapidly assimilated by the phytoplankton, which then is grazed by the mussels.

The market for mussels

Food

The economic basis of mussel farming and harvesting is generally to produce food for human consumption. The world production of mussels today exceeds

1.5 million tons, of which half is produced in Europe. The demand is steadily increasing but the main production areas in Europe have reached a level where they can no longer expand due to the shortage of suitable farm areas (Smaal, 2002). The farms produce a valuable and healthy marine food product, since mussel meat is high in protein with a fat content of only about 2%—of which 40% is Ω 3 long-chain fatty acid molecules (Berge & Austreng, 1989). An increase in production to 50,000 tons annually on the Swedish west coast seen over a 15-year period seems therefore from a market point of view to be quite realistic.

Issues include elevated levels of diarrhetic shellfish toxins (DST) in blue mussels have caused sales problems. These toxins have been recorded along Swedish marine waters in varying amounts every year since 1983, when the first outbreak was reported (Haamer et al., 1990). This is regarded as a minor obstacle which can be handled by the producers who have to follow the regulations set up by EU directive 91/492 EEG. The same directive also regulates the harvest of mussels in relation to the occurrence of pathogenic bacteria and viruses. Chemical and other harmful substances must also be controlled. According to data in the literature and reports from the Swedish west coast, harmful substances are generally low and well below existing limits for use as food, in feed and as fertilizer (Kollberg & Ljungqvist, 2007).

Feed and fertilizer

A mussel farm can be regarded as the engine in an Agro-Agua recycling system of nutrients from sea to land. To optimize the environmental effect of mussel farming, all organisms attached to the lines should be harvested and brought ashore, with nothing discarded back to the sea. While not all harvested mussels can be used for human consumption, the remainder, consisting of small or damaged mussels, is also important for the removal of nutrients and can be used for feed or as an organic fertilizer. Large-scale experiments to evaluate mussel meal as a replacement for fish meal in organic chicken poultry and egg production have been performed (Jönsson, 2007). The results were most promising.

Since mussels are at the second step of the marine food-chain, the use of mussels instead of fish for meal production also has a large ecological advantage.

Furthermore, there is increased public opinion that fish should be left for human consumption, with the exception of pure industrial offal. A possible scenario is that mussel meal for use in feed will become so interesting for the market that special cost-effective farming and harvesting techniques for “feed mussels” will be developed for this purpose.

Further, successful use of waste from mussel processing lines as fertilizer in the organic farming of grain has also been documented (Lindahl et al., 2005; Olrog & Christensson, 2008). Successful composting experiments were done to produce a “mussel fertilizer” which can be stored and used when the farmer needs it and which lacks the bad smell of decomposing mussels.

Conclusions

Many different measures have to be used in the fight against eutrophication. However, economic resources are always short, and therefore it is necessary to use solutions that give the best return—in this case, the most nutrient reductions for the money spent. Mussel farming has shown to be a socio-economically sound measure that meets the requirements of the European Water Framework Directive in a cost-effective way (Sánchez-Hjortberg, 2003; Anon., 2004). It also gives rise to new jobs on the coast. At the same time, the drawbacks seem to be small and manageable, and the licensing procedure gives society the possibility of avoiding conflicts arising from the fact that the coastal zone is of great interest for many activities.

The authors are convinced that mussel farming would be a competitive compensation measure for agricultural emissions in a trade bidding system, and that it has a number of added values. However, if mussel farms are going to be subsidized according to the EU agro-environmental aid program, legislation must regard the sea surface utilized for mussel farming as farmland. This, in turn, requires that the directive regulating the agricultural aid program be updated.

The simplistic beauty of using nature for self-healing challenges today’s focus on high-technological advances to solve all our environmental problems. The usefulness of the mussel farming concept in combination with nutrient emission trading has significance on a global environmental scale.

Acknowledgment The authors would like to thank Dr. Max Troell for valuable and constructive criticism.

References

- Anon., 2001. The Skagerrak—Environmental State and Monitoring Prospects. ISBN 91-89507-04-5 [Available at the Swedish Meteorological and Hydrological Institute, (SMHI)].
- Anon., 2003. Havet—tid för en ny strategi. Betänkande av Havsmiljökommissionen. Statens offentliga utredningar. SOU 2003:72, 2003 (in Swedish with English summary).
- Anon., 2004. Miljøøkonomisk analyse af Mariager Fjord. CEA af Mariager Fjord. Aarhus-och Nordjyllands Amter. <http://www.mariager-fjord.dk/Rapporter/indsatmuligheder/Miljøøkonomisk%20analyse.pdf>.
- Berge, M. G. & E. Austreng, 1989. Blue mussel in feed for rainbow trout. *Aquaculture* 81: 79–90.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Diaz, R. J. & R. Rosenberg, 1995. Marine benthic hypoxia. A review of its ecological effects and the behavioral responses of benthic macrofauna. *Oceanography and Marine Biology: An Annual Review* 33: 245–303.
- Edebo, L., J. Haamer, O. Lindahl, L.-O. Loo & L. Piriz, 2000. Recycling of macronutrients from sea to land using mussel cultivation. *Environmental Pollution* 13: 190–207.
- Elmgren, R., 1984. Trophic dynamics in the enclosed, brackish Baltic Sea. *Rapports et Procès-verbaux des Réunions. Conseil International pour l'Exploration de la Mer* 183: 152–169.
- Haamer, J., P.-O. Andersson, O. Lindahl, S. Lange, X. P. Li & L. Edebo, 1990. Geographic and seasonal variation of Okadaic acid content in farmed mussels, *Mytilus edulis* Linnaeus, 1758, along the Swedish west coast. *Journal of Shellfish Research* 9: 103–108.
- Jönsson, L., 2007. Foderförsök med värphöns och kyckling. In Muslingemel i stedet for fiskemel i økologiske foder till æglæggede høns, kylling og andre husdyr. TemaNord 2007: 540 (in Swedish).
- Kollberg, S. & L. Ljungqvist, 2007. Musslor som livsmedel och råvara inom lantbruket. Ekhaga Stiftelsen projekt: 2005: 48 (in Swedish).
- Lindahl, O., 1995. Long-term studies of primary phytoplankton production in the Gullmar fjord, Sweden. In Skjoldal, H. R., C. Hopkins, K. E. Erikstad & H. P. Leinaas (eds), *Ecology of Fjords and Coastal Waters*. Elsevier Science Publishers B.V., Amsterdam: 105–112.
- Lindahl, O., 2007. Primary production in the Gullmar Fjord. Swedish National Report on Eutrophication Status in the Kattegat and the Skagerrak. OSPAR Assessment 2007: 41–43.
- Lindahl, O., R. Hart, B. Hernroth, S. Kollberg, L.-O. Loo, L. Olrog, A.-S. Rehnstam-Holm, J. Svensson, S. Svensson & U. Syversen, 2005. Improving marine water quality by mussel farming—a profitable measure for Swedish society. *Ambio* 34(2): 131–138.
- Loo, L.-O. & R. Rosenberg, 1983. *Mytilus edulis* culture; growth and production in western Sweden. *Aquaculture* 35: 137–150.
- Lutz, R. A. (ed.), 1980. *Mussel Culture and Harvest: a North American Perspective*. Elsevier Scientific Publishing Company, Amsterdam.
- Newell, R. I. E., 2004. Ecosystem influences of natural and cultivated populations of suspension feeding bivalve molluscs: a review. *Journal of Shellfish Research* 23: 51–61.
- Olrog, L. & E. Christensson, 2008. Användning av musslor och musselrester som gödselmedel i jordbruket. The Rural Economy and Agricultural Societies Report 2008: 1 (in Swedish with English summary).
- Petersen, J. K. & L.-O. Loo, 2004. Miljøkonsekvenser af dyrking av blåmuslinger. Interregproject III-A “Blåskjellanlegg og nitrogenkvoter”. Slutrapport 31.8.2004 (in Danish).
- Riisgård, H. U. & A. Randløv, 1981. Energy budgets, growth and filtration rates in *Mytilus edulis* at different algal concentrations. *Marine Biology* 61: 227–234.
- Ryther, J. H., W. M. Dunstan, K. R. Tenore & J. E. Huguenin, 1972. Controlled eutrophication: increased food production from the sea by recycling human wastes. *BioScience* 22: 144–152.
- Sánchez-Hjortberg, A., 2003. Blåmusselodling på västkusten, ett projektarbete vid SLU:s Vattenbruksinstitution [Available at the Swedish Agricultural University] (in Swedish).
- Smaal, A. C., 2002. European mussel cultivation along the Atlantic coast: production status, problems and perspectives. *Hydrobiologia* 484: 89–98.

Eutrophication and agriculture in Denmark: 20 years of experience and prospects for the future

Niels Peter Nørring · Erik Jørgensen

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 65–70.
DOI: 10.1007/s10750-009-9772-2 © Springer Science+Business Media B.V. 2009

Abstract During the past two decades there has been growing public and political awareness of the consequences of eutrophication in Denmark. By the mid-1980s, the environmental status of inland and coastal waters had deteriorated due to high nutrient loads. Consequently, a number of different Action Plans against water pollution were introduced. In the agricultural sector, focus has been on reductions in nitrogen leaching obtained by the introduction of various measures: a maximum limit to the density of livestock, 9 months' storage capacity for manure, catch crops for at least 6% of the land, enhanced utilization (up to 75%) of nitrogen in manure, etc. The agricultural sector in Denmark has implemented all of these measures, and as a result of the effort, the target for reductions in nitrogen leaching will be reached. Currently, the total loss of nitrogen from farmland is likely to be reduced by approximately 50% compared to the level in the mid-1980s. Some of the measures have been fair and based on sound arguments, and have been implemented with only

minor difficulties, whereas others have proved troublesome and in our opinion disproportionately expensive. Today, further general regulation with equal restrictions toward all farmers regardless of differences in environmental impacts is no longer an acceptable path to follow. In the future, it will be necessary to pinpoint new measures in the most sensitive areas, where the potential for further reductions in nutrient loads is large. Danish Agriculture calls for specific actions—and consequently a shift in environmental management and policy making. Such a revised management strategy is the only path to follow in order to obtain further improvements in environmental conditions. Meanwhile, future development in the agricultural sector will be possible and a win–win situation can be reached.

Keywords Agriculture · Eutrophication · Nutrient reductions · Trend reversal · Needs for specific measures · Management

Introduction

Eutrophication is not a new phenomenon in Denmark. Inland and coastal waters have been under the influence of human activities during thousands of years, in some cases leading to nutrient enrichment and changes in ecosystem structure, functioning and stability (Conley et al., 2002; Ærtebjerg et al., 2003). During the past century, eutrophication processes

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

N. P. Nørring · E. Jørgensen (✉)
Danish Agriculture (Dansk Landbrug), Vesterbrogade 4A,
1620 Copenhagen, Denmark
e-mail: ejo@dansklandbrug.dk

were enhanced due to changes in social structures. The discharge of urban and industrial wastewater directly to watercourses, lakes, and coastal areas increased, and agriculture became more and more intensive, with increasing inputs of nutrients from fertilizers and manure. Gradually, many aquatic habitats deteriorated, and in the 1970s and 1980s public and subsequently political concern arose (Danish Environmental Protection Agency, 1991).

Historically, agriculture has always played an important role in Denmark. Even though its importance is decreasing in terms of employment, the agricultural sector still plays a major role in society. More than 60% of the total area of Denmark is cultivated (Statistics Denmark, 2005) and livestock production and density are high, although with large geographical differences. In some parts of the country, animal density is very close to the upper limit allowed according to national legislation. The environmental impact is, however, not only linked to the use of fertilizer and manure, but it is also—among others—a function of soil characteristics, crop rotation, precipitation, temperature, and vulnerability of the affected aquatic environment (Knudsen et al., 2000).

To understand the reasons and mechanisms behind the structure and behavior in the agricultural sector, it is necessary to focus on the fact that policies covering the sector agriculture—in Denmark as well as in the European Union as a whole—are largely influenced by the Common Agricultural Policy (CAP). The common policy of today is very different from that of 33 years ago, when Denmark joined the EEC. Back then there was price support, inducing a high production with little or no focus on correlated issues such as environmental conditions. In 1992 a shift toward direct payment was introduced with the MacSharry reform, and direct payments without demands for production rates were further enhanced with the Agenda 2000.

In 2003, the CAP reform introduced a decoupling between payments and land use and cross-compliance was introduced. Cross-compliance is used as a tool to ensure that the individual farmer meets certain demands—including some of which are environmentally supportive—before he or she can receive the full payment from the EU. The changes in agricultural policy during the past 15 years have already led to the improvements in environmental conditions, not least in aquatic ecosystems, and it is likely that the

improvements will continue (e.g., Ærtebjerg et al., 2005; Carstensen et al., 2006).

Danish agriculture and environmental policies

A large number of EU Directives, international obligations and the subsequent national legislation have an influence on agricultural practice and production with regard to eutrophication. As an example, the Nitrates Directive contains general definitions as well as specific targets and measures. In addition, Denmark has ratified the OSPAR and Helsinki Conventions, thereby committing the country to specific environmental objectives. In the future, the EU Water Framework Directive (European Communities, 2000) is likely to be the most important legislation concerning the aquatic environment.

EU Directives and international political agreements (e.g., HELCOM, OSPAR, and North Sea Conferences) have influenced a number of national Action Plans on the Aquatic Environment, introduced between 1985 and 2004. The plans and their most important objectives and measures are listed briefly in Table 1. In general, the political focus has been on losses from the root zone, whereas agriculture has always argued that attention should instead be paid to the nutrient load that actually reaches the aquatic environment. Model results from Ringkøbing Fjord and Nissum Fjord suggest that on average 60–70% of the nitrogen leaking through the root zone at field scale is removed before reaching the coastal water, with significant spatial variation throughout the river basins (Ringkøbing County, 2004).

Results from efforts in agriculture: lower nitrogen leaching—mission completed

The various action plans introduced a variety of measures, all of which were supposed to add to the targeted reduction in nitrogen leaching, set by the first Action Plan (1987) to a maximum loss of 145,000 tons of nitrogen per year. In the evaluation of Action Plan II in 2003, it was concluded that the objective in terms of changes in agricultural practices was finally reached (Grant et al., 2006). Consequently, the nitrogen load from agriculture was reduced by 48%. The phosphorus load was more

Table 1 National action plans enacted in Denmark since 1985

| Year | Plan | Objectives | Measures |
|------|-------------------------|---|---|
| 1985 | NPo Plan | | Regulation of animal density Manure: minimum storage capacity |
| 1987 | Action Plan I | 145,000 t reduction of N-loss (50%) 15,000 t reduction of P (80%, primarily WTP) | 50% reduction in N-leaching Winter plant cover Compulsory plans on N-use |
| 1991 | Sustainable agriculture | Same as Action Plan I | Improved utilization of manure Maximum N-standards for crops Fertilizer plans/accounts |
| 1998 | Action Plan II | Same as Action Plan I | Further improved utilization of manure Reduced N-quota (10% below economic optimum) Conversion of cultivated soil to wetlands and forests |
| 2004 | Action Plan III | Further 13% reduction in N-leaching 50% reduction of P-surplus in cultivated soils | Further improved utilization of manure Increased catch crops demands Additional wetlands |

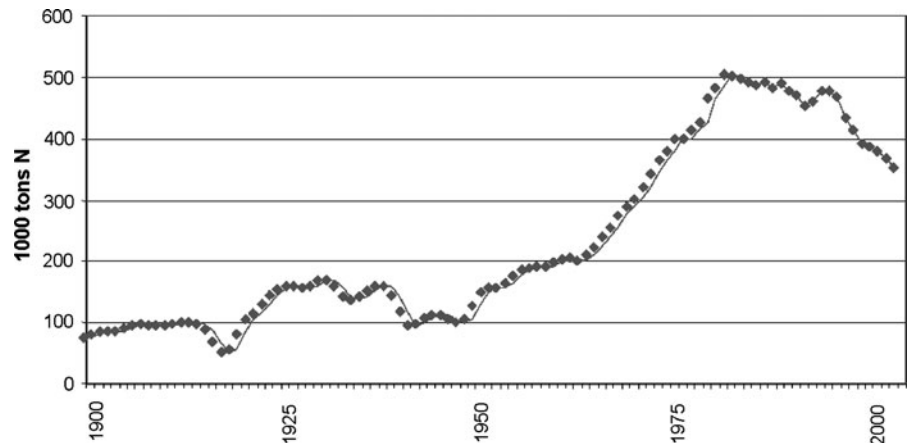
Some important objectives and measures concerning agriculture are listed
WTP wastewater treatment plant

than 80% lower than that before 1984 primarily due to improved wastewater treatment. The objectives set in the Action Plans by politicians and water managers were met by the Danish farmers and improvements in wastewater treatment.

Figure 1 shows the development in annual nitrogen surplus from agriculture (please note that only part of the surplus will actually enter the aquatic recipients because of removal by the denitrification process in the soil). Figure 2 shows the changes in nitrogen leaching from cultivated soils in Denmark since 1985, with a prognosis of the development until 2012 when the last measures are fully implemented.

After a maximum surplus was reached during the 1980s, the surplus has decreased to the same level as in the early 1970s. The observed reduction in nitrogen surplus (Fig. 1) and leaching (Fig. 2) is due to a combination of improved utilization of animal manure (demands today are up to 75% utilization compared to no requirements just 20 years ago) and a reduction in the use of artificial fertilizer. The amount of nitrogen from artificial fertilizer in Denmark was almost halved between 1990 and 2002 (The Danish Plant Directorate, 2003). In the same period, average reductions in EU were only around 10% (The Danish Plant Directorate, 2003).

Fig. 1 Changes in the annual nitrogen surplus from Danish agriculture in the period 1900–2003 (based on data from The Danish Plant Directorate, 2003)



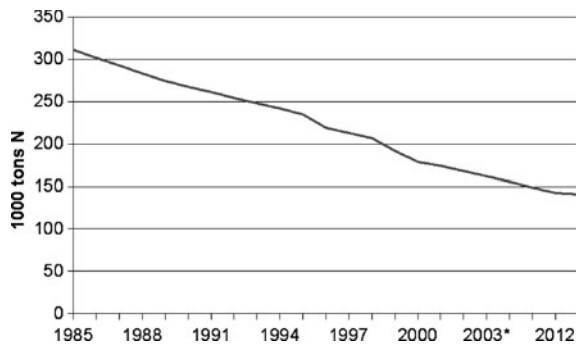


Fig. 2 Development in modeled nitrate leaching from the root zone in cultivated catchments in Denmark, 1985–2003, with prognosis until 2012 [based on data from Grant et al. (2006) and the expected further reduction until 2012 (Action Plan III)]

The large reduction in nitrogen leaching from cultivated soils has induced significant improvements in the nutrient status of lakes, watercourses, and coastal waters (Andersen et al., 2004; Lauridsen et al., 2005; Carstensen et al., 2006).

Nitrogen concentrations in watercourses with a high degree of cultivated soils in their catchments were reduced by about 29% from 1989 to 2003, and there has been a significant decrease in nitrogen concentrations at 50 out of 63 monitoring stations (Andersen et al., 2004). The lower nitrogen concentrations have subsequently led to reduced loads of nitrogen to lakes.

The nitrogen load to the marine environment decreased 43% in the period 1989–2003 (corrected for annual differences in runoff) (Andersen et al., 2004). In addition, Carstensen et al. (2006) showed that the positive changes in nutrient status in the Danish marine areas are statistically significant.

The observed and statistically verified reductions in nitrogen and phosphorus concentrations have led to only minor improvements at the ecosystem level (Andersen et al., 2004; Carstensen et al., 2006). Possible suggested explanations for the limited impact on biota are time lags, changes in basic physical and/or chemical conditions (nutrient pools, sediment texture, and hypoxia), as well as alternative steady-states in ecosystem structures (Scheffer, 1998).

Benefits have not been without costs

The agricultural sector in Denmark has contributed to the reversal of eutrophication in Danish waters

during the past 20 years. By changing their practice, individual farmers have contributed to the decrease in nutrient loads, which has proved to be a success in what is visible to the eye as well as to the statistical experts. Many of the measures introduced have been based on sound science and understanding—improved utilization of nitrogen in manure and requirements for a certain storage capacity for slurry, to mention just a few. However, the use of general regulations applied equally to all farmers regardless of their actual environmental impact has reached its limits—and in some cases well exceeded them.

The second Action Plan from 1998 introduced a fixed reduction in the nitrogen quota that each farmer can apply to their fields of 10% below the economic optimum in terms of yield without any compensation to the farmer. This new national quota of nitrogen was fixed with regard to the existing types and distribution of crops in 1998. Changes in crop distribution since 1998 toward improved breeds with higher yields and larger nitrogen demands have caused the gap between the allowed and optimal yield to grow from 10% up to almost 15%. This restriction is not only expensive in terms of low yields compared to farmers in other countries, but is also resulting in poor quality crops (low protein content due to a lack of nitrogen). This is surely not a strategy to be recommended in the future, neither in Denmark nor in other countries that are about to reduce nutrient losses from agriculture.

Future management of aquatic habitats—an urgent need for re-thinking

As mentioned in the previous section, general regulation is only feasible and acceptable to a certain extent. In Denmark, the usefulness of this approach is outdated as the simple solutions have already been used. Environmental managers must put their habitual thinking aside and look ahead for new options. With the Water Framework Directive, we are facing new challenges and possibilities. Hopefully, a new approach will replace the present old-fashioned strategy with equal rules and requirements for all areas, regardless of their individual impact on the environment. It does, however, require the ability to quantify environmental impacts on a smaller scale

with reasonable certainty, which is by no means a trivial task.

Danish Agriculture calls for the allocation of efforts in the right areas, both regarding sectors and on a geographical (micro-) scale. In addition, it is very important to have a holistic view, with focus on a wider range of measures than today and inclusion of cumulative effects (one measure in one specific area will often add to improvements in other areas).

The following list contains a set of points that, from our point of view, are very important in future work (the list only contains some suggestions, and is not all-inclusive):

- detailed cost–benefit and cost-effectiveness analyses must be made before measures are selected and implemented;
- measures must be implemented only after very thorough sensitivity mapping and analysis;
- time lags must be taken into account when judging the impact of implemented measures;
- reference conditions should not be the ultimate target—some degree of human activity must be recognized and accepted;
- the fact that conditions are changing continuously (stable baseline conditions do not exist) should be accepted;
- measures should target load and impact; and
- the agricultural sector should be actively involved in the early planning process: win–win situations and ownership will appear.

It is very important to focus on detailed and specific measures. Therefore, sensitivity mapping and precise modeling tools are essential in order to reach the objectives—and to set and adjust the objectives at the right level (e.g., Ringkøbing County, 2004).

The general lack of significant improvements in ecological status after the reductions in nutrient inputs and concentrations that have been observed in most cases (e.g., Ærtebjerg et al., 2005), and discussed on several occasions during the Eutro2006 Symposium in Nyborg in June 2006, is often due to profound changes in a number of basic conditions such as changes in sediment texture, hypoxia, internal loading from nutrient pools in the sediments, changes in hydrology, physical modifications, and climate. The dynamic nature of basic conditions and a need for socio-economic considerations imply that environmental

objectives should be set with reference to both the existing and predicted future conditions.

If site-specific environmental impacts are adopted in the WFD implementation, extensive cost–benefit will be needed. Prioritizing between measures considering cost–benefit and environmental impacts at the local scale will be challenging for local administrations, which at present are not equipped to perform such in-depth evaluations. The risk of generalized rules is high if the quality of environmental impact assessment varies significantly.

Finally, Danish Agriculture calls for a close cooperation between managers and the agriculture sector. It is very important to have involvement at early stages in the planning process to ensure the best solutions for all parties. Participation will lead to ownership of the project, and thus create win–win situations.

References

- Ærtebjerg, G., J. Bendtsen, J. Carstensen, et al., 2005. Marine områder 2004—tilstand og udvikling i miljø-og naturkvaliteten. National Environmental Research Institute. Report no. 551 (in Danish with English summary).
- Ærtebjerg, G., J. H. Andersen & O. S. Hansen (eds), 2003. Nutrients and Eutrophication in Danish Marine Waters. A Challenge for Science and Management. National Environmental Research Institute (Available via <http://eutro.dmu.dk>).
- Andersen, J., M. S. Boutrup, L. M. Svendsen, J. Bøgestrand, R. Grant, J. P. Jensen, T. Ellermann, G. Ærtebjerg, L. F. Jørgensen & M. W. Pedersen, 2004. Vandmiljø 2004. Tilstand og udvikling—faglig sammenfatning. National Environmental Research Institute. Report no. 517 (in Danish).
- Carstensen, J., D. J. Conley, J. H. Andersen & G. Ærtebjerg, 2006. Coastal eutrophication and trend reversal: A Danish case study. *Limnology and Oceanography* 51: 398–408.
- Conley, D. J., S. Markager, J. Andersen, T. Ellermann & L. M. Svendsen, 2002. Coastal eutrophication and the Danish National Aquatic Monitoring and Assessment Program. *Estuaries* 25: 706–719.
- Danish Environmental Protection Agency, 1991. Environmental Impacts of Nutrient Emissions in Denmark. Redegørelse fra Miljøstyrelsen, No. 1.
- European Communities, 2000. Directive 2000/60/EC of the European Parliament and of the Council of the 23 October 2000 establishing a framework for Community action in the field of water policy.
- Grant, R., K. Nielsen & J. Waagepetersen, 2006. Reducing nitrogen loading of inland and marine waters—evaluation of Danish policy measures to reduce nitrogen loss from farmland. *Ambio* 35: 117–123.

- Knudsen, L., H. S. Østergaard & E. Schultz, 2000. Kvælstof—et næringsstof og et miljøproblem. Danish Agricultural Advisory Service (in Danish).
- Lauridsen, T. L., J. P. Jensen, M. Søndergaard, E. Jeppesen, A. Strzelczak & L. Sortkjær, 2005. Søer 2004. NOVANA. National Environmental Research Institute. Report no. 533 (in Danish).
- Ringkøbing County, 2004. Undersøgelser af Ringkøbing Fjord og opland (in Danish).
- Scheffer, M., 1998. Ecology of Shallow Lakes. Chapman & Hall, London.
- Statistics Denmark, 2005. Statistical Yearbook.
- The Danish Plant Directorate, 2003. Fertilizer Statistics: Consumption of Fertilizer in Denmark 2002/2003.

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

Jørgen Dan Petersen · Nanna Rask ·
Harley Bundgaard Madsen · Ole Tyrsted Jørgensen ·
Stig Eggert Petersen · Susanne V. Koch Nielsen ·
Cathrine Bøgh Pedersen · Mikael Hjorth Jensen

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 71–89.
DOI: 10.1007/s10750-009-9774-0 © Springer Science+Business Media B.V. 2009

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

Electronic supplementary material The online version of this article (doi:10.1007/s10750-009-9774-0) contains supplementary material, which is available to authorized users.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

The authors Jørgen Dan Petersen and Nanna Rask are no longer in operation with ‘Nature Management and Water Environment Division’.

J. D. Petersen · N. Rask · H. B. Madsen ·
O. T. Jørgensen · S. E. Petersen · S. V. Koch Nielsen ·
C. B. Pedersen · M. H. Jensen
Nature Management and Water Environment Division,
5220 Odense, SØ, Fyn County, Denmark

Present Address:

H. B. Madsen · O. T. Jørgensen · S. E. Petersen ·
S. V. Koch Nielsen · C. B. Pedersen · M. H. Jensen (✉)
Danish Ministry of the Environment, Environment Centre
Odense, 5220 Odense, SØ, Denmark
e-mail: mihje@ode.mim.dk

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 km² (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 km². The 16-km² inner, mesohaline part of the estuary (Seden Strand) has a mean depth of 0.8 m, while the 46-km² 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

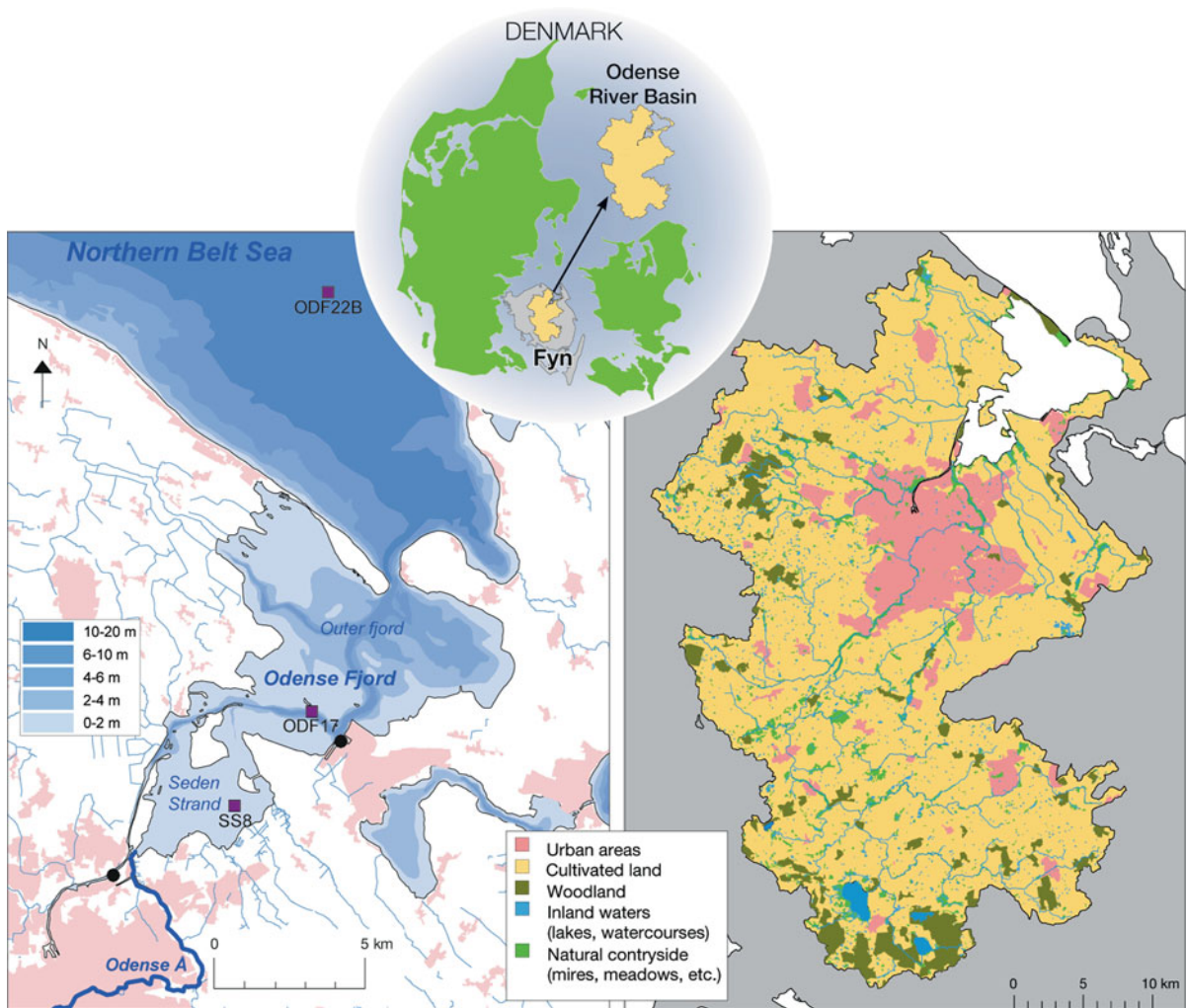


Fig. 1 Odense Fjord indicating the boundaries of the inner fjord (Seden Strand) and outer fjord (*left*). The monitoring stations SS8 in Seden Strand, ODF17 in the outer part of the

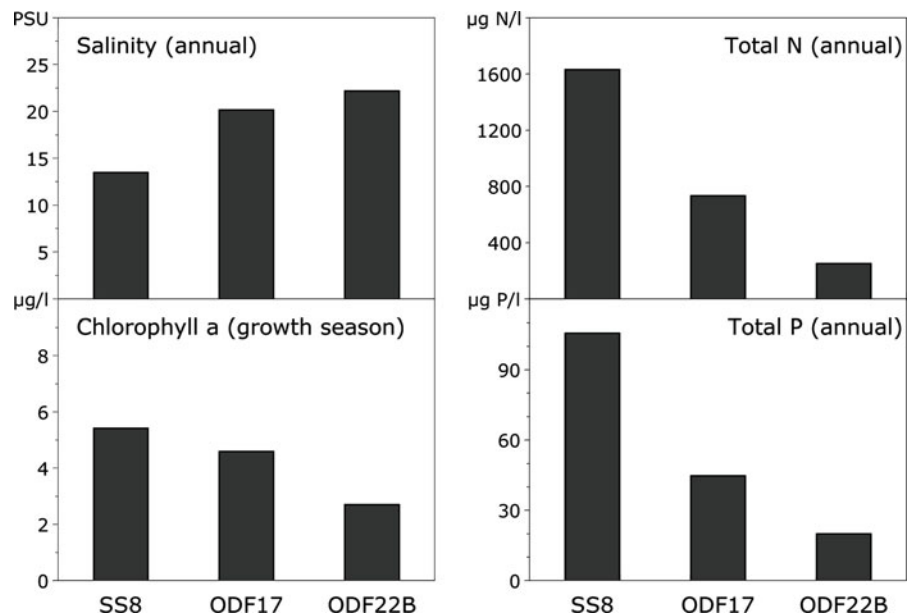
estuary and ODF22B in the border area outside the fjord are indicated. In the map on the right, the catchment area is shown with associated land use

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 m³ (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).

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



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

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

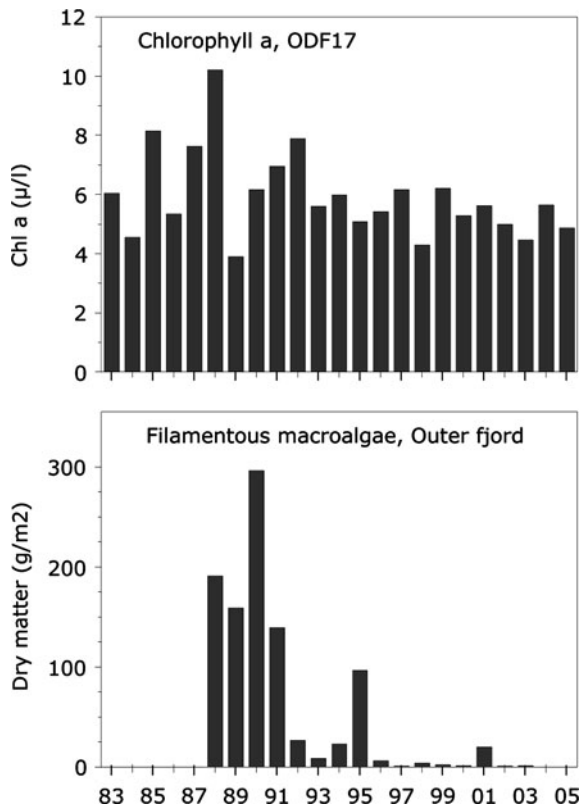


Fig. 3 Concentrations of chlorophyll *a* (May–October mean; stn. ODF17) (upper) and biomass of filamentous macroalgae during summer in the outer part of Odense Fjord (lower), 1983–2005

(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 filtering 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)

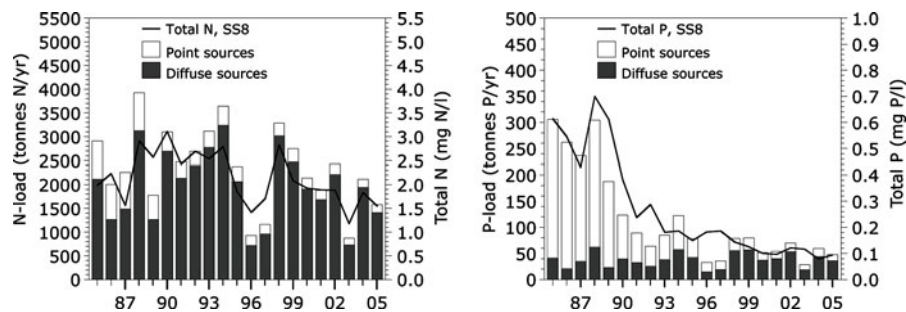
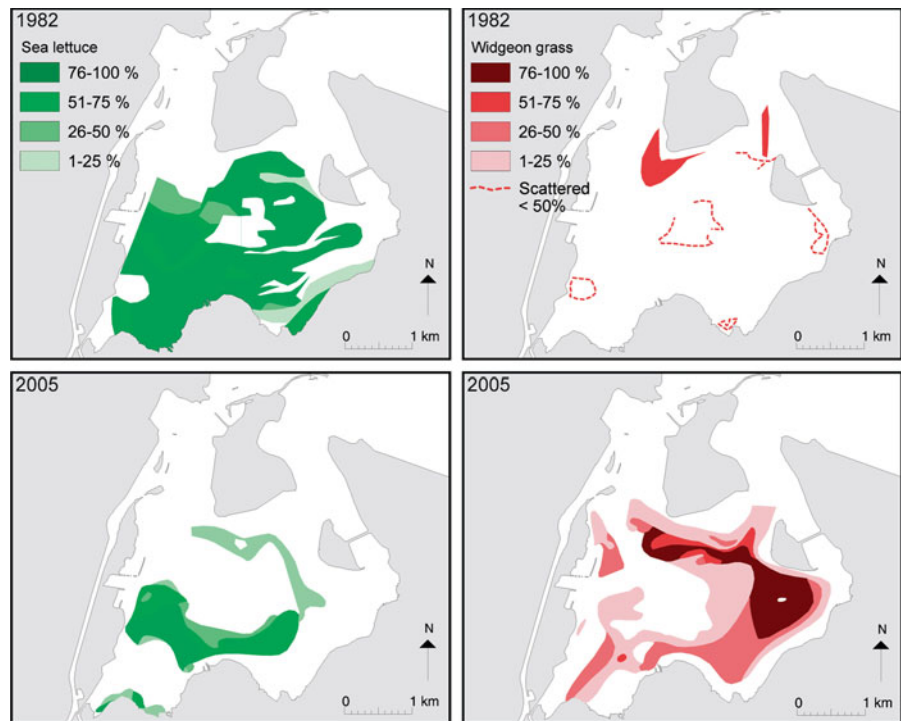


Fig. 5 Annual land-based nitrogen load (left) and phosphorus load (right) to Odense Fjord and annual mean total nitrogen and phosphorus concentrations at stn. SS8 in Seden Strand,

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, $\mu\text{g N l}^{-1}$ | | Total P, $\mu\text{g N l}^{-1}$ | |
|--|-------------------|-------------|-----------------|---------------------------------|-------|---------------------------------|-------|
| | 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:

$$\ln(Z) = 0.755 \times \ln(\text{TN}) + 6,039$$

$$r^2 = 0.547, n = 128 \quad (1)$$

$$Z = 0.787 \times \text{SD} + 0.339 \quad r^2 = 0.606, n = 101, \quad (2)$$

where Z is eelgrass depth limit (m), TN is total nitrogen ($\mu\text{g N l}^{-1}$) 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 $\mu\text{g N l}^{-1}$ 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

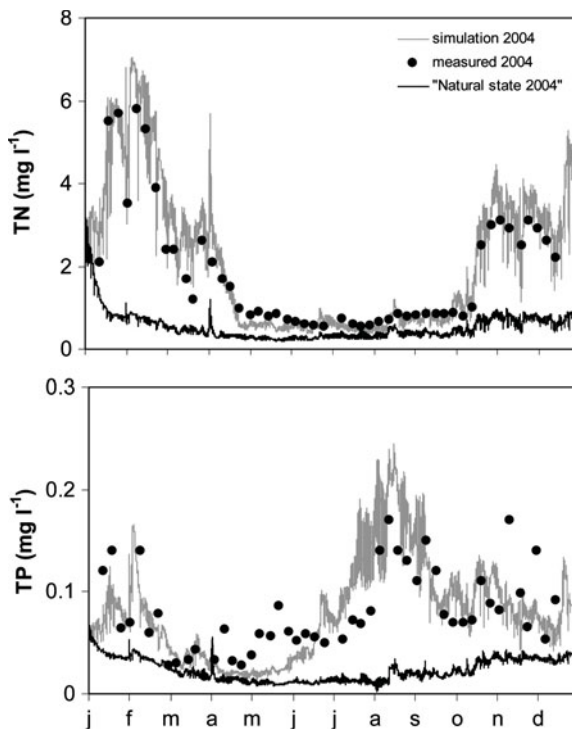


Fig. 6 Seasonal variation of total nitrogen and total phosphorus in MIKE3 model runs for 2004 (simulation 2004) shown together with measured values (measured 2004). Model output from the ‘Natural state 2004’ scenario run with reference nutrient loading (see text) is also shown

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: $29 \text{ } \mu\text{g P l}^{-1}$ at stn. SS8 in the inner fjord and $22 \text{ } \mu\text{g P l}^{-1}$ 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 $320 \text{ } \mu\text{g N l}^{-1}$ at stn. SS8 and ODF17, respectively. This corresponds

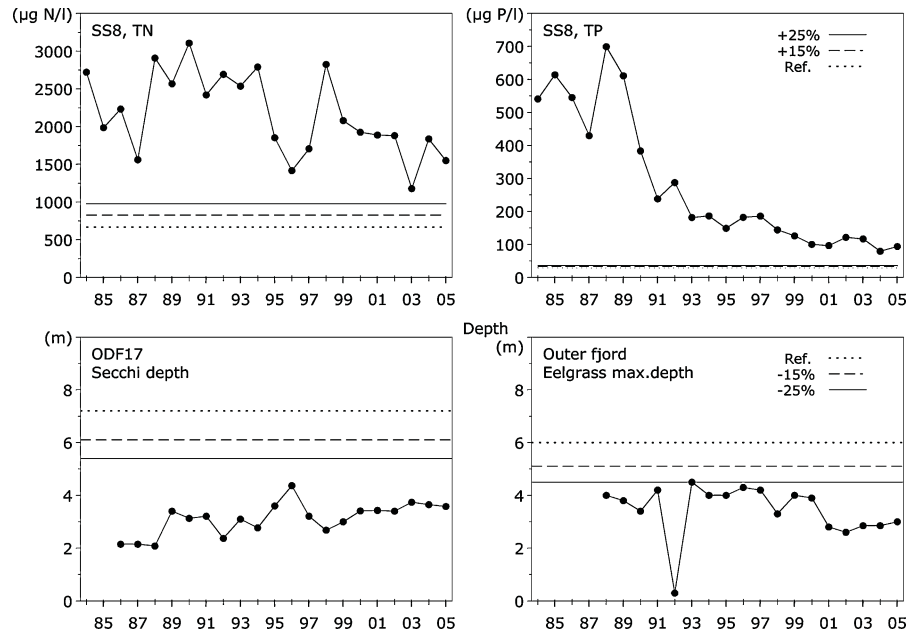
well with the above-mentioned 666 and $374 \text{ } \mu\text{g N l}^{-1}$ (where a combination of historical eelgrass data and empirical modelling was used), bearing in mind that the estimates are based on independent and very different approaches. We used the latter nitrogen data as reference conditions 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

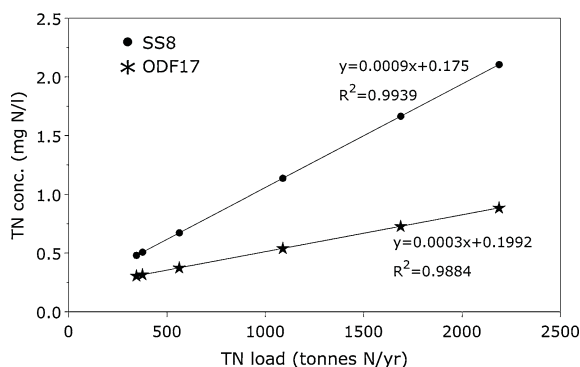


Fig. 8 Annual surface water concentration of total nitrogen in the inner (stn. SS8) and outer (stn. ODF17) parts of Odense Fjord as a function of the annual N load of Odense Fjord using various scenario runs of the MIKE 3 model (model year: 2004)

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 hydro-morphological 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/ivers | 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 ~50 and ~1 tonnes per year, respectively (11 largest lakes) |
| Coastal waters (Odense Fjord) | 100 | Nutrient loads from agriculture, hazardous substances | Reduced N load of ~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 ^{-1a} |

^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.

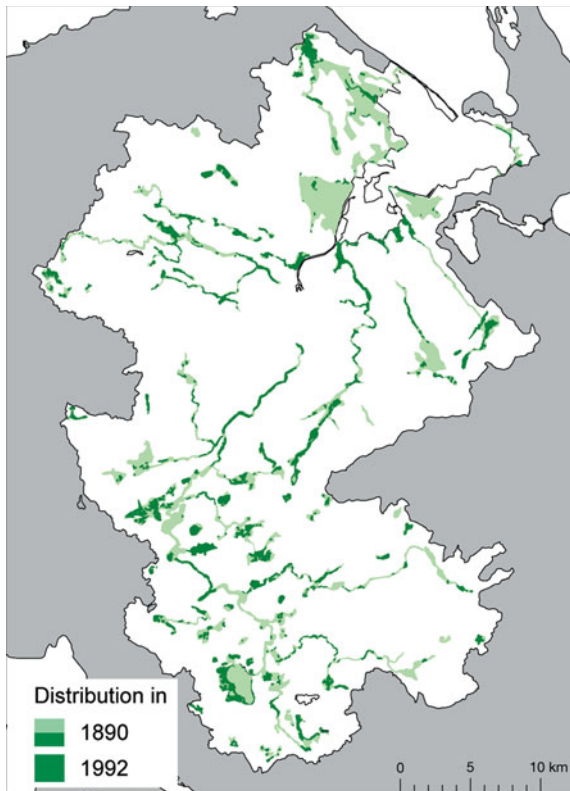


Fig. 9 Distribution of wetlands (meadows and bogs) in Odense River Basin, 1890 (sum of light and dark green) and 1992 (dark green only)

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 sub-basins, 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-km² 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.

Acknowledgements All staff at the (now no longer operating) Nature Management and Water Environment Division, Fyn County, are thanked for their skilled efforts over many years. The division won the 2004 Swedish Baltic Sea Water Award due to its "outstanding professional standard, visionary strategies and strong commitment for nearly 30 years". For this article, thanks are due to Birte Vindt, Inga Husted Holm, Inge Møllegaard and Flemming Nørgaard for help with the graphic work and data treatment.

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 2000/60/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 Rabalais, 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. Nærings saltbegrænsning 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. Sømmod, 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, København (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. Klimaforandrings 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 O₂ 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.

Relationship between eutrophication reference conditions and boundary settings considering OSPAR recommendations and the Water Framework Directive—examples from the German Bight

Dilek Topcu · Uwe Brockmann · Ulrich Claussen

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 91–106.
DOI: 10.1007/s10750-009-9778-9 © Springer Science+Business Media B.V. 2009

Abstract In order to allow for natural variability, the original OSPAR assessment procedure for eutrophication (Comprehensive Procedure) sets the threshold between Non-Problem/Problem Area (elevated levels) at 50% above natural background concentrations, which is equivalent to the boundary setting good/moderate for the EU Water Framework Directive (WFD). The 50% level corresponds to the recent natural variability of nutrient gradients in coastal and estuarine waters in the German Bight. Based on this threshold, a proposal is given for the additional boundary settings required for the WFD assessments. Examples, based on concentrations of total nitrogen and other correlated eutrophication components, are presented. However, for eutrophication effects such as oxygen deficiency, reduced transparency and increased transboundary loads, especially for offshore

regions, 50% exceedance of the natural background surpasses ‘slight differences’ as recommended by the boundary good/moderate for the WFD. For this reason, 15% is proposed as the boundary setting for good/moderate and discussed for different parameters. Overlapping between recent means and their standard deviations and the four boundary settings for the WFD cannot be avoided, thereby causing weak assessments. Since the part of the variability of recent data is caused by hydrodynamics, coupled with salinity variation, the variability could be reduced to some degree by relating the data to mean salinities. By doing this, the significance of classifications could be improved. The application of this procedure is discussed for examples from the German Bight.

Keywords Eutrophication · Assessment · Boundary setting · Reference conditions · Nutrients · Oxygen depletion

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

D. Topcu (✉) · U. Brockmann
Institute for Biogeochemistry and Marine Chemistry,
Hamburg University, Martin-Luther-King-Platz 6,
20146 Hamburg, Germany
e-mail: dilek.topcu@uni-hamburg.de

U. Claussen
German Federal Environmental Agency, Wörlitzer Platz 1,
06844 Dessau, Germany

Introduction

Eutrophication assessments need boundary settings related to natural background conditions that allow, as far as possible, precise, transparent effect-related results, which are required for focused reduction measures.

Boundary settings should primarily be based on the different stages of eutrophication effects, such as

the degree of effects (concentrations, extensions, durations and frequencies) in relation to natural background concentrations. However, the ability to relate the effects quantitatively to causative factors is limited owing to the complexity and variability of interacting processes.

For this reason, a pragmatic first approach was followed by OSPAR for the definition of a general threshold between 'Non-Problem Area' (NPA) and 'Problem Area' (PA) of 50% above the regional specific background concentrations as 'elevated level', "in order to allow for natural variability" (OSPAR, 2005). Variability around 50% is often observed for recent data in transitional and coastal waters. OSPAR and the EU Water Framework Directive (WFD) use similar parameters and basic boundary settings for assessing the eutrophication status. The threshold between Non-Problem Area and Problem Area was suggested by OSPAR as a definition for Quality Objectives and also as the boundary between 'good' and 'moderate' for the WFD (European Communities, 2000; OSPAR, 2004a). This boundary can be seen as a 'key boundary' because by surpassing this value measures are required to improve the environment, controlled by sufficient monitoring (European Communities, 2000; OSPAR, 2005).

Since for the OSPAR assessment only one threshold is used, the significance of the deviation of recent data from this threshold has been discussed extensively (OSPAR, 2004b). However, by introducing the WFD with four boundary settings, overlaps with recent means and their standard deviations are difficult to avoid along observed gradients, causing less significant deviations and assessments. However, focusing on the main (key) boundary setting between good and moderate, the significance requirements for the WFD are similar to OSPAR assessment criteria. In order to keep the variability as low as possible, data may be normalized to specific salinities, thereby reducing hydrodynamic influences.

However, it is evident that the 50% deviation can represent much more than the proposed WFD 'slight differences' from natural background conditions. 'Slight differences' would be something around 20% above natural background values and accordingly 15% or 25% as the boundary setting between good and moderate has been discussed already, but related to high reference values (Andersen et al., 2004).

Fig. 1 Map of recent concentrations of TN (μM) (*top left*) and associated standard deviations with salinity of 31 indicated (*top right*) in the German Bight, divided into regular squares of 145 km^2 , with mixing diagrams for TN concentrations (*middle*) and standard deviations (*bottom*) for surface waters in all the seasons (1998–2002)

Therefore, in addition to the key boundary setting of 50% above natural background conditions, 15% will be discussed as a key boundary for different eutrophication processes, considering effect relationships and data variability.

Boundary settings

Natural background concentrations that reflect pristine conditions, not affected by anthropogenic activities (European Communities, 2000), are the basis for boundary settings. The most important boundary setting between good and moderate (transferable to Non-Problem/Problem Area) is defined as 'slight deviation' from reference conditions.

The OSPAR approach, using 50% above background concentrations as the threshold, is still suitable for the assessment of the inner German Bight because recent (1998–2002) mean nutrient and chlorophyll concentrations reflect a regional variability of about 50% as standard deviation. As an example, the surface concentrations of total nitrogen (TN) are shown (Fig. 1, top). For the estimation of the regional variability, the inner German Bight was divided into regular squares of 145 km^2 size. This is in the range of 10% of the maximum extension, allowing a corresponding precision of assessments. For each square, means and standard deviations were calculated.

The recent mean concentrations of TN were less than $20 \mu\text{M}$ offshore, increasing to more than $200 \mu\text{M}$ in the estuaries. A salinity of 31 indicates roughly the border between the coastal area to be assessed according to the WFD and the offshore area, both assessed by OSPAR.

The variability surpassed 50% (as standard deviation) mainly along the border of the coastal water, caused by the permanently changing shapes and extensions of river plumes, and was especially high around a salinity of 30 as shown by the mixing diagram with means for the different squares and the standard deviations plotted against the salinity (Fig. 1, bottom). The reasons are the low TN concentrations and the changing gradients along the river plume fronts.

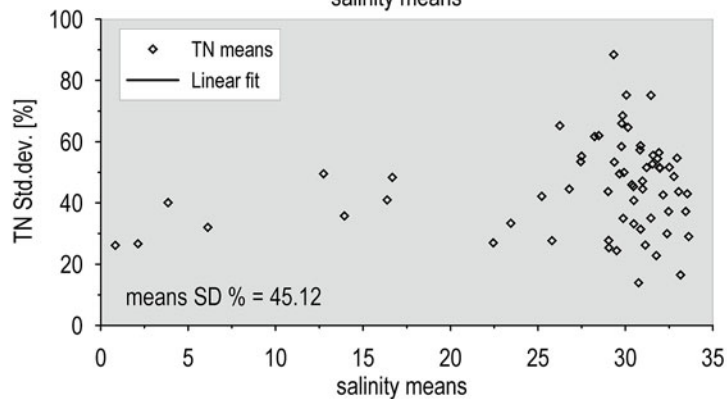
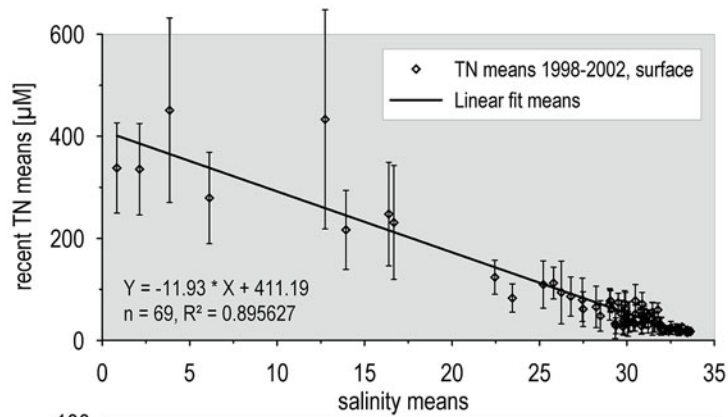
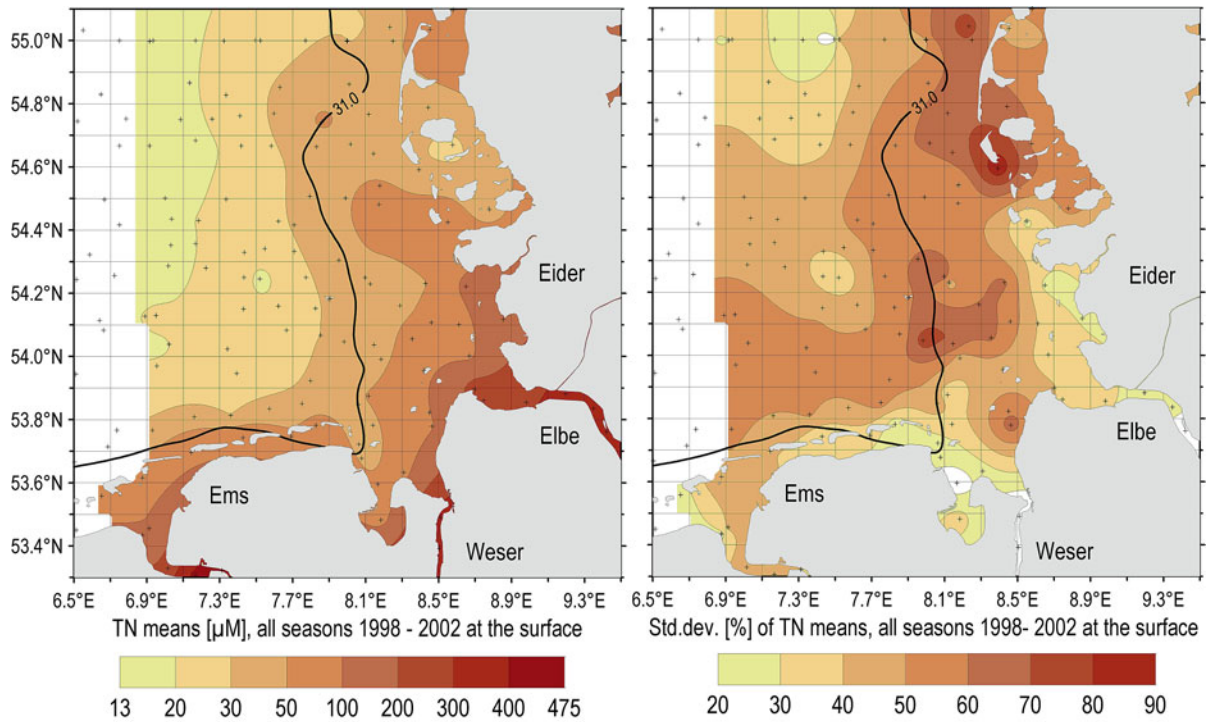


Table 1 Scheme of OSPAR/WFD thresholds, based on a 50% key threshold

| OSPAR WFD class | Further assessment Reference | Non-problem area | | Problem area | | |
|--|------------------------------------|------------------|-------------------------|-------------------|-----------------|------------|
| | | High | Good | Moderate | Poor | Bad |
| % Deviation from background | 0 | <17 | 17– 50 | 50 –150 | 150–450 | >450 |
| EQRs [100/(100 + deviation)] | 1.0 | >0.85 | 0.85–0.67 | 0.67–0.40 | 0.40–0.18 | <0.18 |
| TN [μM] | 11.1 | <13 | 13–17 | 17–28 | 28–61 | >61 |
| Oxygen saturation [%] | 87 | >84 | 84–80 | 80–66 | 66–25 | <25 |
| O ₂ depletion effects (Table 3) | | No effects | 80% min. requirement | 70% first effects | 50% first kills | <40% kills |

EQR: Ecological Quality Ratio

The annual changing riverine nutrient discharges to the German Bight vary between 20% and 60% (Pätsch & Lenhart, 2004). Therefore, it is recommended to apply a threshold of 50% above the reference value for increasing eutrophication effects as a first approach in nearshore waters (Table 1) until further refinement, for example by reducing the variability of recent data, is achieved.

The 50% deviation from background concentrations corresponds to the high/good boundary of the WFD (European Communities, 2000). The natural background conditions are indicated as reference conditions for the boundary settings. By OSPAR, these values are defined as ‘high’ status.

Successive boundary settings, which can also be defined as percentage deviations from reference conditions, are proposed according to the WFD requirement of five classes. The differences between these boundary settings were simply defined on a regular basis to keep the procedure as transparent as possible. A factor of 3 for stepwise deviations was chosen, considering the range of current nutrient gradients in the estuaries of the German Bight. Boundary settings between high and good are proposed to be 17% above reference (a third of 50%), those of moderate/poor and poor/bad are taken as 150% (three times 50%) and 450% (nine times 50%) above natural background (Table 1, line 2).

By dividing the reference conditions (= 100%) by the percentages of the thresholds in relation to the reference, the corresponding Ecology Quality Ratios (EQRs) were calculated (0.85–0.18), considering the recommendations that the ‘high’ class is close to 1 and the ‘bad’ class approaches 0 (European Communities, 2002) (Table 1, line 3).

TN concentrations are presented as the master variable, with a reference concentration of 11.1 μM for the offshore area (recent mean surface concentration in the central North Sea at salinities between 34.5 and 35), due to its manifold quantitative relationship to other eutrophication parameters, such as chlorophyll (Nielsen et al., 2002b; Tett et al., 2003; Smith, 2006). The rounded boundary settings are correspondingly 17 μM (50% above reference) for good/moderate, 28 μM for moderate/poor and >61 μM for bad conditions (Table 1, line 4).

Oxygen depletion occurs occasionally during summer in the stratified part of the German Bight (Dethlefsen & von Westernhagen, 1983; Rachor & Albrecht, 1983; Brockmann & Eberlein, 1986). Thermal stratification starts mostly in May (Tomczak & Goedecke, 1964) and initial oxygen depletion is observed in July. Assuming simple stationary conditions, the oxygen demand was calculated for the TN concentrations at the thresholds, equivalent to carbon and nitrogen, by applying the Redfield ratio C:N = 6.75 (Redfield et al., 1963) by which 1 μM N is equivalent to 0.28 mg O₂ l⁻¹, and by using an efficiency factor of 0.4 for the German Bight: mg O₂ l⁻¹ = 0.28 μM TN \times 0.4.

The factor of 0.4 was introduced based on observed correlations between particulate organic carbon and oxygen depletion in the bottom waters. It is evident that this efficiency of 40% includes influences of residence time, diffusion and exchange processes. The resulting oxygen demand is subtracted from 9.1 mg l⁻¹ oxygen concentration which is equivalent to 100% saturation at a mean salinity of 34 and assumed 10°C in the bottom water (Table 2). The resulting potential oxygen consumption, equivalent to

Table 2 TN/Oxygen conversion (example for a 50% key boundary setting = 150% of background)

| % | $\mu\text{M TN } \%$ in col. 1 | $\text{mg O}_2 \text{ l}^{-1}$ ($\mu\text{M TN} \times 0.28$)* | $\times 0.4 \text{ mg O}_2 \text{ l}^{-1}$ ** | Diff from $9.1 \text{ mg O}_2 \text{ l}^{-1}$ *** | Sat. % |
|-----|-----------------------------------|---|---|--|--------|
| 100 | 11.1 | 3.1 | 1.24 | 7.86 | 87 |
| 117 | 13.0 | 3.64 | 1.46 | 7.64 | 84 |
| 150 | 16.7 | 4.68 | 1.87 | 7.23 | 80 |
| 250 | 27.8 | 7.78 | 3.11 | 5.99 | 66 |
| 550 | 61.1 | 17.1 | 6.84 | 2.26 | 25 |

* According to the Redfield ratios, ** corresponding to recent relations (see text), *** $9.1 \text{ mg O}_2 \text{ l}^{-1}$ corresponds to oxygen saturation (at a salinity of 34 and 10°C)

the TN thresholds, was transferred to saturation (%) according to Benson & Krause (1984) (Table 1, line 5).

The natural background concentration of $11.1 \mu\text{M TN}$ corresponds to $1.2 \text{ mg O}_2 \text{ l}^{-1}$ consumption in the German Bight, resulting in $7.9 \text{ mg O}_2 \text{ l}^{-1}$ residual concentration = 87% saturation, assuming that only 40% of organic matter will be decomposed during these processes. A 50% deviation of TN concentrations from background values would correspond to 80% oxygen saturation in bottom waters.

However, comparing this value with the effect levels of oxygen deficiency, which have been extracted from the literature and compiled in Table 3, the proposed boundary settings are too high. Following the precautionary principle and considering that (i) monitoring of oxygen depletion in stratified bottom waters is usually not performed with a sufficient resolution in space and time and (ii) bottom water is sometimes warmer than 10°C (in the shallow German Bight up to 20°C), it is assumed that 80% oxygen saturation is already worse than good and more than a 'slight deviation' from background because 80% is the minimum requirement for fish cultivation. Therefore, values below 85% should be assessed already as moderate (Table 4).

At 60–70% oxygen saturation significant effects have been observed, considering increased surface/volume reactions of sea-cucumbers as a general indicator. For this reason, 76% is recommended as the boundary between moderate and poor. First kills occur below 50% oxygen saturation, which should be classified at least as poor. Because at around 33% saturation extended mortality of macrofauna was observed in the German Bight (Rachor, 1983; Bauerfeind et al., 1986; Niermann et al., 1990), 35% should be set as the poor/bad boundary. These field

observations of severe oxygen depletion effects reflect mainly late stages of longer-lasting eutrophication processes, and are therefore only a rough indicator for setting effect-related boundaries.

For these reasons, 15% deviation from the natural background is proposed as an alternative boundary setting for good/moderate, especially for offshore areas where the occurrence of oxygen depletion indicates significant eutrophication (Table 4). Applying again regular steps, now with an assumed factor of 5, a similar differentiation as for the 50% approach can be achieved for the successive boundaries. However, the boundary for high/good, calculated by this scheme as 3%, has been defined instead as 7%, about half of the boundary for good/moderate, in order to take natural variability at low concentrations into account.

The key boundary setting of 15% results in a TN of $12.8 \mu\text{M}$ and potential O_2 depletion of 85% saturation. This is significantly above the first effect level of 80% saturation and is therefore proposed as the boundary for good/moderate. Since several different effects have been observed at 60–70% oxygen saturation, the suggested 75% above reference for the border between moderate/poor, corresponding to $19 \mu\text{M TN}$ or 76% oxygen saturation, is sufficiently above this value (Table 4). A TN concentration of $53 \mu\text{M}$, corresponding to 375% above the reference which is proposed as the boundary setting between poor and bad, is equivalent to an oxygen depletion of 35% in the German Bight. This degree of oxygen depletion causes many lethal effects (Table 3).

The Secchi depth is another sensitive assessment parameter, especially in shallow coastal areas with flat-bottom slopes, because it controls the extension of macrophytes which are an important ecosystem

Table 3 Effects of oxygen deficiencies

| Oxygen saturation (%) | Organisms | Observed responses | Comments | Literature |
|-----------------------|---|---|---------------------------------|--------------------------------------|
| 1 | Oligochaetes, polychaetes, molluscs, | Kills | | Gamenick et al. (1996) (G) |
| 1, 6 | <i>Crangon crangon</i> | Kills | | Hagerman & Vismann (1995) (G) |
| 2 | Different invertebrates | Kills | Experiments, North & Baltic Sea | Theede et al. (1969) (D) |
| 5–30 | <i>Platichthys flesus</i> , flatfish | Mortality, reproduction | | Tallqvist et al. (1999) (G) |
| 10, 19 | Fish & benthic organisms | Kills | German Bight | Dethlefsen & von Westernhagen (1983) |
| 10–30 | Macrozoobenthos | Damage of benthos, fish diseases | German Bight, Central North Sea | Rachor (1985) |
| 12 | <i>Crangon crangon</i> | Changed behavior | | Hagerman & Vismann (1995) (G) |
| <13 | Polychaetes | Kills | Limfjorden (Denmark) | Jørgensen (1980) (D) |
| 13 | <i>Amphiura filiformis</i> , Echinoderm | Emerging | Experiments | Rosenberg et al. (1991) (D) |
| 14 | Algae | Reproduction reduced | | Peckolt & Rivers (1995) (G) |
| 14, 22 | Polychaetes | Emerging | Chesapeake | Pihl et al. (1992) (D) |
| 14, 28 | Fish | Migration | Chesapeake | Pihl et al. (1991) (D) |
| 16, 29 | <i>Mytilus edulis</i> | Delayed development | | Wang & Widdows (1991) (G) |
| 18 | Bottom fauna | Kills, emerging | German Bight | Niermann (1990) |
| 25 | Zooplankton | Kills | Experiments | Roman et al. (1993) |
| 25 | <i>Nephrops norvegicus</i> (crustacean) | Emerging from tubes or burrows | | Baden et al. (1990) (D) |
| 25–43, 50 | Macrofauna | Weakening, mortality | German Bight | Rachor (1983) |
| 25–40 | Cod, whiting | Migration | | Wu (2002) |
| 33 | Macrofauna | Decrease | German Bight | Niermann et al. (1990) |
| 31–33 | Macrofauna | Mortality, decrease of abundance & richness | German Bight | Bauerfeind et al. (1986) |
| 31 | Most benthic organisms | Survival | 10°C, salinity 34 assumed | Wu (2002) |
| 33–44 | <i>Monoporeia affinis</i> (Crustacean) | Activity reduced | | Johansson (1997); Wu (2002) |
| 35 | <i>Saduria entomon</i> | Reduced consumption | of <i>Bathyporeia pilosa</i> | Sandberg & Bonsdorff (1996) |
| 36–60 | Freshwater fish | Kills | At 20°C | Missouri, 2005 |
| 37–48 | Crustaceans | Kills | Experiments | Stickle et al. 1989 |
| 38 | Atlantic sturgeon | Kills | Experiments | Secor & Gunderson (1998) |
| 45–73 | <i>Gadus morhua</i> | Activity & growth affected | | Chabot & Dutil (1999) (G) |
| 48 | <i>Penaeus setiferus</i> crustacean | Reproduction and metabolism affected | | Rosas et al. (1999) (G) |
| 50 | Sensitive fish | Avoidance, reduced growth | | Breitburg (2002) |
| 50–60 | Gobiid fish | Increased respiration Changed metabolism | | Petersen & Petersen (1988) |
| 50–80 | Different fish species | Blood undersaturation | | Davis (1975) |
| 54–63 | Fish & bottom fauna | Beginning of negative effects | 10°C, salinity 10 & 34 | Sedin (2002) |

Table 3 continued

| Oxygen saturation (%) | Organisms | Observed responses | Comments | Literature |
|-----------------------|--|--------------------------------------|------------------|---------------------------|
| 60–70 | <i>Holothuria forskalli</i> (cucumber) | Increased surf./vol. ratio | | Astall & Jones (1991) (D) |
| 62 | Cold-water fish | Effects on cold water fishes | 10°C, salinity 1 | Behar (1997) |
| 74 | <i>Salmo trutta</i> | Reduced kidney, increased hemoglobin | Experiments | Seager et al. 2000 |
| 78 | Fish | Minimum requirement | Bioassay | Fishdoc (2005) |
| 85 | <i>Sparus aurata</i> | Increase of ammonia toxicity | | Wajsbrodt et al. (1991) |
| 93 | Echinoderms | Reproduction affected | | Spicer (1995) (G) |

Data are partly transferred from reviews by Diaz & Rosenberg (1995) (D), Gray et al. (2002) (G)

Table 4 Scheme of OSPAR/WFD thresholds, based on a 15% key threshold

| OSPAR WFD class | Further assessment Reference | Non-problem area | | Problem area | | |
|----------------------------------|------------------------------------|------------------|-------------------------|-------------------|-----------------|---------------|
| | | High | Good | Moderate | Poor | Bad |
| % Deviation from background | 0 | <7 | 7–15 | 15–75 | 75–375 | >375 |
| EQRs [100/(100 + deviation)] | 1.0 | >0.93 | 0.93–0.87 | 0.87–0.57 | 0.57–0.21 | <0.21 |
| TN [µM] | 11.1 | <11.9 | 11.9–12.8 | 12.8–19 | 19–53 | >53 |
| Oxygen saturation [%] | 87 | >86 | 86–85 | 85–76 | 76–35 | <35 |
| O ₂ depletion-effects | | No effects | 80% min. requirement | 70% first effects | 50% first kills | <40% kills |
| Secchi depth [%] | 100 | >92 | 92–85 | 85–55 | 55–18 | <18 |
| Secchi depth [m] | 7.8 | >7.2 | 7.2–6.6 | 6.6–4.3 | 4.3–1.4 | <1.4 |
| Flow/content [kt N] | 6.7 | 7.2 | 7.2–7.7 | 7.7–11.5 | 11.5–32 | >32 |

EQR: Ecological Quality Ratio

component (Nielsen et al., 2002a). Similar to the findings of Nielsen et al. (2002a), the Secchi depth was correlated with TN concentrations in the German Bight during summer, resulting in the equation: $\ln [m] = -1.11 \times \ln [TN \mu M] + 4.72$. A pristine Secchi depth would be 7.8 m in the open German Bight, deduced from 11.1 µM TN.

The key good/moderate boundary of 12.8 µM TN (15% above background concentration) corresponds with 6.6 m to 85% of the reference depth, whereas 17 µM TN (50% above background concentration, Table 1) would result in 4.8 m which is only 62% of the reference Secchi depth. This is already far beyond good conditions or slight disturbance because this Secchi depth would restrict the possible extension of macrophytes significantly (Nielsen et al., 2002a), by about 30% in the shallow German Bight.

Further aspects for setting boundaries are the acceptable offshore concentrations, affected by long-distance transboundary transports, which are addressed also by OSPAR (2005). A rough budget calculation of N and phosphorus (P) fluxes in the German Bight (24,400 km², mean depth 19.5 m, 476 km³ volume) resulted in a mean content of 74 kt N for natural background concentrations of about 11 µM TN. Reflecting the contradictory results of recent hydrodynamic model exercises (Smith et al., 1996) a very simple dynamic approach was chosen, assuming a constant outflow of 3 cm s⁻¹ through the northern border at 55.1° N (Fig. 1) (Mittelstaedt et al., 1983), which is balanced by a constant inflow of 1.8 cm s⁻¹ along the larger western border at 6.5° E, in order to receive a closed budget of water masses.

Table 5 Mean annual budget of pristine TN and TP in the German Bight

| | Water [km ³ y ⁻¹] | TN [$\times 10^3$ t N y ⁻¹] | | | TP [$\times 10^3$ t P y ⁻¹] | | |
|-----------------------------|--|--|---------------|---------------|--|----------------|----------------|
| | | Pristine | 15% threshold | 50% threshold | Pristine | 15% threshold | 50% threshold |
| Transboundary import | 3198 | 497.0 | 572.0 | 746.0 | 71.4 | 82.1 | 107.0 |
| River discharges | 37 | 11.0 | 13.0 | 17.0 | 0.5 | 0.6 | 0.8 |
| Atmospheric input | ± 0 | 2.4 | 2.8 | 3.6 | – | – | – |
| Transboundary export | 3235 | 503.0 | 578.0 | 755.0 | 72.2 | 83.0 | 108.0 |
| Trapping/ remobilization | 0 (balanced) | 7.4 lost | 9.8 Lost | 11.6 lost | 0.3 net export | 0.3 net export | 0.2 net export |

The natural annual flow of nitrogen nutrients through the German Bight of 497 kt N y⁻¹ amounts to seven times the N content, corresponding to the budget of the water masses (Table 5). Allowing a boundary of 15% above background concentrations, the nutrient content of the German Bight will be exchanged nearly eight times per year. At a 50% boundary setting 10 times the N content would be transported through the German Bight. The difference of 249 kt N y⁻¹ above the background flow of 497 kt N y⁻¹ would correspond to three times the natural background nutrient content (74 kt N). Since the difference between the two boundary settings amounts to 174 kt N y⁻¹ which is 10 times the annual historical river discharges (17 kt N y⁻¹, assuming 50% as threshold), a 50% boundary setting for 'good/moderate' cannot be accepted. The reason is that a mean flushing time of about 8 weeks allows the passing nutrients and organic matter an extended participation in the local turnover processes, including the formation of blooms, transient accumulation and significant oxygen depletion within a similar time period.

Owing to the pattern of the residual current system in the southern North Sea (OSPAR, 2000), it can be assumed that a significant load of nutrients and subsequently produced organic matter originate from the southern and western coastal areas.

Differentiating boundary settings along salinity gradients

The relationship of pristine background concentrations to salinity gradients can be established analogous to mixing diagrams by combination and extrapolation of freshwater, coastal and offshore values (Fig. 2).

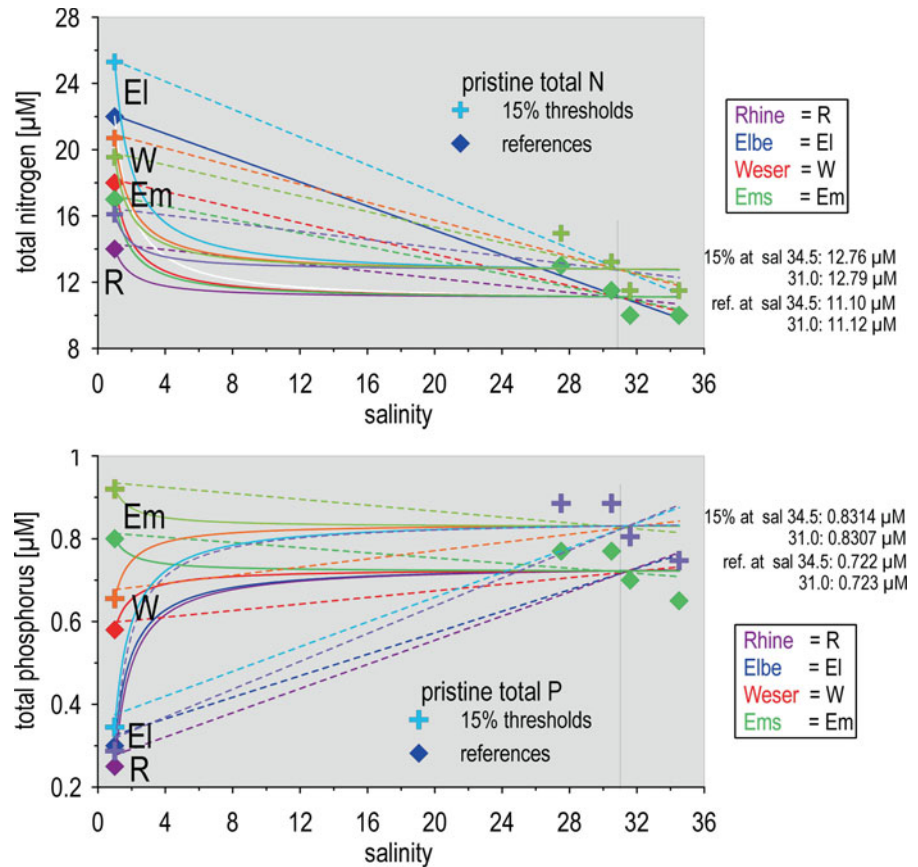
Pristine freshwater endmember concentrations were taken from a model (MONERIS), in which the different soil types were considered (Behrendt et al., 2003). Natural background concentrations for coastal waters are means of published extrapolated historical data (Zevenboom, 1994; Van Raaphorst et al., 2000) and for offshore waters recent means from the central North Sea were used. Natural background concentrations for freshwaters are river-specific. For TP, they are partly below reference conditions in the coastal water.

While linear mixing is the dominant process in the estuaries, nutrient recycling and different nutrient sources become more important in the open water, causing non-linear mixing. In order to approach offshore conditions with their unlimited dilution potential, hyperbolic fits of reference values were calculated and applied for higher salinities. The transition with linear fits is given by crossing points, at a salinity of 31 (Fig. 2). Consequently, for the calculation in the transitional and coastal waters, linear regressions were used up to a salinity of 31, above which the hyperbolic fits were applied, allowing a consistent estimation of background values.

The boundaries can be calculated correspondingly along the salinity gradients, adding 15% to the reference conditions. The final offshore values are fixed at 12.8 μ M TN and 0.83 μ M TP by the hyperbolic fits.

As an example, the application of the different approaches along the salinity gradients in the German Bight and the German exclusive economic zone is shown by the differences between recent (means 1998–2002) and pristine TN concentrations at the surface as percentages of pristine values (Fig. 3). The main salinities are indicated by isopleths. There is no inconsistency of classification at the salinity of 31, the border between linear and hyperbolic calculations

Fig. 2 Pristine mixing diagrams with thresholds for nitrogen (*top*) and phosphorus (*bottom*)



of reference conditions. Using the 50% key boundary approach, the areas assessed as high and good start already west of 7° E, while using the 15% key boundary they start only west of 6.5° E. Accordingly, the areas specified as bad and poor are also more extended by the latter approach. However, the shapes of the differently classified areas are similar.

Boundary settings and data variability

For the assessments, in addition to the representativity of the data, the significance of differences between gradients of recent data and boundary settings is important. As an example, for the variability of recent and pristine data affecting the precision of an assessment, a transect along the Elbe estuary and plume is shown, compiling means of the WFD types crossed and their standard deviations (Fig. 4).

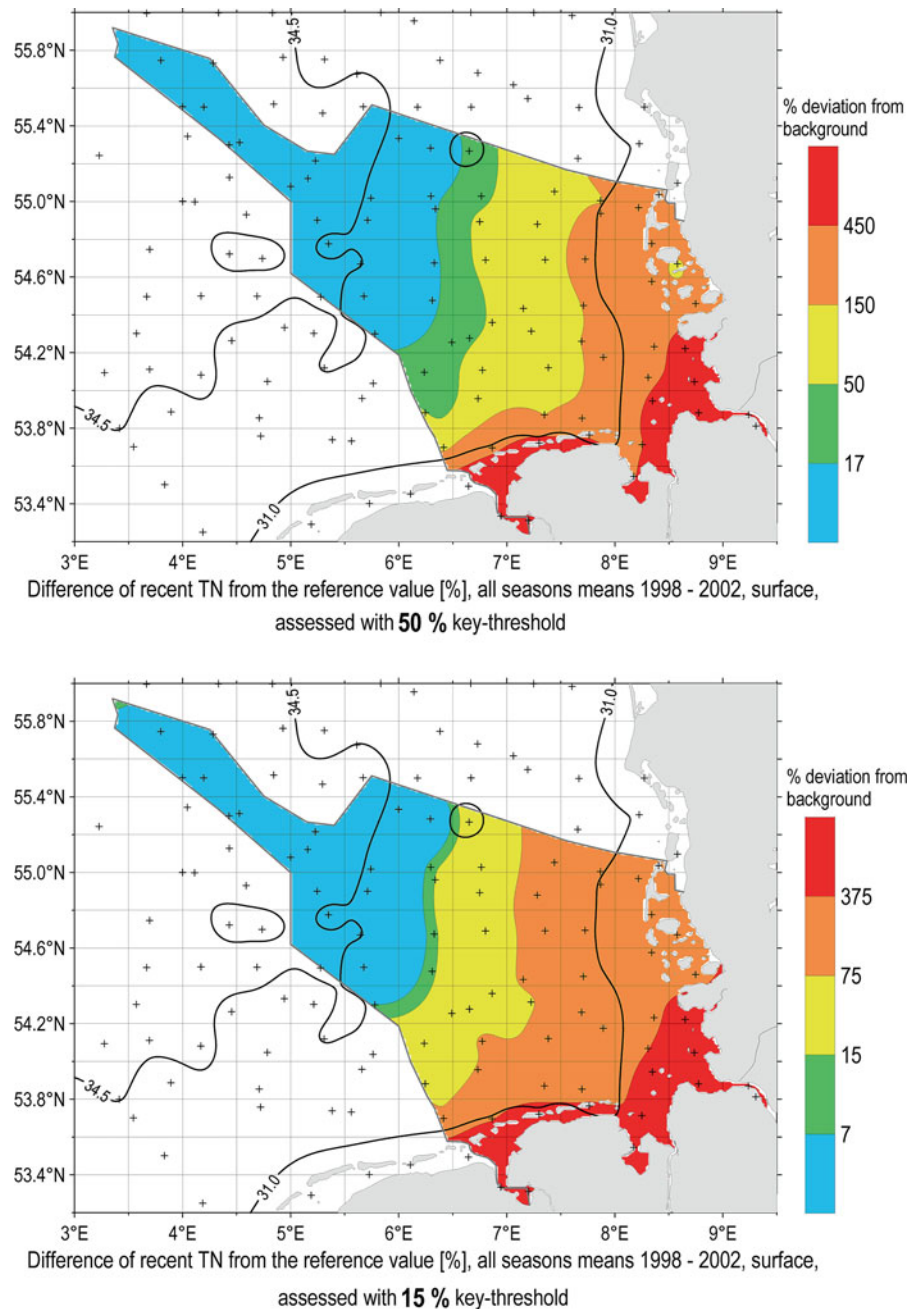
The transect begins in the inner estuary (Elbe 6, type definition transitional water according to the

WFD), passing a polyhaline moderately exposed type (Elbe 4), exposed coastal water (Elbe 3, Eider 3), and offshore coastal water of Schleswig-Holstein (SH) affected by the Elbe river plume. Reference conditions, boundary settings and recent data are indicated by means and their standard deviations.

Reference conditions and recent concentrations of TN decrease with increasing salinity in the area influenced by the Elbe. Means of recent data cross the boundary setting between bad and poor (at 375%) within the coastal water. Standard deviations of recent data are in the range of 100 µM (20–70%), while those of boundaries which were calculated in relation to recent salinity gradients are below 10 µM (5–10%).

Recent means in the estuary and the two types further downstream were significantly above the boundary at 375% (above natural background concentrations) between poor and bad. In the type Eider 3, the recent mean and its standard deviations were not significantly above this value and in the offshore

Fig. 3 Differences between recent (means of 1998–2002) and pristine concentrations of TN (μM) as % of pristine values, assessed with the 50% (*top*) and 15% (*bottom*) approaches



coastal water (SH) recent concentrations overlapped between the boundary settings poor/bad and moderate/poor. However, the locations of means in relation to the boundary settings can be used for the assessment, which is bad for the main part of this transect.

A 3-D plot is presented for a related area visualizing the overlapping between boundary settings and recent

mean concentrations with their variability. Looking from the coast northwest into the German Bight area, the plot shows the gradients of boundary settings and recent (1998–2002) mean TN concentrations with their standard deviations (Fig. 5). The variability (as μM) was especially high within the estuaries. Offshore, the recent data fell below the boundary setting for poor/bad and later below the moderate/poor boundary. The

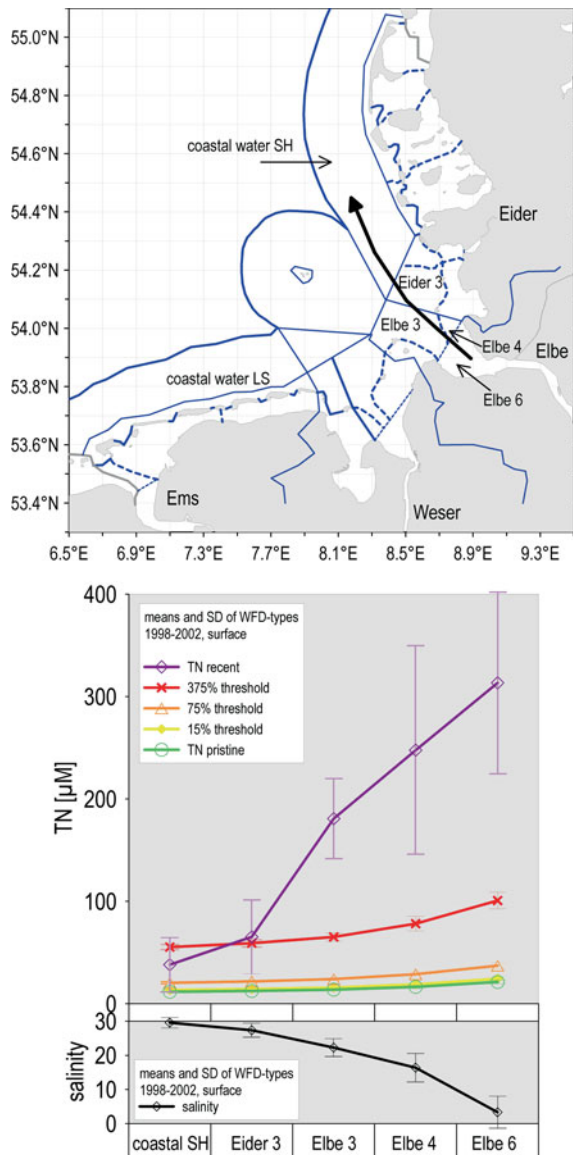


Fig. 4 Transect of recent mean TN concentrations (1998–2002), pristine values and thresholds of the 15% approach along the WFD types (indicated by blue lines in the map above) and water masses of the Elbe estuary and its plume

boundary setting high/good was not presented. In an enlarged section, the extent of overlapping in this area is evident.

Overlapping cannot always be avoided, especially not for four boundary settings, but it may be reduced to some degree by excluding the hydrodynamic variability, which is often coupled with salinity gradients. The reasons for the variability are, in addition to biogeochemical processes, the hydrodynamics that do not

need to be assessed. For this reason the variability, for example caused by moving river plumes, can be excluded, thereby reducing the overlapping to some degree. This dynamic approach is compared with a static analysis as presented in Fig. 1.

In order to reduce the influence of hydrodynamic variability, the German Bight was divided into the individual river plume mixing areas and, for each area, mixing diagrams were plotted (Fig. 6). The general mixing diagram of TN for the German Bight showed a significant negative correlation with salinity during the period 1998–2002. As an example, for the Ems estuary and the Ems plume area for each 1-PSU (Practical Salinity Unit) step the mean TN concentration and its standard deviation were calculated. The mean standard deviations were about 15% below those of the squares of 145 km² (Fig. 1).

For the salinity-related means and standard deviations, the mean geographic positions were calculated and the data transferred to a map. The resulting gradients were compared with the original TN distribution (Fig. 1) by the calculated differences (Fig. 7). Mainly the differences in concentrations were below 10%. By this procedure, the standard deviations could be reduced especially in the river plume mixing area by more than 20%. However, differences between original data and recalculated values were significant (>20 µM) in parts of the same region as well. Thus, the reduction of salinity-related variability was not generally possible and for some areas the assessments would remain weak with extended overlapping between boundary settings and monitoring data. Using a higher data density, salinity-related variability could be further reduced.

Conclusions and perspectives

Considering causal relationships between different eutrophication parameters, contradictions between reference values can be avoided and decreasing effects (oxygen depletions and Secchi depth limitations) can be coupled to increasing effects (nutrient concentrations) allowing similar deductions of background conditions and related assessment boundaries.

The mainly linear mixing of terrestrial-origin nutrient discharges in transitional and coastal waters with strong salinity gradients allows the calculation of salinity-related natural background concentrations

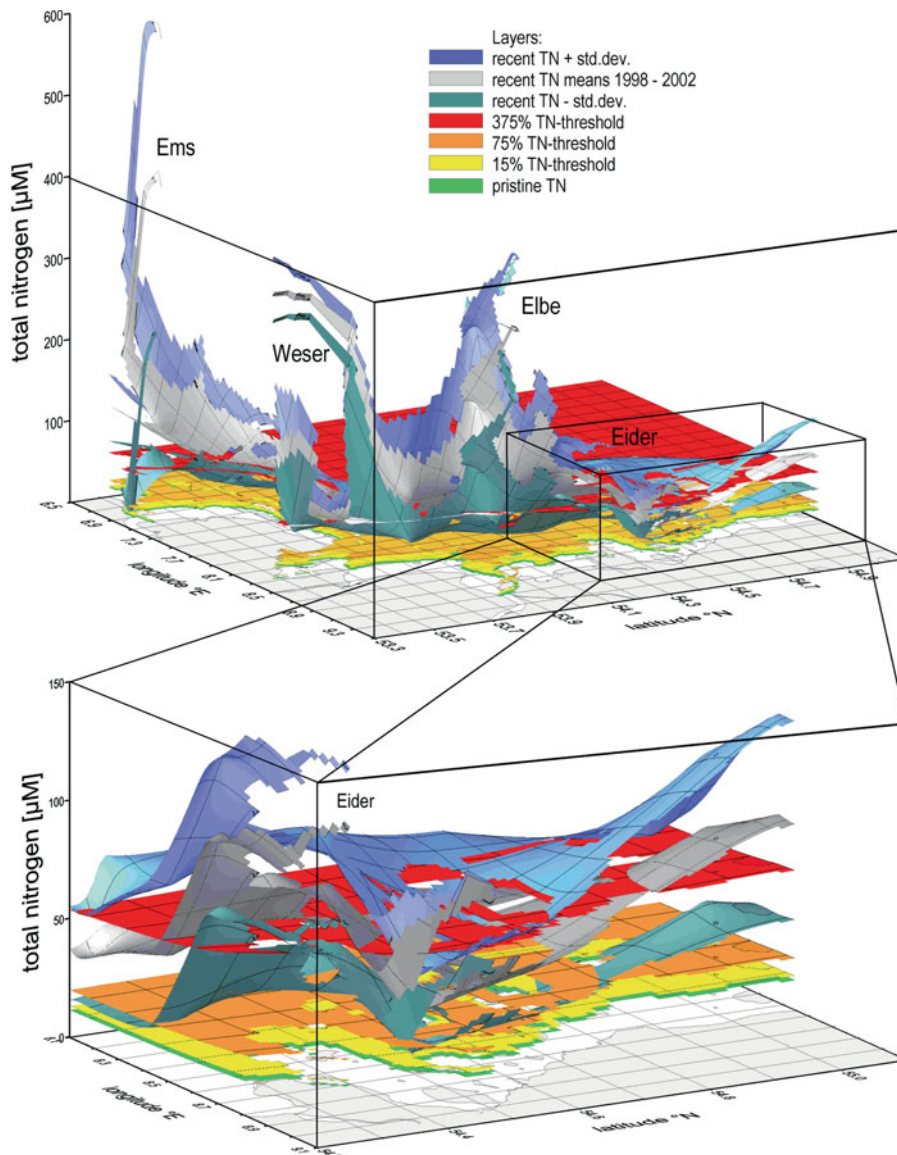


Fig. 5 3-D plot of recent mean TN concentrations and their standard deviations together with thresholds of the 15% approach in the German Bight. View to the northwest towards the sea. The enlarged section represents the North Frisian Coast

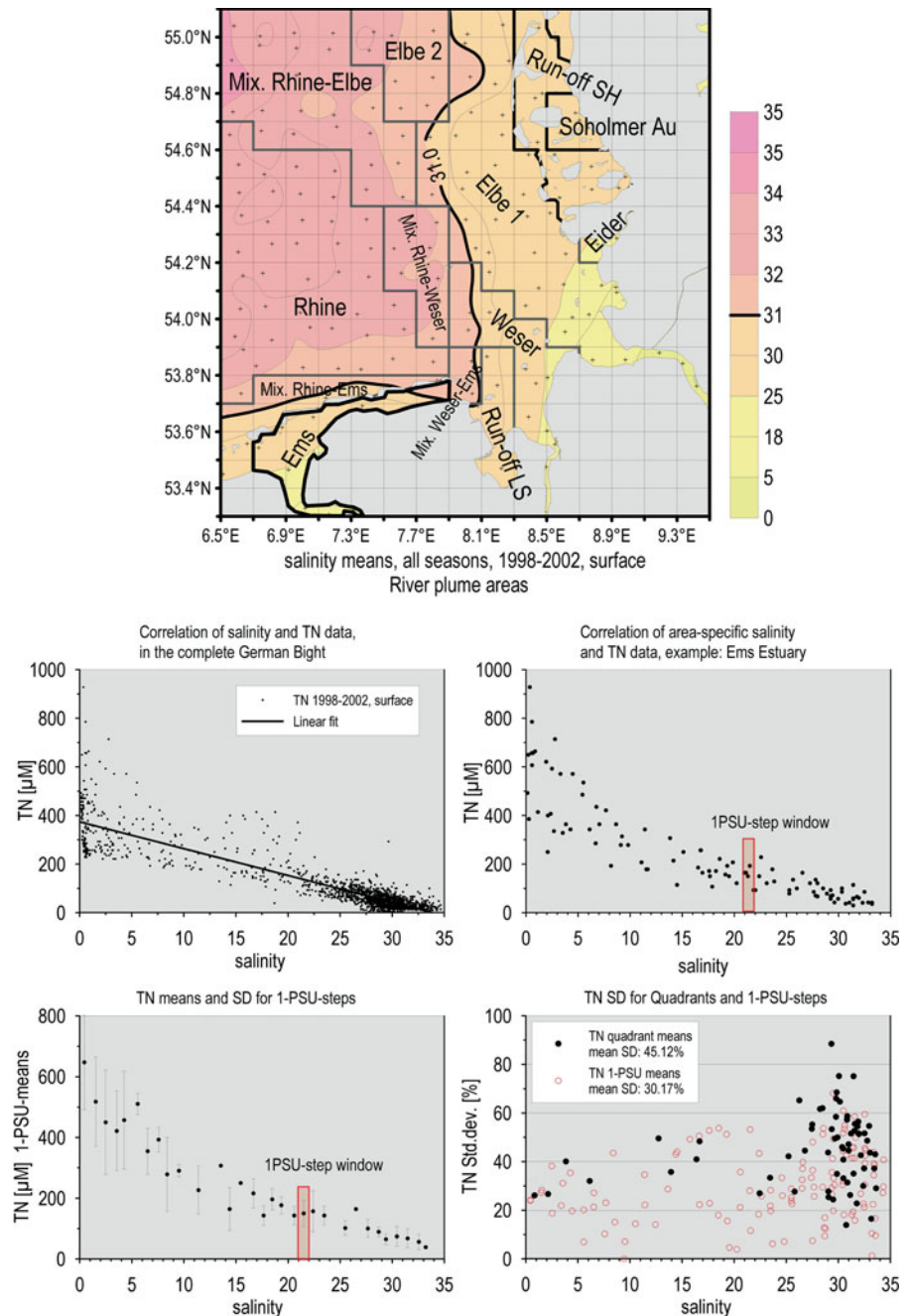
and boundary settings. This is also possible for directly coupled effects. However, reflecting the unlimited dilution potential of open waters, fixed offshore values should be defined, for example by approaching offshore values using hyperbolic fittings. In combination with linear fittings, based on the same data, for inshore areas, still consistent assessments can be achieved.

In order to keep the boundary setting as transparent as possible, a percentage relation to natural background concentrations is recommended.

Different boundaries, as required by the WFD, may be calculated using constant factors (e.g. 3 and 5 as here proposed) of percentage deviations. Ecological Quality Ratios can be calculated by dividing 100% of the natural background by $100 + x\%$ above the reference value of the boundary setting.

Boundary settings may be related to natural (e.g. hydrodynamic) variability, considering undesirable effects as well. Considering variability, an excess of 50% over natural background values has been introduced by OSPAR as a key boundary. This relation,

Fig. 6 Map of the German Bight divided into river plume areas (*top*), and mixing diagrams of recent (1998–2002) TN (μM) concentrations for the whole German Bight (*middle plot, left*) and the Ems estuary (*middle plot, right*). Deduced 1-PSU means and their standard deviations for the Ems (*bottom plot, left*) were compared with standard deviations from the squares in Fig. 1 for the Ems estuary (*bottom plot, right*)



however, is too high for some eutrophication effects, and 15% is proposed here as the key boundary especially for offshore waters, based on oxygen depletion, reduction of Secchi depth and extension of transboundary fluxes. At present, the 50% approach appears to be sufficient in near-coastal waters where current nutrient concentrations still exceed natural background concentrations in many areas.

Overlapping of boundary settings with regional gradients including their variability cannot be avoided, especially not for the application of the four boundary settings of WFD. In order to achieve the most significant assessments, the focus should be placed on the key boundaries (good/moderate for WFD or elevated level for OSPAR). Variability can be reduced to some degree by relating the monitoring data to mean local/

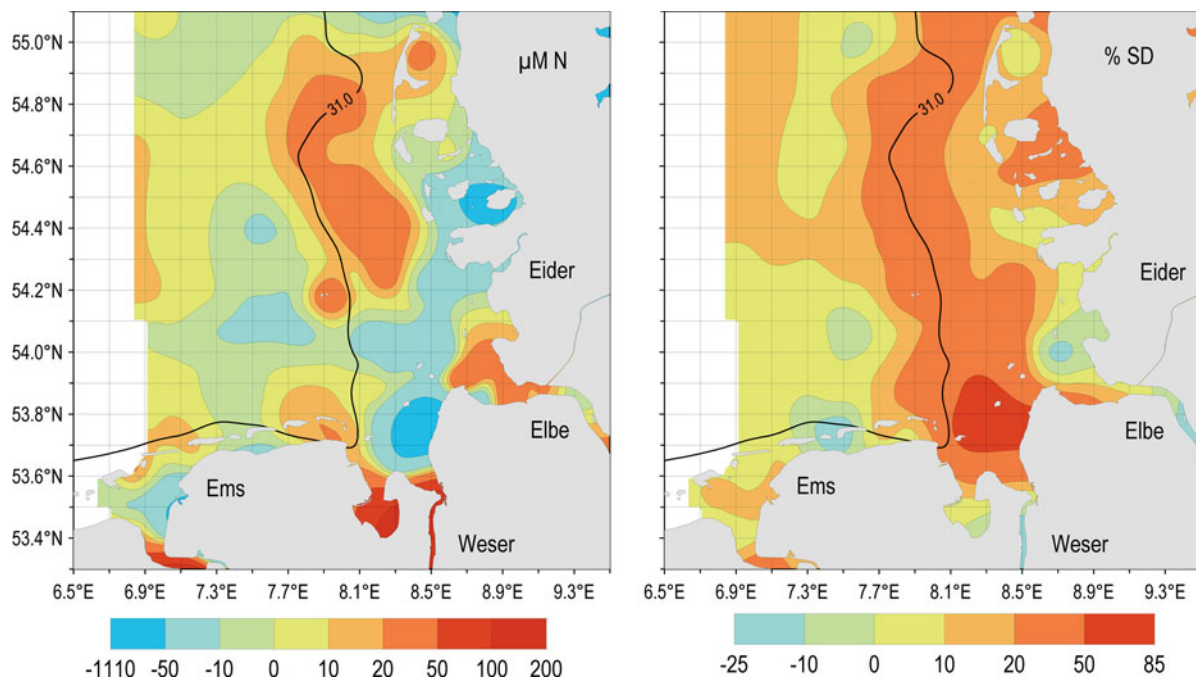


Fig. 7 Differences between TN quadrant means (Fig. 1) and salinity-related (1-PSU step) means (μM) (*left*) and differences in the standard deviations (%) (*right*) in the German Bight in surface waters for all the seasons (1998–2002)

regional salinities. However, ultimately recent means will be assessed in relation to mean boundary settings, and overlapping and variability (specific for areas, time periods and parameters) that affect the precision of classifications should be reported as part of the assessments.

Acknowledgements We thank the several data originators mostly anonymous, and the data centres for the compilation of the nutrient data (ARGE Elbe = Arbeitsgemeinschaft Elbe, Hamburg; MUDAB = Meeresumwelt Datenbank, Hamburg; Wadden Sea Secretariat, Wilhelmshaven). We are grateful to Monika Schütt & Thomas Raabe, Hamburg, for their essential assistance, and the two anonymous experts who reviewed and improved this manuscript.

References

- Andersen, J. H., D. J. Conley & S. Hedal, 2004. Palaeoecology, reference conditions and classification of ecological status: the EU Water Framework Directive in practice. *Marine Pollution Bulletin* 49: 283–290.
- Astall, C. M. & M. B. Jones, 1991. Respiration and biometry in the sea cucumber *Holothuria forskali*. *Journal of Marine Biological Association of the United Kingdom* 71: 73–81.
- Baden, S. P., L. Pihl & R. Rosenberg, 1990. Effects of oxygen depletion on the ecology, blood physiology and fishery of the Norway lobster *Nephrops norvegicus*. *Marine Ecology Progress Series* 67: 141–155.
- Bauerfeind, E., W. Hickel, U. Niermann & H. von Westernhagen, 1986. Sauerstoff-Defizit in tiefen Rinnen der Deutschen Bucht: Ursachen und biologische Auswirkungen. *Biologische Anstalt Helgoland, Hamburg, Jahresbericht* 1986: 72–79.
- Behar, S., 1997. Testing the Waters: Chemical and Physical Vital Signs of a River. River Watch Network, Montpolier.
- Behrendt, H., M. Bach, R. Kunkel, D. Opitz, W.-G. Pagenkopf, G. Scholz & F. Wendland, 2003. Internationale Harmonisierung der Quantifizierung von Nährstoffeinträgen aus diffusen und punktuellen Quellen in die Oberflächengewässer Deutschlands. UBA-FB 000446, Texte 82/03.
- Benson, B. B. & D. Krause Jr., 1984. The concentration and isotopic fractionation of oxygen dissolved in freshwater and seawater in equilibrium with the atmosphere. *Limnology and Oceanography* 29: 620–632.
- Breitbart, D., 2002. Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries. *Estuaries* 2: 767–781.
- Brockmann, U. & K. Eberlein, 1986. River input of nutrients into the German Bight. In Skreslet, S. (ed.), *The Role of Freshwater Outflow in Coastal Marine Ecosystems*. NATO ASI Series G7. Springer, Berlin: 231–240.
- Chabot, D. & J. Dutil, 1999. Reduced growth of Atlantic cod in non-lethal hypoxic conditions. *Journal of Fish Biology* 55: 472–491.
- Davis, J. C., 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a

- review. Journal of Fish Research Board of Canada 32: 2295–2332.
- Dethlefsen, V. & H. von Westernhagen, 1983. Oxygen deficiency and effects on bottom fauna in the eastern German Bight 1982. *Meeresforschung* 30: 42–53.
- Diaz, R. J. & R. Rosenberg, 1995. Oceanography. *Marine Biology Annual Review* 33: 245–303.
- European Communities, 2000. Water Framework Directive. Directive 2000/60/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.
- European Communities, 2002. Guidance on typology, reference conditions and classification systems for transitional and coastal waters. Water Framework Directive, Common Implementation Strategy, Working Group 2.4, (COAST).
- Fishdoc, 2005. <http://www.fishdoc.co.uk/water/temperature.htm>.
- Gamenick, I., A. Jahn, K. Vopel & O. Giere, 1996. Hypoxia and sulphide as structuring factors in a macrozoobenthic community on the Baltic Sea shore: colonisation studies and tolerance experiments. *Marine Ecology Progress Series* 144: 73–85.
- Gray, J. S., R. S. Wu & Y. Y. Or, 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series* 238: 249–279.
- Hagerman, L. & B. Vismann, 1995. Anaerobic metabolism in the shrimp *Crangon crangon* exposed to hypoxia, anoxia and hydrogen sulfide. *Marine Biology* 123: 235–240.
- Johansson, B., 1997. Behavioural response to gradually declining oxygen concentration by Baltic Sea macrobenthic crustaceans. *Marine Biology* 129: 71–78.
- Jørgensen, B. B., 1980. Seasonal oxygen depletion in the bottom waters of a Danish fjord and its effect on the benthic community. *OIKOS* 34: 68–76.
- Missouri, 2005. <http://www.cares.missouri.edu/dardenne/StrTmUpd.htm>.
- Mittelstaedt, E., W. Lange, C. Brockmann & K. C. Soetje, 1983. Die Strömungen in der Deutschen Bucht. BSH Map, Nr. 2347.
- Nielsen, S. L., K. Sand-Jensen, J. Borum & O. Geertz-Hansen, 2002a. Depth colonisation of eelgrass (*Zostera marina*) as determined by water transparency in Danish coastal waters. *Estuaries* 25: 1025–1032.
- Nielsen, S. L., K. Sand-Jensen, J. Borum & O. Geertz-Hansen, 2002b. Phytoplankton, nutrients, and transparency in Danish coastal waters. *Estuaries* 25: 930–937.
- Niermann, U., 1990. Oxygen deficiency in the south eastern North Sea in summer 1989. ICES, Copenhagen, C.M.1990.
- Niermann, U., E. Bauerfeind, W. Hickel & H. von Westernhagen, 1990. The recovery of benthos following the impact of low oxygen concentrations in the German Bight. *Netherlands Journal of Sea Research* 25: 215–226.
- OSPAR, 2000. OSPAR Quality Status Report 2000: Region II—Greater North Sea. OSPAR Commission.
- OSPAR, 2004a. Similarities and Synergies between the OSPAR Comprehensive Procedure, OSPAR Ecological Quality Objectives related to Eutrophication (EcoQOS-Eutro) and the EC Water Framework Directive. Submission from the OSPAR Eutrophication Committee to the Water Framework Directive Intercalibration Workshop, 11 February 2004. OSPAR Commission.
- OSPAR, 2004b. Testing for exceedence of target/threshold values. Joint meeting of the Eutrophication Task Group and the Working Group on Monitoring. Working Document 3, OSPAR Commission.
- OSPAR, 2005. Common Procedure for the Identification of the Eutrophication Status of the OSPAR Maritime Area. Reference number: 2005-3.4 OSPAR Commission.
- Pätsch, J. & H.-J. Lenhart, 2004. Daily loads of nutrients, total alkalinity, dissolved inorganic carbon and dissolved organic carbon of the European continental rivers for the years 1977–2002. *Berichte aus dem Zentrum für Meeres- und Klimaforschung, Universität Hamburg*. Nr. 48.
- Peckolt, P. & J. S. Rivers, 1995. Physiological responses of the opportunistic macroalgae *Cladophora vagabunda* (L.) van den Hoek and *Gracilaria tikvahiae* (McLachlan) to environmental disturbances associated with eutrophication. *Journal of Experimental Marine Biology and Ecology* 190: 1–16.
- Petersen, J. K. & G. I. Petersen, 1988. Sandkutlingens respiration og vakst under hypoxi. Masters thesis, University of Copenhagen.
- Pihl, L., S. P. Baden & R. J. Diaz, 1991. Effects of periodic hypoxia on distribution of demersal fish and crustaceans. *Marine Biology* 108: 349–360.
- Pihl, L., S. P. Baden, R. J. Diaz & L. C. Schaffner, 1992. Hypoxia-induced structural changes in the diet of bottom-feeding fish and crustacea. *Marine Biology* 112: 349–361.
- Rachor, E., 1983. Extreme Sauerstoffverhältnisse in der Deutschen Bucht. *Arbeiten des Deutschen Fischerei-Verbandes (Hamburg)*, Heft 37 (Beiträge zur Eutrophie der Deutschen Bucht): 15–27.
- Rachor, E., 1985. Eutrophierung in der Nordsee—Bedrohung durch Sauerstoffmangel. *Abhandlungen des Naturwissenschaftlichen Vereins zu Bremen* 40: 283–292.
- Rachor, E. & H. Albrecht, 1983. Sauerstoffmangel im Bodenwasser der Deutschen Bucht. *Veröffentlichungen des Instituts für Meeresforschung in Bremerhaven* 19: 209–227.
- Redfield, A. C., B. H. Ketchum & F. A. Richards, 1963. The influence of organisms on the composition of sea water. In Hill, M. N. (ed.), *The Sea*. Wiley Interscience, New York: 26–77.
- Roman, M. R., A. L. Gauzens, K. Rhinehart & J. R. White, 1993. Effects of low oxygen waters on Chesapeake Bay zooplankton. *Limnology and Oceanography* 38: 1603–1614.
- Rosas, C., E. Martinez, G. Gaxiola, R. Brito, A. Sanchez & L. A. Soto, 1999. The effect of dissolved oxygen and salinity on oxygen consumption, ammonia excretion and osmotic pressure of *Penaeus setiferus* (Linnaeus) juveniles. *Journal of Experimental Marine Biology and Ecology* 234: 41–57.
- Rosenberg, R., B. Hellman & B. Johansson, 1991. Hypoxic tolerance of marine benthic fauna. *Marine Ecology Progress Series* 79: 127–131.
- Sandberg, E. & E. Bonsdorff, 1996. Effects of predation and oxygen deficiency on different age classes of the amphipod *Monoporeia affinis*. *Journal of Sea Research* 35: 345–351.
- Seager, J., I. Milne, M. Mallett & I. Sims, 2000. Effects of short-term oxygen depletion on fish. *Environmental Toxicology and Chemistry* 19: 2937–2942.

- Secor, D. H. & T. E. Gunderson, 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon, *Acipenser oxyrinchus*. *Fish Bulletin* 96: 603–613.
- Sedin, R., 2002. Bottom-level oxygen in coastal and marine waters. Swedish EPA, <http://www.internat.naturvardsverket.se/documents/legal/assess/assedoc/coastdoc/bottoxy.htm>.
- Smith, V. H., 2006. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51: 377–384.
- Smith, S. V., P. E. Damm, M. D. Skogen, R. A. Flather & J. Pätsch, 1996. An investigation into the variability of circulation and transport on the north-west European shelf using three hydrodynamic models. *Deutsche Hydrographische Zeitschrift* 48: 325–347.
- Spicer, J. I., 1995. Oxygen and acid–base status of the sea urchin *Psammechinus miliaris* during environmental hypoxia. *Marine Biology* 124: 71–76.
- Stickle, W. B., M. A. Kapper, L.-L. Liu, E. Gnaiger & S. Y. Wang, 1989. Metabolic adaptations of several species of crustaceans and molluscs to Hypoxia: tolerance and microrimetric studies. *Biological Bulletin* 177: 303–312.
- Tallqvist, M., K. E. Sandberg & E. Bonsdorff, 1999. Juvenile flounder, *Platichthys flesus* (L.), under hypoxia: effects on tolerance, ventilation rate and predation efficiency. *Journal of Experimental Marine Biology and Ecology* 242: 75–93.
- Tett, P., L. Gilpin, H. Svendsen, C. P. Erlandsson, U. Larsson, S. Kratzer, E. Fouilland, C. Janzen, J.-Y. Lee, C. Grenz, A. Newton, J. G. Ferreira, T. Fernandes & S. Scory, 2003. Eutrophication and some European waters of restricted exchange. *Continental Shelf Research* 23: 1635–1671.
- Theede, H., A. Ponat, K. Hiroki & C. Schlieper, 1969. Studies on the resistance of marine bottom invertebrates to oxygen deficiency and hydrogen sulphide. *Marine Biology* 2: 325–337.
- Tomczak, G. & E. Goedecke, 1964. Die thermische Schichtung der Nordsee auf Grund des mittleren Jahresganges der Temperatur in ½°- und 1°-Feldern. *Deutsche Hydrographische Zeitschrift, Ergänzungsheft B (4°)*, no. 8.
- Van Raaphorst, W., V.N. de Jonge, D. Dijkhuizen & B. Frederiks, 2000. Natural background concentrations of phosphorus and nitrogen in the Dutch Wadden Sea. RIKZ/2000.013, The Hague.
- Wajsbrodt, N., A. Gasith, M. D. Krom & D. M. Popper, 1991. Acute toxicity of ammonia to juvenile gilthead seabream *Sparus aurata* under reduced oxygen levels. *Aquaculture* 92: 277–288.
- Wang, W. X. & J. Widdows, 1991. Physiological responses of mussel larvae *Mytilus edulis* to environmental hypoxia and anoxia. *Marine Ecology Progress Series* 70: 223–236.
- Wu, R. S. S., 2002. Hypoxia: from molecular responses to ecosystem responses. *Marine Pollution Bulletin* 45: 35–45.
- Zevenboom, W., 1994. Assessment of eutrophication and its effects in marine waters. *Deutsche Hydrographische Zeitschrift Supplement 1*: 141–170.

Modelling the distribution and growth of ‘problem’ green seaweed in the Medway estuary, UK

J. N. Aldridge · M. Trimmer

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 107–122.
DOI: 10.1007/s10750-009-9760-6 © UK Crown Copyright 2009

Abstract The results of a modelling study to investigate the mechanisms controlling macroalgal growth within the Medway estuary, UK, are presented. Intertidal zone bathymetry, tidal dynamics and turbidity control the time available for nutrient uptake and photosynthesis, and were used as a basis for predicting areas where macroalgae will grow. Tidal bed stress was also considered as a controlling factor for the presence of the less robust green macroalgae species. Two approaches to predicting macroalgal distributions were applied: (1) a simple ‘suitability index’ method based on tidal flooding and drying, taking account of the conflict between time available for nutrient uptake and for photosynthesis; and (2) a biological macroalgal growth model that includes a

detailed treatment of nutrient uptake and plant growth. The former approach assigns a value between zero and one for the suitability of a location for macroalgal occurrence, while the latter predicts the full macroalgal growth dynamics over an annual cycle. Tidal bed stress was included in both approaches as an independent modifier of macroalgal occurrence/growth. Results were compared with aerial survey maps of observed vegetation cover and time series of measured biomass density. Both approaches gave good predictions of non-species-specific vegetation cover in the intertidal zone of the Medway. Tidal bed stress was found to be a strong predictor of the specific occurrence of *Enteromorpha* spp. and *Ulva* spp., with these species favouring areas of low tidal energy. It was concluded that light and a lack of suitable regions with low tidal bed stress, rather than nutrients, were the main factors limiting excessive growth of *Enteromorpha* spp. and *Ulva* spp. in the estuary. Although this study was focussed on the Medway, the results are likely to be applicable to a broad range of relatively turbid, meso- and macro-tidal estuaries.

Electronic supplementary material The online version of this article (doi:10.1007/s10750-009-9760-6) contains supplementary material, which is available to authorized users.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

J. N. Aldridge (✉)
Cefas, Pakefield Rd., Lowestoft, Suffolk NR33 0HT, UK
e-mail: john.aldridge@cefas.co.uk

M. Trimmer
School of Biological Sciences, Queen Mary University
of London, London E1 4NS, UK

Keywords Macroalgae · Models · Medway estuary · Nutrients · Light · Bed stress

Introduction

Macroalgae provide one of the more visible signs of eutrophication in the coastal zone, often occurring as

a proliferation of algal weeds in predominantly sheltered sites. Prolific growth of particular macroalgae species may affect the quality of the estuarine environment by covering large areas with opportunistic algae, often displacing other species and decreasing biodiversity (Tubbs & Tubbs, 1983; Lavery et al., 1991; den Hartog, 1994). The organisms responsible are mainly the smaller fast-growing green algae, such as *Enteromorpha* spp., *Ulva* spp., *Chaetomorpha* and *Cladophora*, although some brown algae such as *Ectocarpus* occupy a similar ecological niche. Within the UK, these species tend to proliferate in sheltered muddy and sandy intertidal zones. High uptake rates of nutrients, combined with the ability to store nutrients (Bjornsater & Wheeler, 1990; Aisha & Shabana, 1995), allow estuarine macroalgae to proliferate in those systems that have high but variable nutrient availability, for example, owing to the dynamics of tidal wetting and drying. Where coastal waters are hypereutrophic, nutrient demands can be satisfied quickly by tidal exchange (Sfriso, 1995; Cefas, 2003) and the high affinity of *Enteromorpha* and *Ulva* for nitrate (NO_3^-) and ammonium (NH_4^+) puts them at an advantage compared with more slowly growing species (Pedersen & Borum, 1997). Also, nutrient recycling by remineralization of detrital material, coming either from coastal primary production or from the macroalgae itself (Trimmer et al., 2000; Eilola & Stigebrandt, 2001), can support macroalgal biomass additional to that associated with direct inputs.

However, not all sites with elevated nutrient concentrations have high populations of macroalgae. Nitrate and ammonium are abundant in estuaries across the UK (Nedwell et al., 2002), but this is not matched by the distribution of nuisance blooms of the ephemeral macroalgae *Enteromorpha* spp. and *Ulva* spp. Other factors such as the extent and nature of the intertidal zone and the turbidity of the water are likely to play a role. For many species, attachment sites are required, such as shell debris, saltmarsh or scattered rocks, to provide at least temporary anchorage. In addition, high sedimentation rates in the short term may smother any potential growth, while tide and wave exposure and associated sediment disturbance may also limit the potential for accumulation of biomass. For example, observational evidence indicates that the green seaweeds *Enteromorpha* spp. and *Ulva* spp. prefer sheltered locations in an estuary

(Lowthion et al., 1985; Sfriso et al., 1989; Martins & Pardal, 2001).

In this modelling study, the authors test the hypothesis that factors other than nutrient concentrations play the key role in the occurrence of problem macroalgae in some estuaries. The Medway estuary is an example of a hypereutrophic estuary with an extensive intertidal zone, with large areas covered by problem green macroalgae, but which nevertheless has a significantly smaller percentage coverage than other UK estuaries with lower nutrient concentrations (e.g. Langstone Harbour; Cefas, 2003). The Medway estuary is thus a good location to begin to investigate the above hypothesis. Although this study is focussed on the Medway estuary in particular, the results are likely to be applicable to a broad range of relatively turbid, meso- and macro-tidal estuaries. In this article, the aim is to give a description of the modelling approaches used and the comparison with observed macroalgal cover. At this stage, comparison between observations and the model results is presented in a semi-quantitative and descriptive manner rather than quantitative statistical fits.

Methods

Study region

The Medway estuary can be regarded as part of the greater Thames estuary in southern England (Fig. 1). The region surrounding the estuary has a mixture of industrial, urban and rural land uses. A number of sewage treatment works (STW) discharge into the estuary directly or into the Medway River that enters at the western end of the estuary. At the eastern end, tidal exchange occurs with the hypereutrophic water from the Thames. Because of a history of industrial pollution, recreational use of the Medway estuary is limited. Nevertheless, with the cessation of much industrial discharge and control of organic inputs, water quality has improved from the late 1990s (Pratt, 2001). The estuary provides an important habitat for waterfowl and is classified as a Special Protection Area under the EC Birds Directive. The majority of the estuary is intertidal, with only the main channels inundated at low water. Spring tidal range at the mouth of the estuary is 5.2 m and tidal velocities in the channels at spring tides can approach 1 m s^{-1} .

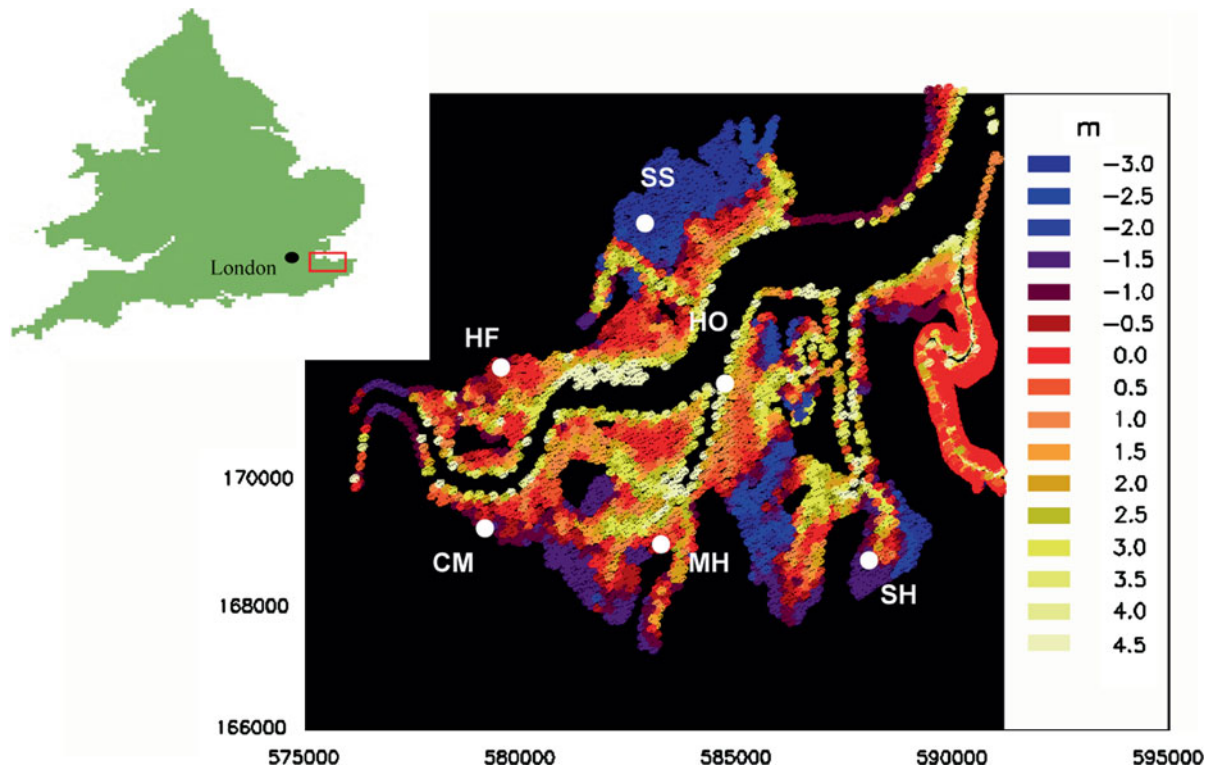


Fig. 1 Depth distribution (m) of the estuary intertidal zone of the Medway. Datum is ordnance datum Newlyn (approximately mean sea level). Land and depths below 4.5 m are shown *black*. Locations referred to in the text are also marked:

MH Motney Hill STW, *SH* Strawberry Hill, *CM* Copperhouse Marsh, *HF* Hoo Flats, *SS* Stoke Saltings, *HO* Ham Ooze. Vertical and horizontal scales are northings and eastings in the British National Grid system

The estuary is turbid, with measurements of suspended load approaching 100 mg l^{-1} . The estuary is hypernutrified, with both summer and winter nutrient concentrations measured in the main channels remaining above the recognized half-saturation constants for green macroalgae ($K_{\text{NO}_3} \sim 0.3 \text{ mg l}^{-1}$ was used in the model; Fig. 2A). There is evidence that over the summer and autumn, biological nitrogen (N) demand reduces the concentration of nitrate in the water in some parts of the estuary, for example, in the mouth relative to the head (Fig. 2B), although even here nitrate remained above typical values of K_{NO_3} . The relatively small decrease in dissolved available inorganic nitrogen (DAIN) concentrations seen in summer is strong evidence that biological processes do not dominate the nutrient dynamics on an estuary-wide scale. A summary of data collected by the UK Environment Agency and an assessment of the water quality status of the Medway (Pratt, 2001) provided much useful background information for this study.

Approach

Two approaches were applied to predicting the distribution of macroalgae: (1) a simple 'suitability index' (SI) method, based on the proportion of time any given location is under water, the associated light climate and the time available for nutrient uptake from the water, and (2) a full macroalgal growth model that, in addition, takes account of macroalgal turnover and nutrient concentrations to predict biomass. Both approaches included an empirical correction for the effect of tidal bed stress. Physically, it is envisaged that *Enteromorpha* spp. and *Ulva* spp. require a relatively stable substrate and at bed stresses above a threshold, sediment movement inhibits their persistence and development. Predictions of tidal height, tidal currents and nutrient concentrations were required to implement both approaches. These were obtained from a separate hydrodynamic and nutrient modelling study of the estuary (HR Wallingford, 2004). For both approaches, success was judged

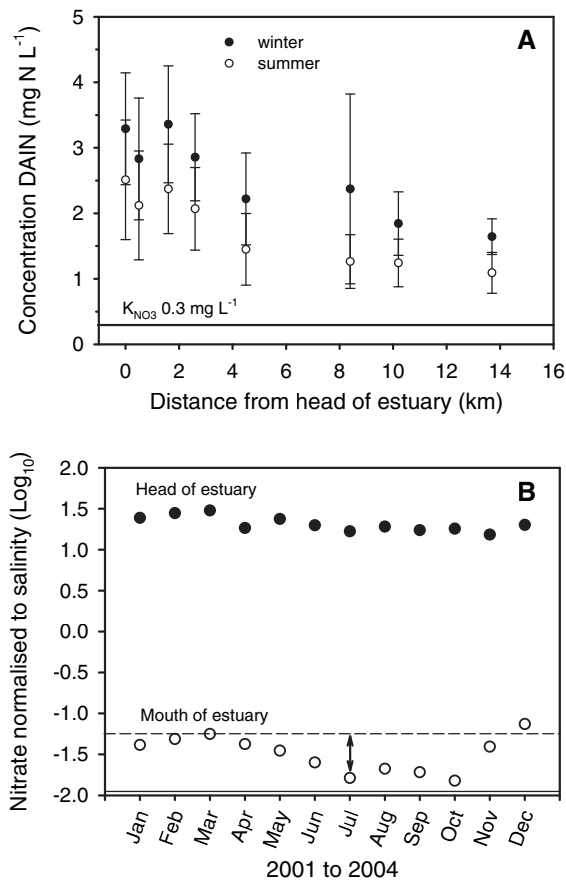


Fig. 2 **A** Mean (monthly spot measurements 2001–2004) concentrations for dissolved available inorganic nitrogen (DAIN) as nitrate plus ammonium along the Medway estuary in the winter and summer (± 1 SD). Note that the half-saturation constant used in the model for nitrate (K_{NO_3}) of 0.3 mg l^{-1} is indicated. **B** Seasonal variation in mean monthly spot sample concentrations for nitrate normalized for changes in salinity from the head and mouth of the estuary. The *dashed line* represents the mean winter concentration for NO_3^- and the *solid line* the half-saturation constant (both normalized). *Arrow* indicates some potential biological removal of nitrate at the mouth (Environment Agency, UK, unpublished data)

based on the reproduction of observed spatial distributions of vegetation cover. For the full macroalgal growth model, agreement with bi-monthly observations of macroalgal biomass at two locations was also taken into account.

Hydrodynamics and nutrient transport

Bathymetric information and tidal dynamics were obtained from a supporting, but separate study (HR Wallingford, 2004). For completeness, a brief

description is given here. A high-resolution model of the complete Thames estuary, including the Medway estuary, was implemented using the finite element TELEMAC-2D hydrodynamic model (Jones, 2005) and transport model WQFLOW-2D. Typical mesh resolution within the Medway estuary was 100 m. Simulations included the principal lunar and solar diurnal tidal constituents yielding predictions of the time-varying water depth above the seabed, with appropriate wetting and drying of mesh points, over a spring-neap cycle. Validation of the hydrodynamic and nutrient transport models is described in HR Wallingford (2004) and consisted of comparison against current meter data and observed nitrate, ammonium and oxygen concentrations measured during a summer and winter field study. Agreement with currents was generally very good. Nutrient concentrations were also in general agreement with observations, apart from high up near the head of the estuary where model concentrations were higher than measured.

Time series of tidal heights together with the peak tidal current were made available at each model mesh point in the Medway for use by both macroalgal prediction methods. Nutrient concentrations were calculated on the same mesh using the WQFLOW-2D transport model, with inputs of NO_3^- and NH_4^+ from the STW and diffuse sources for the entire Thames system, including offshore boundary concentrations. Tidally averaged concentrations of nitrate and ammonium were calculated for representative summer and winter conditions. In order to generate a complete annual cycle from the modelled winter and summer nutrient distributions, monthly values of observed concentrations provided by the UK Environment Agency were used. These covered several stations in the Medway and spanned the period from February 2001 to December 2003. The data were spatially and temporally averaged to produce a ‘mean’ annual cycle. The cycle was then normalized so that the annual minimum and maximum values at any location corresponded to the modelled summer and winter values, respectively, with the time series then smoothly interpolating between them.

Suitability index method

In order to assess the suitability of a given location for plant growth, the position within the intertidal zone with respect to the tidal rise and fall will play a

key role (Dring, 1986). In turbid estuaries, light limitation is likely to inhibit growth at positions low down in the intertidal zone. However, high up in the intertidal zone, growth will be limited because of infrequent tidal inundation restricting nutrient uptake, together with enhanced mortality from the impact of desiccation and thermal exposure. This leads to the idea of an intermediate region where both light and nutrients are sufficient for plant growth to occur.

To define this region, detailed bathymetric information, suspended sediment concentrations (SSC) and an assumed SSC light attenuation relationship were combined with hydrodynamic model predictions of tidal elevation. For each location, changes in water depth due to the rise and fall of the tide were obtained from the hydrodynamic simulations. If $P_{>d}$ represents the proportion of time the water depth is greater than d and if the depth of the photic zone is D_* , then the proportion of time that photosynthesis is possible is $(1 - P_{>D_*})$. The proportion of time the location is wet and thus able to uptake nutrients is $P_{>0}$. These were then combined to give a 'score' or SI of the form:

$$G = P_{>0}(1 - P_{>D_*}). \quad (1)$$

Higher values of G were taken to indicate greater potential for macroalgal growth. The index is zero for locations always dry or always below D_* .

A common criterion for defining the critical depth D_* is where the incident light is attenuated to 1% of the surface value and this was used for this study. Note that the full water depth at any given time was used to compare with D_* , and therefore, it was the light climate at the seabed that was assumed to be relevant. As growth takes place, many macroalgal species are able to extend into the water column and enhance their exposure to light. However, assuming little over-wintering of adult plants, initial growth will be associated with juvenile plants confined to the seabed; therefore, to predict the presence or absence of growth, the full water depth was appropriate. Light attenuation was estimated as a function of SSC according to Sanders et al. (1997) modified with a linear extrapolation for low turbidity. Given limited knowledge of the time-varying distribution of SSC, a uniform concentration of 80 mg l^{-1} was used based on measurements by the UK Environment Agency.

An optional dependence of the index on the distribution of bed stress was included using the empirical factor:

$$\gamma = 1/(1 + \text{Exp}[(\tau_{\text{max}} - \tau_c)/\delta]) \quad (2)$$

that penalized locations where the maximum bed stress τ_{max} was greater than a critical value τ_c , while having no effect when $\tau_{\text{max}} < \tau_c$. The parameter δ controls how sharp the transition between no effect and growth inhibition is at τ_c . This concept is related to the classic distribution of macroalgae on sheltered versus non-sheltered shores (Ballentine, 1961) and the authors speculate that it is related to the stability of the bed. Peak bed stresses from the TELEMAC model were used to calculate the SI at each mesh point. The modified score is given by

$$G_\gamma = \gamma P_{>D_*}(1 - P_{>0}). \quad (3)$$

Although relatively straightforward to understand and apply if bathymetric and tidal height information is available, the SI does not take into account nutrient concentrations and does not yield a prediction of biomass. Both of these limitations are addressed by combining the above ideas with a model of macroalgal growth, which is described next.

Macroalgal growth model

The numerical model formulation is similar to a number of models that have appeared in the literature (e.g. Solidoro et al., 1997; Martins & Marques, 2002). The full formulation is given in the supplementary material—Appendix—and only a brief description is given here. The model is based on equations describing the uptake of nitrogen and photosynthesis with separate affinities for nitrate and ammonium. Species of *Ulva* are relatively impoverished with respect to phosphorus (Atkinson & Smith, 1983) and it was assumed that phosphorus does not play a role in limiting growth, although this cannot be ruled out. It is known that *Enteromorpha* spp. and *Ulva* spp. can store unassimilated nitrogen within their cells (e.g. Mohsen et al., 1974; Rosenberg & Ramus, 1982) and the model formulations presented in Solidoro et al. (1997), Martins & Marques (2002) and others, were followed where the nutrient uptake step and the photosynthesis/assimilation steps were separated and modelled as a two-stage process. Field measurements

by Trimmer et al. (2000) indicate nutrient uptake to be relatively insensitive to light conditions. This allows the modelled macroalgae to absorb nitrogen when in the water, even if photosynthesis is inhibited, and then to grow using stored nutrients and dissolved or atmospheric carbon dioxide (CO₂) when water depth decreases and light levels become sufficient. Although reduced oxygen saturation can be observed in the estuary (Pratt, 2001), it was assumed that the intensity and duration of occurrence was not sufficient to significantly increase macroalgal mortality.

Photosynthesis was modelled as a function of internal nitrogen concentration and light intensity. Light regulation of photosynthesis, including photo-inhibition at high light intensity, was modelled using the form adopted by Martins & Marques (2002). The annual variation in surface light intensity was derived from observed monthly daylight averages measured in London and then modified to account for the seasonal day length. Attenuation of the surface light intensity was taken to be a function of SSC and water depth. As for the SI approach, a uniform 80 mg l⁻¹ value for SSC was used, and the light intensity was calculated at the sediment surface. As discussed previously, this is appropriate for ascertaining whether growth at a given location will be initiated. A possible modification to the model could take into account the macroalgal height in the water column once the initial growth had been initiated. However, although this modification might affect the calculated biomass, it would not change the predicted presence or absence of growth at any given location. Temperature was also included as a factor limiting macroalgal growth. An idealized annual temperature curve based on typical summer and winter temperatures for the region was used. An optional dependence of macroalgal mortality on the value of the bed stress was included such that with increasing values above the threshold, macroalgal mortality was increased.

Independent macroalgal models were run corresponding to each point on the hydrodynamic model mesh using the tidal height and average nutrient concentrations derived from the hydrodynamic model calculations. A semi-coupled approach was devised that allowed the applied hydrodynamic transport model concentrations, which pertain to the situation with no macroalgal growth, to be corrected for a given macroalgal nutrient demand. A parameter (the 'refresh rate') controlled how quickly depleted

nutrients were replaced, mimicking the physical advection and diffusion processes within the estuary. This was implemented by re-supplying nutrients at a rate proportional to the difference between the reference concentration from the hydrodynamic model and the concentration corrected for macroalgal nutrient demand (Eqs. 15 and 16 in the supplementary material—Appendix). Thus, the refresh rate is the local equivalent of an estuary flushing rate and can, in principle, be calculated at each point within the estuary from the hydrodynamic transport model. In practice, it was used as a calibration parameter to obtain broadly correct overall biomass within the estuary. The main limitations of the approach are that interactions between macroalgae in neighbouring grid cells are neglected (for example, competition for the nutrient pool) and this restricts its applicability to estuaries such as the Medway where there is an excess of nutrients. Nevertheless, over intertidal areas with significant macroalgal biomass, it is likely that local nutrient concentrations may start to become limiting due to biological demand. In the macroalgal growth model simulations, this is implied by a dependence of the macroalgal biomass density on the refresh rate parameter.

Vegetation cover

Observed vegetation coverage for August 2002 and 2003 was obtained from a separate study that used aerial surveys with False Colour Infra Red (FCIR) and Compact Airborne Spectrographic Imager (CASI) (Babey & Soffer, 1992) equipment and ground-truthed by field observations (Brown, 2004). Observed distributions of vegetation cover were compared with predicted macroalgal distributions. In addition, estimates of macroalgal biomass (kg m⁻² wet weight) for the years 2001–2003 were available at selected locations based on bi-monthly surveys undertaken from March to November inclusive (Environment Agency, UK, unpublished data). When comparing with the macroalgal growth model, dry weight predictions were converted to wet weights assuming a wet-to-dry weight ratio of 7.0 (Pratt, Environment Agency, UK, pers. comm.).

Before assessing the success of the predictive models, it is important to consider the differences in the observed distributions of vegetation cover for the two available years. Remotely sensed distributions of

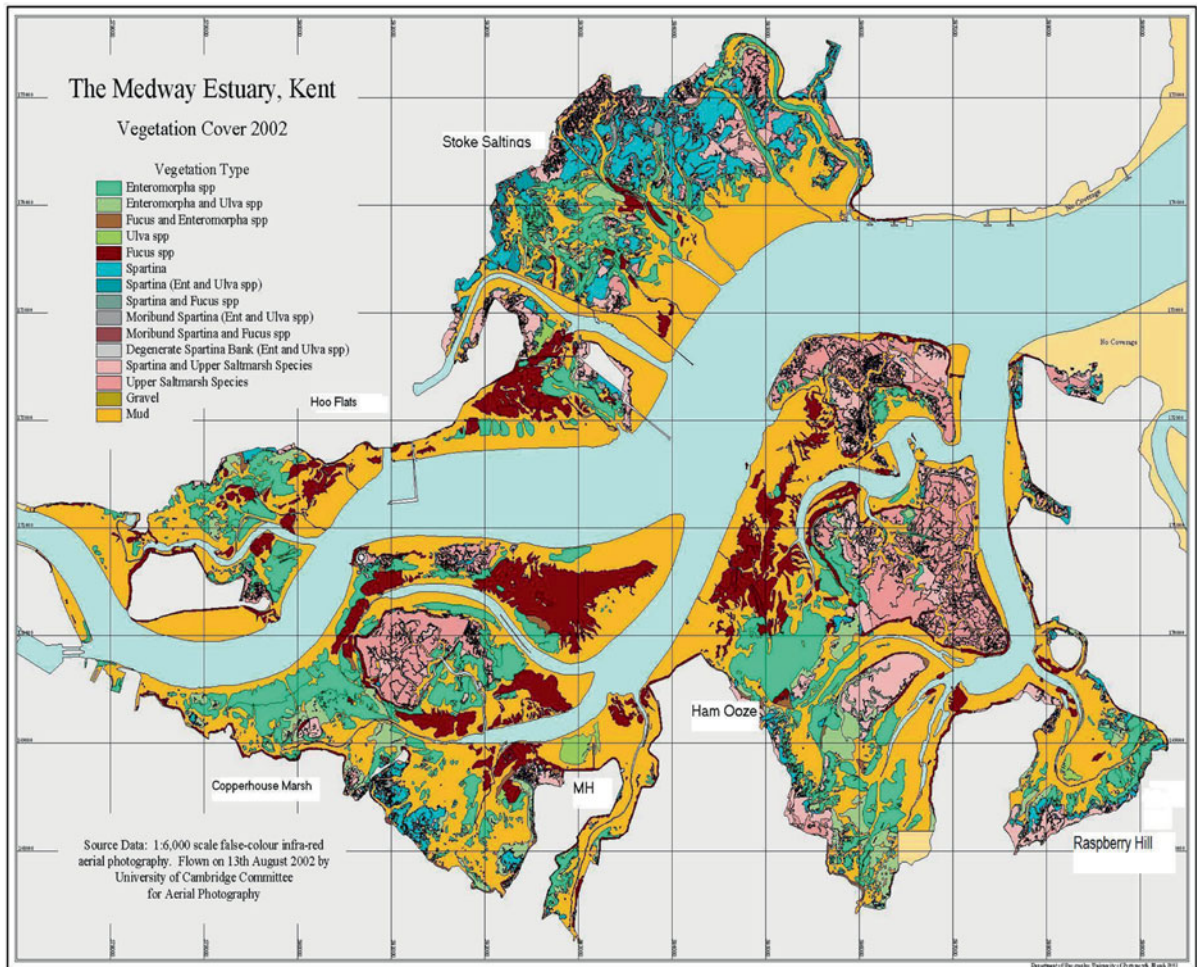


Fig. 3 Distributions of vegetation derived from remote sensing for the Medway, August 2002. Locations referred to in the text are also marked. *MH* Motney Hill STW (Environment Agency, UK, unpublished data. Reproduced with permission)

vegetation based on over-flights for August 2002 using FCIR (Fig. 3) and for August 2003 using CASI (Fig. 4) show similar, although not identical, distributions of vegetation. Differences may reflect both real inter-annual variability and differences in interpretation between the two remote sensing approaches. Crucially, the main areas affected by green seaweed are consistent between the two images, with significant coverage at Ham Ooze, Copperhouse Marsh, the Hoo flats and Stoke Saltings. However, it does appear that the area occupied by green seaweed in 2003 was more limited than that in 2002 and the same is true for the brown fucoids. Some areas marked as mud or covered with macroalgae in August 2002 appear to be predominantly covered by diatom mats in August 2003. In general,

there is a heterogeneous distribution of species in the intertidal zone, and the dominant areas of coverage clearly represent different groups of macroalgae with, for example, the large fucoid beds clearly separate from the *Enteromorpha* spp. beds.

In the following discussions, the ‘non-species-specific’ distribution will refer to the distribution of all biological types categorized in Figs. 3 and 4, apart from the diatoms.

Results

The predicted distribution of bed stress derived from hydrodynamic model simulations is shown in Fig. 5. Due to frictional damping of flow, maximum bed

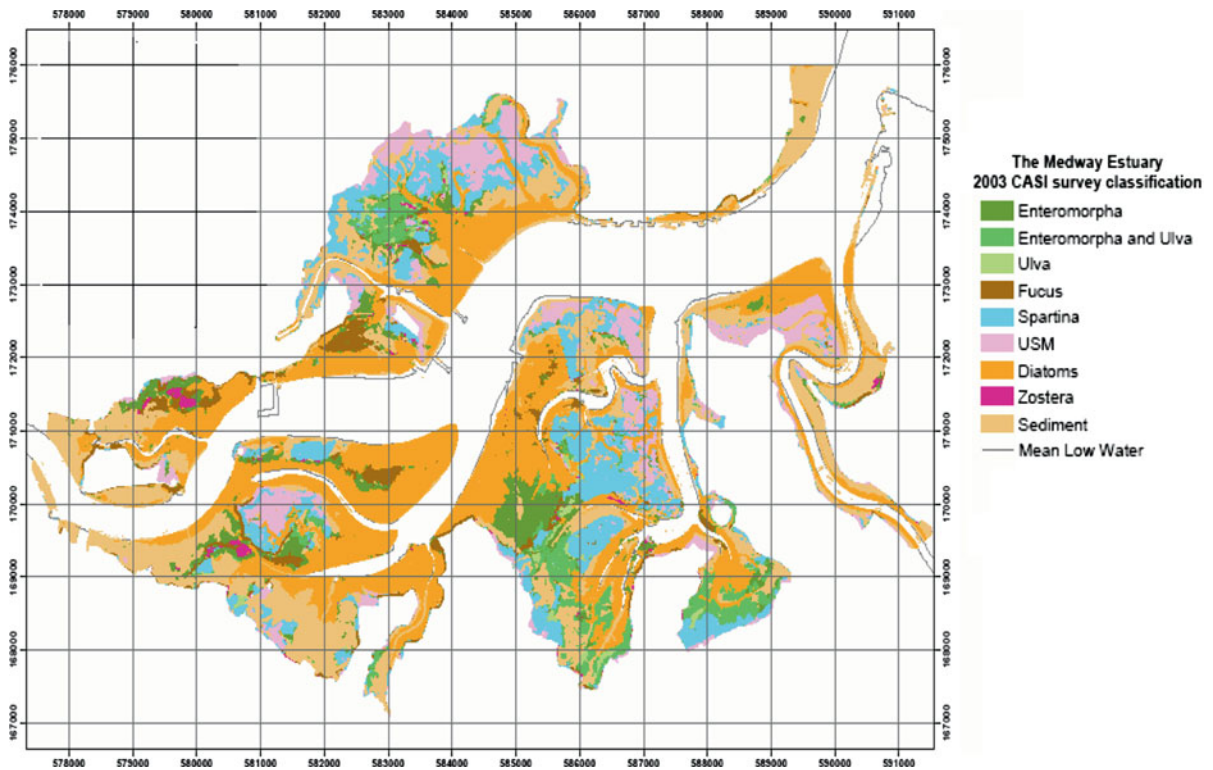


Fig. 4 Distributions of vegetation derived from remote sensing for the Medway, August 2003 (Environment Agency, UK, unpublished data. Reproduced with permission)

stresses are largely associated with areas of the intertidal zone closest to the main channels. However, the distribution is rather complex and distinct areas of high stress are separated from each other by bands of low stress (top left). Large areas of the intertidal zone are subject to minimal bed stress disturbance (peak values $<0.25 \text{ N m}^{-2}$). As will be shown later, areas of low bed stress are closely associated with the occurrence of green macroalgae *Enteromorpha* spp. and *Ulva* spp.

Application of the suitability index method

As previously described, the SI approach assigns a score between zero and one with locations having the largest value assumed to be most favourable for growth. The calculated distribution of this score 'G' is shown in Fig. 6, where it is assumed that macroalgae are absent when $G < 0.22$. If this is compared with the observed cover in August (Figs. 3, 4), good agreement is found for the non-species-specific distribution, although with a tendency for the

predicted regions of growth to be somewhat more extensive than that observed. Locations of significant vegetation cover such as at Copperhouse Marshes and the Hoo flats are well reproduced. Predictions match better the 2002 distributions than those observed in 2003. Discrepancies include the Stoke Saltings, where the predicted growth is much lower down in the intertidal region than that observed, and the region south of the Ham Ooze, where the observed macroalgal cover is absent in the SI results. Both of these derive ultimately from the underlying bathymetry, which shows these regions high up and therefore infrequently inundated. Clearly, a further refinement would be to investigate how accurate the bathymetry is as this has a major influence on the predicted distributions of growth.

If the results are weighted by the bed stress factor (Eq. 2, with $\tau_c = 0.5 \text{ N m}^{-2}$ and $\delta = 0.1$), then the SI method yields a predicted distribution of potential growth (Fig. 7) that is in good agreement with the observed species-specific distribution of *Enteromorpha* spp. and *Ulva* spp. shown in both Figs. 3 and 4.

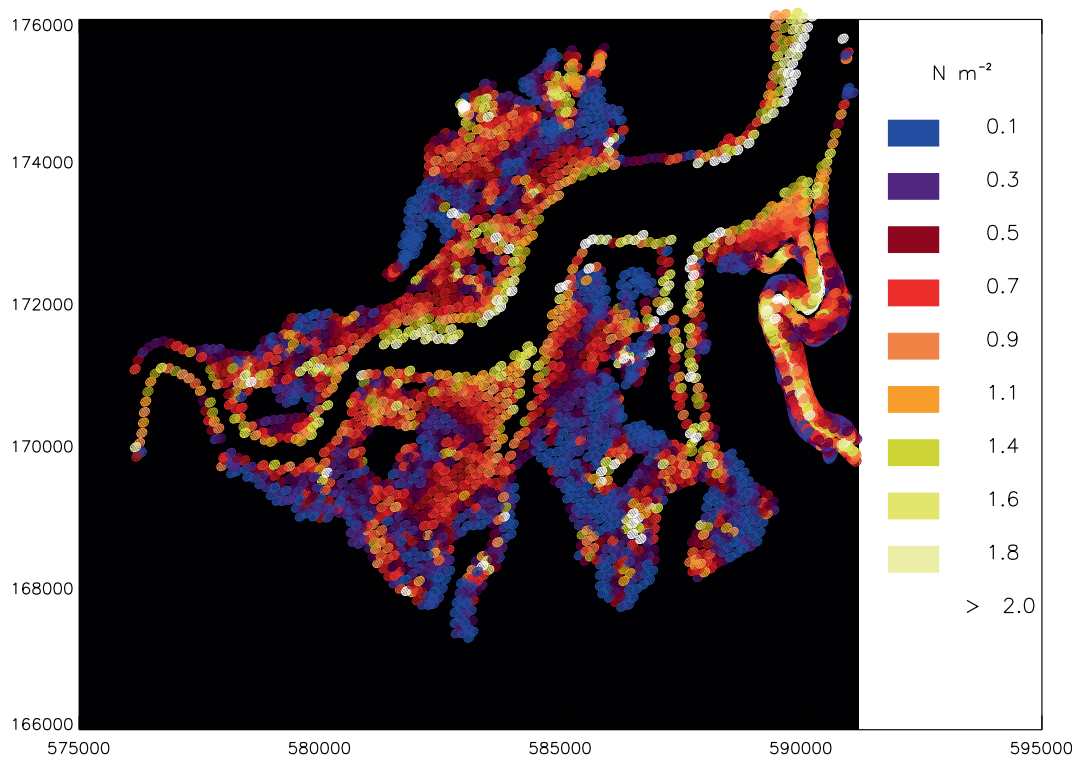


Fig. 5 Predicted distribution of peak bed stress in the Medway (N m^{-2}) based on the TELEMAC-2D model of the Thames and Medway estuaries (HR Wallingford, UK)

Indeed, the distribution of TELEMAC-derived peak bed stress (Fig. 5) on its own visually correlates remarkably closely with the observed distribution of *Enteromorpha* spp. and *Ulva* spp.

Application of the macroalgal model

Before looking at the spatial prediction of cover, a comparison was made with the three-year time series of observed macroalgal (*Ulva* spp.) biomass collected at Motney Hill and Raspberry Hill by the Environment Agency (unpublished data). The observed biomass values (Fig. 8) show significant variability, both inter-annually and in terms of spatial variability at the site (indicated by the error bars in the plots). That said, as a first attempt, this study was concerned with average conditions, and model simulations were run with the same idealized light and nutrient forcing for each year, yielding a repeating annual cycle. No attempt was made to reproduce the observed inter-annual variability but only to obtain a reasonable overall magnitude of the biomass over the three years for which observations were available. Thus, for year

1 at Motney Hill and year 2 at Raspberry Hill, the model significantly over-predicts observed values, although for the final year, the model reproduces observations rather well at both locations. In terms of verifying the model behaviour, consideration of the shape of the macroalgal biomass curve over the year is more important than the overall magnitude (which was partly set by calibration). Across the two sites, there was good correlation ($r = 0.82$, $P < 0.001$) between modelled biomass (mean for each month) and observed biomass (Fig. 8C) but on average the model over-predicted the total biomass by 100% with $\text{Ob} = \text{Mod} \cdot 0.5 + 0.05$ ($r^2 = 0.68$, $P < 0.001$). The observations suggest that biomass increases through the spring and summer months, dropping to a low value for the final survey in November (the clear exception is at Motney Hill for the first year). This is reproduced by the model predictions that show a steady increase in biomass from spring to early autumn when decreased light and temperature act to curtail growth. The significance of the observed steady increase through the season is that substantial nutrient limitation cannot be occurring, otherwise

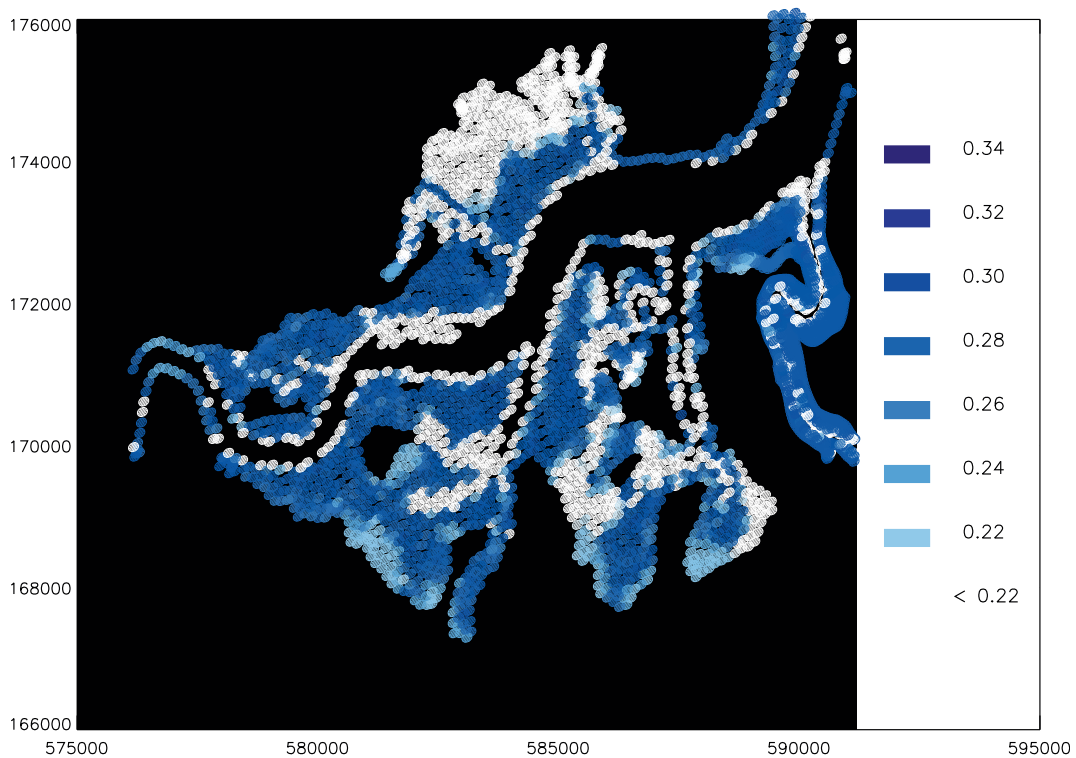


Fig. 6 Suitability index score with higher values indicating greater potential sites for macroalgal growth. A value less than 0.22 is taken as implying no growth. The score takes account of

the balance between light limitation low in the intertidal zone and nutrient supply limitation higher up

biomass would level off before the end of the growing season.

The predictions of non-species-specific biomass for August (Fig. 9) yielded a distribution that visually is indistinguishable from that produced by the SI method (Fig. 6), although it needs to be remembered that one is a non-dimensional score while the other is an actual biomass prediction. Visual comparison with the observed cover shows many similarities, especially compared to the August 2002 survey (Fig. 3). A further simulation was carried out in which bed stress was included as an inhibitor of macroalgal growth. In this simulation, macroalgal mortality begins to be significant at locations where peak bed stress exceeds 0.5 N m^{-2} . This threshold value, although reasonable, was not set by reference to any independent external data. Again, the resulting biomass prediction for August (Fig. 10) was visually very well correlated to the non-dimensional score produced from the SI approach (Fig. 7) and a good match to the observed distributions of *Enteromorpha* spp. and *Ulva* spp. The predictive approaches

identified the main areas of occurrence remarkably well (the Ham Ooze, Copperhouse Marshes and Strawberry Hill), although again there are discrepancies. For example, the predicted distribution at Strawberry Hill has green macroalgae close to the main channels, while the observed coverage is at the landward edge of the intertidal zone. As noted for the SI approach, the resulting distribution of growth is strongly linked to the bed stress distribution, which on its own appears to be a remarkably good predictor of the presence of green macroalgae.

Discussion

In the first instance, the results show that both the SI and the macroalgal growth modelling approaches are robust in predicting the distribution of all macroalgae, i.e. the approach successfully describes the intertidal zone from both a physical and (for the growth model) a nutrient perspective. It also indicates that bed stress is very significant and powerful in explaining the

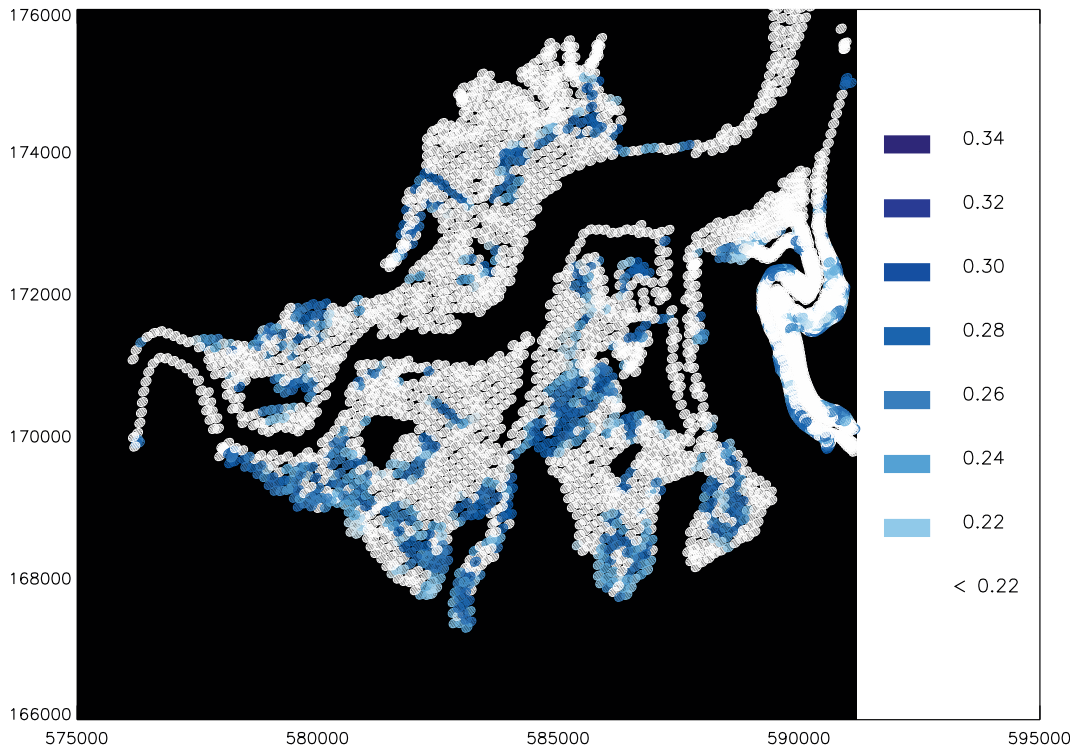


Fig. 7 As for Fig. 6, but with the bed-stress modifier penalizing locations where peak bed stress exceeds 0.5 N m^{-2}

distribution of the problem green algae *Enteromorpha* spp. and *Ulva* spp. (compare Fig. 3 and Fig. 5). The likely rationale behind the effect of bed stress is that both *Spartina* spp. and fucoid species are physically more robust, as evidenced by their classic distribution on exposed shorelines, whereas the green species *Enteromorpha* spp. and *Ulva* spp. are more typical of sheltered shores (Ballentine, 1961).

Light limitation has the effect of reducing growth rates from optimal values and, at some locations, inhibiting growth entirely. Given that large areas of macroalgal growth are observed in the Medway, clearly light is sufficient for growth to occur at many locations, even if the growth rates are inhibited. It is likely that strict nutrient limitation for nitrate rarely occurs in the Medway because concentrations throughout the year are far above the established limiting values for *Enteromorpha* spp. and *Ulva* spp. (Lapointe & Tenore, 1981; Pedersen and Borum, 1997). Although not entirely beyond question (because of the significant inter-annual variability and limited time resolution in the observations), this conclusion is supported by the observed steady

increase in the macroalgal biomass over the growing season (Fig. 8). The macroalgal growth model simulations are consistent with this interpretation. Nevertheless, the dependence of the modelled biomass density on the refresh rate parameter indicates that, although not limiting, some inhibition of growth due to a local draw-down of nutrients in intertidal regions is occurring in the simulations. We note in passing that observed concentrations of ammonium in the estuary channels ($\sim 0.2 \text{ mg l}^{-1}$) are well below the half-saturation constant (0.7 mg l^{-1}) used in the modelled Menten–Michaelis uptake function (Eq. 8 in the supplementary material—Appendix) so that uptake for this nitrogen species is expected to be strongly inhibited. Thus, STW sources, which locally raise ammonium concentrations, potentially provide an important additional nitrogen source, removal of which would be expected to lead to decreases in macroalgal growth.

The situation of steady, non-nutrient limited growth in the Medway contrasts with the typical growth pattern seen in other, less turbid UK estuaries, such as Langstone Harbour (Trimmer et al., 2000;

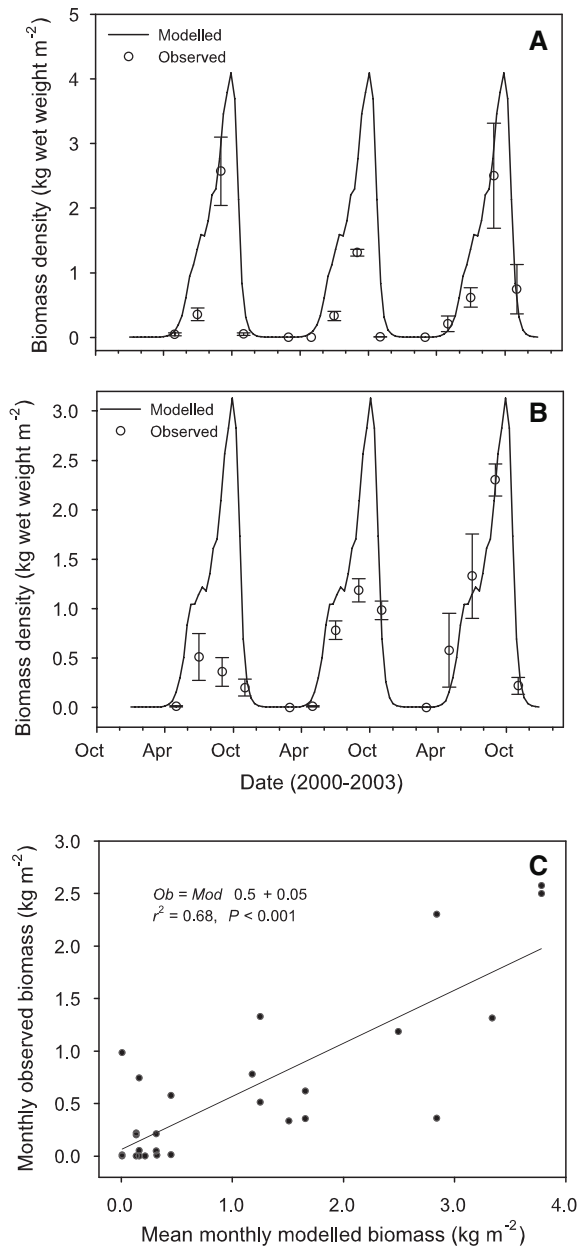


Fig. 8 Monthly observed and modelled wet-weight biomass densities (kg m^{-2}) at **A** Raspberry Hill and **B** Motney Hill. Observed values are averages from four replicate quadrants with standard error bars (observations taken by the Environment Agency, UK, unpublished data). **C** Scatter plot of modelled biomass (mean monthly value) against mean observed biomass for each respective month. *Note:* slope of the 1st order regression is significantly different to 1, i.e. the model over-estimates biomass but the 'shape' is consistent

Pye, 2000). As shown in Cefas (2003), the annual pattern of macroalgal density in Langstone Harbour can be explained primarily as a direct response to

nutrient limitation. Relatively clear waters allow peak values of biomass to be reached in late spring leading to extensive areas of macroalgal growth that bring nutrient concentrations down to limiting values. The standing stock thereafter decreases before remaining approximately constant during summer, supported by input from coastal water, direct sewage inputs and remineralization of organic material in the bed sediments (Trimmer et al., 2000).

The simpler SI approach and the full macroalgal growth model show very similar results for macroalgal distribution. Is it therefore necessary to use a full macroalgal growth model for the Medway? To answer this, it should be noted that the SI yields only a non-dimensional number rather than a biomass prediction but for many practical applications, biomass density is also required. For example, if the potential benefits and risks of changes in anthropogenic inputs are to be evaluated then a prediction of the resulting change in biomass needs to be made requiring use of a model that includes macroalgal growth. A further point is that the present implementation of the SI method does not include a clear cut-off at which no growth is predicted; such regions are only given a 'low' value that the user must then interpret as implying no macroalgae. The macroalgal growth model yields a clear cut-off for presence or absence of macroalgae based on the balance of growth and loss rates. Nonetheless, the SI is relatively easy to understand and to apply and has potential as a preliminary assessment tool. At present, it is not clear whether the approach is applicable to other estuaries. Results reported in Cefas (2003) for estuaries on the UK south coast show some agreement between observed cover and the results of this method. However, more detailed studies are required to confirm this.

Whilst both the SI method and macroalgal growth model predictions of intertidal vegetation cover appear robust, it is important to appreciate their limitations. The calculation of the tidal inundation is based on chart-derived bathymetric data, and whilst clearly representative of intertidal depths, such depths may change over time depending on local sediment transport, deposition etc. Both approaches require estimates of tidal inundation and drying and calculations of bed stresses in the intertidal zone. Highly simplified methods can be applied to derive these quantities, for example, by taking bathymetry and applying a single sinusoidal tidal curve based on

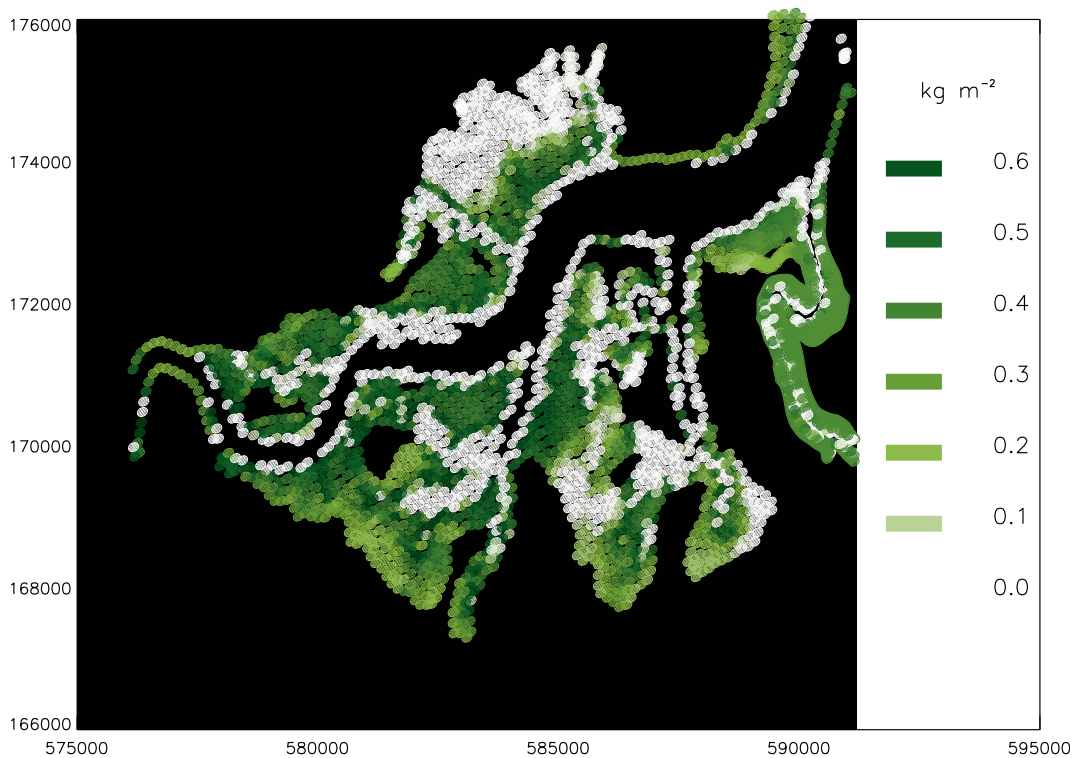


Fig. 9 Macroalgal growth model predictions of biomass (kg m^{-2} dry) for mid-August

observed tidal range to obtain tidal exposure, and perhaps relating bed stress to intertidal slope. Nevertheless, hydrodynamic modelling is probably required to have any real confidence about these calculations. However, hydrodynamic modelling of complex wetting and drying changes in the intertidal zone is extremely challenging and difficult to validate. In this study, light attenuation was based on an average SSC concentration for the entire Medway system. Although hydrodynamic models can include this variable, robust predictive modelling of suspended load in the intertidal zone is probably beyond present modelling capabilities. Unfortunately, measurement of this quantity in the intertidal zone also entails significant challenges, thus presenting problems for more detailed modelling of the role of light limitation. Nevertheless, the results from this study suggest that even with rather crude assumptions (uniform average value based on measurements in the main channels), good agreement with observed distributions of intertidal growth is possible.

In a broader context, this study provides strong evidence that while nutrient loads are undoubtedly a necessary factor for the occurrence of problematic

quantities of green macroalgae, they are not in themselves sufficient. For relatively turbid estuaries with significant tidal flows, abiotic factors (light, tidal inundation and drying, and hydrodynamic bed stress) play a key role in determining the distribution and density of macroalgae (Alvera-Azcarate et al., 2003). The importance of bed stress in particular as a predictor of green macroalgal growth is supported by other studies (Lowthion et al., 1985; Trancoso et al., 2005). From this, we might advance the hypothesis more generally that the occurrence of significant quantities of problem green algae will only take place in intertidal regions that have extended areas where current speeds are low.

The methodology presented here can be taken forwards in a number of ways. Initially and most importantly, application to other estuaries is required to assess the generality of the results. Further work should also consider inter-annual variability. Observational evidence (Figs. 3, 4, 8) suggests significant inter-annual variability in the magnitude, and to a lesser extent location, of macroalgal cover, presumably driven by annual differences in local conditions, for example, light, hydrology, grazing etc. Also,

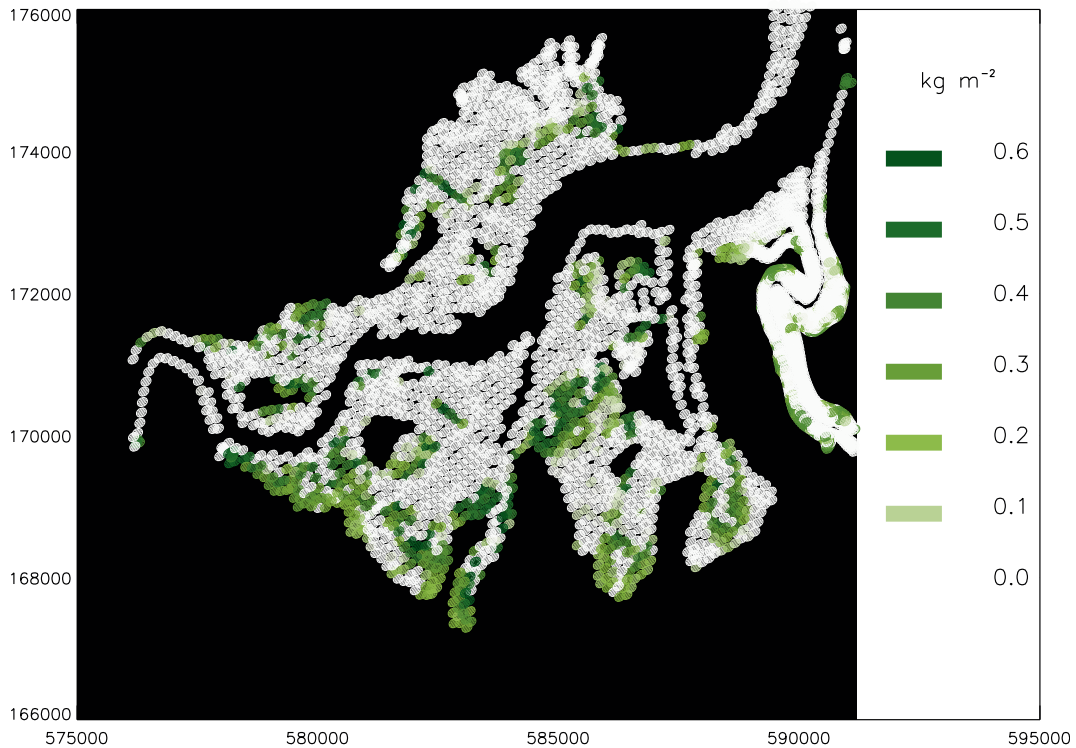


Fig. 10 As for Fig. 9, but with enhanced macroalgal mortality where peak bed stress exceeds 0.5 N m^{-2}

progress in predicting the distribution of macroalgal and other species within the intertidal zone would benefit from a multi-species modelling approach. This should include growth models for fucoids (and other groups), with species-specific parameter values for nutrient uptake rates, maximum growth rates, turnover rates, susceptibility to bed stress etc. Competitive exclusion (Gurney & Nisbet, 1998) should lead to different species types occupying favoured niches within the intertidal zone. Coupling of the macroalgal growth model to nutrient transport models is also required if macroalgal models are to be applied more generally, particularly to estuaries where the growth of marine vegetation dominates the nutrient dynamics.

Summary and conclusions

Intertidal zone bathymetry, tidal dynamics and turbidity control the time available for nutrient uptake and photosynthesis and were used as a basis for predicting areas where macroalgae will grow. Tidal

bed stress was also considered as a controlling factor for the presence of macroalgae. With this as a basis, two approaches to predicting the distribution of marine vegetation were applied: (1) a simple 'scoring' method and (2) a full biological macroalgal growth model with nutrient uptake. Both approaches gave reasonable predictions of non-species-specific macroalgal cover in the Medway estuary. Tidal bed stress was found to be a strong predictor of the specific occurrence of *Enteromorpha* spp. and *Ulva* spp., with these species favouring areas of low tidal energy.

Within the Medway estuary, nutrient supply does not appear to play a key limiting role. In contrast, high turbidity means that light limitation is important in restraining growth rates, with peak biomass being reached only late in the growing season. This, together with the enhanced levels of bed stress, probably explains why the hypernutrified Medway estuary does not support a greater total biomass of nuisance macroalgal species and may explain their absence in the intertidal reaches of the Thames, Wash and other hypernutrified estuaries.

Acknowledgements The authors would like to acknowledge the support of the UK Environment Agency (southern region) in funding this work and supplying data. In particular, Dave Lothian who commissioned the work and Alastair Pratt who supplied much useful advice and data. We also acknowledge the work of Elfed Jones at HR Wallingford in setting up and running the hydrodynamic and nutrient transport models of the Thames and Medway estuaries. Support under Cefas contracts DP193 and Defra under contract E2202 is acknowledged for enabling this work to be presented and prepared for publication. We also acknowledge the detailed and useful comments from two anonymous referees that helped to significantly improve the original manuscript.

References

- Aisha, K. A. & E. F. Shabana, 1995. Pulse feeding with nitrate and phosphate in relation to tissue composition and nutrient uptake by some macroalgae from the Red Sea at the Ghardaqa (Egypt). *Journal of Basic Microbiology* 35: 135–145.
- Alvera-Azcarate, A., J. G. Ferreira & J. P. Nunes, 2003. Modelling eutrophication in mesotidal and macrotidal: the role of intertidal seaweeds. *Estuarine, Coastal and Shelf Science* 57: 715–724.
- Atkinson, M. & S. V. Smith, 1983. C:N:P Ratios of benthic marine plants. *Limnology and Oceanography* 28: 568–574.
- Babey, S. K. & R. J. Soffer, 1992. Radiometric calibration of the Compact Airborne Spectrographic Imager (CASI). *Canadian Journal of Remote Sensing* 18: 233–242.
- Ballentine, W. J., 1961. A biologically defined exposure scale for comparative description of rocky shores. *Field Study* 1: 1.
- Bjornstater, B. R. & P. A. Wheeler, 1990. Effect of nitrogen and phosphorus supply on growth and tissue composition of *Ulva fenestrata* and *Enteromorpha intestinalis* (Ulvales, Chlorophyta). *Journal of Phycology* 26: 603–611.
- Brown, K., 2004. Medway CASI Survey, Project PM_0282. UK Environment Agency, Lower Bristol Road, Bath, BA2 9ES, UK.
- Cefas, 2003. Investigation of factors controlling the presence of macroalgae in some estuaries of the Southern Water Region. Centre for Environment, Fisheries & Aquaculture Science. Contract C1642.
- den Hartog, C., 1994. Suffocation of a littoral *Zostera* bed by *Enteromorpha radiata*. *Aquatic Botany* 47: 21–28.
- Dring, M. J., 1986. *The Biology of Marine Plants*. Contemporary Biology. Edward Arnold, London.
- Eilola, K. & V. Stigebrandt, 2001. Modelling filamentous algae mats in shallow bays. EU Life Algae Project, Rapport 2001:38, Länsstyrelsen i Västra Götaland, Göteborg.
- Gurney, W. S. C. & R. M. Nisbet, 1998. *Ecological Dynamics*. Oxford University Press, New York.
- HR Wallingford, 2004. Water quality model development for the Medway and Swale. Report EX4662, Dec 2004.
- Jones, J. E., 2005. An intercomparison between finite difference and finite element (TELEMAC) approaches to modelling west coast of Britain tides. *Ocean Dynamics* 55: 178–198.
- Lapointe, B. E. & K. R. Tenore, 1981. Experimental outdoor studies with *Ulva fasciata* Delile. I. Interaction of light and nitrogen on nutrient uptake, growth, and biochemical composition. *Journal of Experimental Marine Biology and Ecology* 53: 135–152.
- Lavery, P. S., R. J. Lukatelich & A. J. McCoomb, 1991. Changes in the biomass and species composition of macroalgae in a eutrophic estuary. *Estuarine, Coastal and Shelf Science* 33: 1–22.
- Lowthion, D., P. G. Soulsby & M. C. M. Houston, 1985. Investigation of a eutrophic tidal basin: 1. Factors affecting the distribution and biomass of macroalgae. *Marine Environmental Research* 15: 263–284.
- Martins, I. & J. C. Marques, 2002. A model for the growth of opportunistic macroalgae (*Enteromorpha* spp.) in tidal estuaries. *Estuarine, Coastal and Shelf Science* 55: 247–257.
- Martins, I. & M. Pardal, 2001. Hydrodynamics as a major factor controlling the occurrence of green macro-algal blooms in a eutrophic estuary: a case study on the influence of precipitation and river management. *Estuarine, Coastal and Shelf Science* 52: 165–177.
- Mohsen, A. F., A. F. Khaeafa & A. Metwalli, 1974. Effect of different nitrogen sources on growth, reproduction, amino acid, fat and sugar contents in *Ulva fasaata* Delile. *Botanica Marina* 17: 218–222.
- Nedwell, D. B., L. F. Dong, A. Sage & G. J. C. Underwood, 2002. Variations of the nutrients loads to the mainland U.K. estuaries: correlation with catchment areas, urbanization and coastal eutrophication. *Estuarine, Coastal and Shelf Science* 54: 951–970.
- 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.
- Pratt, A., 2001. Review of sensitive water status, Medway estuary, Kent. Environment Agency (Southern Region) Report, ref 10173.
- Pye, K., 2000. The effects of eutrophication on the marine benthic flora of Langstone Harbour, south coast of England. PhD thesis, University of Portsmouth.
- Rosenberg, G. & J. Ramus, 1982. Ecological growth strategies in the seaweeds *Gracilaria foliifera* (Rhodophyceae) and *Ulva* spp. (Chlorophyceae): soluble nitrogen and reserve carbohydrates. *Marine Biology* 66: 251–259.
- Sanders, R., C. Klein & T. Jickells, 1997. Biogeochemical nutrient cycling in the upper Great Ouse estuary, Norfolk, U.K. *Estuarine, Coastal and Shelf Science* 44: 543–555.
- Sfriso, A., 1995. Temporal and spatial responses of growth of *Ulva rigida* C. Ag. to environmental and tissue concentrations of nutrients in the Lagoon of Venice. *Botanica Marina* 38: 557–573.
- Sfriso, A., B. Pavoni & A. Marcomini, 1989. Macroalgae and phytoplankton standing crops in the central Venice Lagoon: primary production and nutrient balance. *Science of the Total Environment* 80: 139–159.
- Solidoro, C., G. Pecenic, R. Pastres, D. Franco & C. Dejak, 1997. Modelling macroalgae (*Ulva rigida*) in the Venice

- Lagoon: model structure identification and first parameter estimation. *Ecological Modelling* 94: 191–206.
- Trancoso, A. R., S. Saraiva, L. Fernandes, P. Pina, P. Leitao & R. Neves, 2005. Modelling macroalgae using a 3D hydrodynamic-ecological model in a shallow, temperate estuary. *Ecological Modelling* 187: 232–246.
- Trimmer, M., D. B. Nedwell, D. B. Sivyer & S. J. Malcolm, 2000. Seasonal organic mineralisation and denitrification in intertidal sediments and their relationship to abundance of *Enteromorpha* sp. and *Ulva* sp. *Marine Ecology Progress Series* 203: 67–80.
- Tubbs, C. R. & J. M. Tubbs, 1983. Macroalgal mats in Langstone Harbour, Hampshire, England. *Marine Pollution Bulletin* 14: 148–149.

Short-term effect of oxic to anoxic transition on benthic microbial activity and solute fluxes in organic-rich phytotreatment ponds

M. Bartoli · L. Vezzulli · D. Nizzoli · R. Azzoni ·
S. Porrello · M. Moreno · M. Fabiano · P. Viaroli

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 123–136.
DOI: 10.1007/s10750-009-9761-5 © Springer Science+Business Media B.V. 2009

Abstract Manipulative experiments to test the short-term effect of oxygen depletion events on microbial activity and benthic fluxes in organic-rich sediments were carried out in March and June 2004. Oxic–anoxic transitions were induced by prolonged dark incubation of sealed sediment cores collected in phytotreatment ponds. Benthic fluxes of oxygen (O₂), carbon dioxide (CO₂), inorganic nutrients, and free sulfides were measured before (oxic) and after (anoxic) the transition occurred. A multifactorial

design was employed for monitoring esoenzymatic activity, heterotrophic bacterial production, total prokaryotic abundance, actively respiring bacterial cells, and the biochemical composition of sedimentary organic matter. The oxic to anoxic transition resulted in a significant increase of esoenzymatic activity and bacterial production in March, due to the profound modification of the benthic community and the release of labile organic compounds which followed the onset of anoxia. In parallel, net efflux rates of dissolved inorganic carbon (DIC) and ammonium (NH₄⁺) sharply decreased, soluble reactive phosphorus (SRP) influx reversed, and sulfide was buffered within the oxidized sediments. From March to June, ponds evolved toward oxygen deficit and reducing conditions in the upper sediment horizon, losing benthic fauna and biogeochemical buffering capacity. Thus, the oxic to anoxic transition had a much smaller effect on microbial activity and net flux exchange, while S²⁻ was consistently delivered from the sediment to the water column. Overall data from this study suggest that the short-term response of benthic microbial activity and solute fluxes to anoxic events may have a significant impact on sediment biogeochemistry (e.g., at the oxic–anoxic interface), and that this impact may vary greatly depending on the sediment features, mainly its organic content and redox condition.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from
the Second International Symposium on Research and
Management of Eutrophication in Coastal Ecosystems,
20–23 June 2006, Nyborg, Denmark

M. Bartoli (✉) · D. Nizzoli · R. Azzoni · P. Viaroli
Department of Environmental Sciences, University of
Parma, Parco Area Delle Scienze 33a, 43100 Parma, Italy
e-mail: bartoli@dsa.unipr.it

L. Vezzulli
Department of Biology, University of Genoa,
Viale Benedetto XV, 5-16132 Genoa, Italy

S. Porrello
ISPRA - ex ICRAM (Central Institute for Marine
Research), Via di Casalotti 300, 00100 Rome, Italy

M. Moreno · M. Fabiano
Department for the Study of Territory and its Resources,
University of Genoa, Corso Europa, 26-16132 Genoa,
Italy

Keywords Oxic–anoxic transition ·
Microbial activity · Solute fluxes ·

Organic-rich sediments · Biogeochemical buffers · Eutrophication

Introduction

In many brackish and shallow marine areas, increasing impacts of human activities arising from eutrophication, tourism, and aquaculture have resulted in dramatic changes in the structure and functioning of the benthic systems (Castel et al., 1996; Crossland et al., 2005; Rabalais et al., 2007). Demonstrated examples of such changes include the shift of primary producer communities from rooted phanerogams to drifting macroalgae, different abundance and community structures of meio- and macrofauna, decoupling between sedimentation and regeneration processes, anoxia, and occurrence of dystrophic events (Gee et al., 1985; Duarte, 1995; Schramm, 1999; De Wit et al., 2001).

Notwithstanding the great progress in studying processes that have major ecological consequences during the development of marine eutrophication (Kristensen et al., 1995; Jørgensen, 1996), crucial steps in the ecosystem shift from hypoxia to full anoxia are still questioned (Meyer-Reil & Köster, 2000). The unpredictable and non-linear behavior of stressed coastal ecosystems requires accurate measurements and powerful modeling tools to understand the key microbial and biogeochemical processes (Meyer-Reil & Köster, 2000). With this aim, several studies have been carried out to investigate benthic metabolism under the different environmental conditions which are commonly found before and after eutrophication is established. For example, the qualitative nature of the mineralization processes within oxic and anoxic environments has been investigated, leading to a general conclusion that microbial biomass production and organic matter mineralization display lower rates under anoxic conditions, in particular, when the organic carbon bulk is mainly refractory (Kristensen et al., 1995; Sun et al., 1997; Amtoft Neubauer et al., 2004; Lomstein et al., 2006). The occurrence of steep vertical oxygen gradients and the establishment of an oxic–anoxic interface (OAI) within the sediment layers constitute a natural model for investigation of the oxic to anoxic transition (Brune et al., 2000). Studies on the structure and activity of the benthic community at

the OAI have been carried out in sediment and microbial mats, where they highlighted a complex structural and functional organization of microbial groups in relation to a narrow stratification of physicochemical variables (Canfield & DesMarais, 1993; Bernard & Fenchel, 1995). Investigations applying molecular techniques have confirmed the occurrence of a distinct stratification of microbial populations in the vicinity of OAI and the identification of this area as the basis of key biogeochemical processes such as the aerobic terminal oxidation of reduced compounds by lithotrophic bacteria (Fenchel et al., 1998).

However, a full understanding of the ecological role which the OAI plays in marine sediment remains challenging. Holmer (1999) investigated the dynamic nature of the OAI, assessing the effect of oxygen depletion on organic matter mineralization in little to non-bioturbated marine sediments. Contrasting patterns were evidenced during the oxic to anoxic transition, with high mineralization rates recorded within the first few hours. The short-term response of the microbial activity following an oxygen depletion event is likely to play an important role in sediment biogeochemistry due to the dynamic nature of the OAI at micrometer scales. In fact, the oscillation of the OAI within the sediment matrix is primarily influenced by the sedimentation rates and the organic input. It is also dependent on diffusion processes, bioturbation activities, and the presence of rizo-phytes. Complex redox mosaics are the rule in marine sediments (Aller, 1994), and it is reasonable to assume that repetitive oscillations of the OAI make short-term responses of microbial activity quantitatively significant to the overall sediment biogeochemistry. In this context, organic enrichment of the coastal environment may profoundly affect OAI establishment and dynamics, thus affecting benthic metabolism and biogeochemistry (Amtoft Neubauer et al., 2004; Lomstein et al., 2006).

This study investigates the short-term effects of oxygen depletion on benthic microbial activity and biogeochemistry using a manipulative experimental approach. In particular, the authors present results from two experiments in which microbial activity and solute fluxes across the sediment–water interface were measured following induced anoxia in eutrophic brackish ponds receiving effluents from a marine fish farm.

Materials and methods

Study area

Surface sediments were collected from phytotreatment ponds receiving effluents from a land-based fish farm, which is located close to the Orbetello lagoon (Tyrrhenian Sea, central Italy) (Fig. 1). The phytotreatment plant consists of four ponds arranged in series with a surface area of $\sim 2,600 \text{ m}^2$ each and a mean depth of $\sim 0.7 \text{ m}$. Total water volume is $9,000 \text{ m}^3$ and $7,000 \text{ m}^3$ in the fish and phytotreatment ponds, respectively, and water flow is 140 l s^{-1} . Detailed descriptions of the fish farm, phytotreatment ponds, and main ecological studies were reported by Porrello et al. (2003) and Bartoli et al. (2005).

Owing to the limited hydrodynamic regime, water shallowness, and high organic load, phytotreatment ponds feature environmental conditions commonly found in coastal eutrophic environments, such as highly productive marginal areas of Mediterranean coastal lagoons that are subjected to summer collapse of primary production and anoxia (Castel et al., 1996; Viaroli et al., 1996; De Wit et al., 2001).

Water and sediment sampling, core maintenance, and incubation

Two intensive field campaigns were conducted in March and June 2004; on each occasion, water and

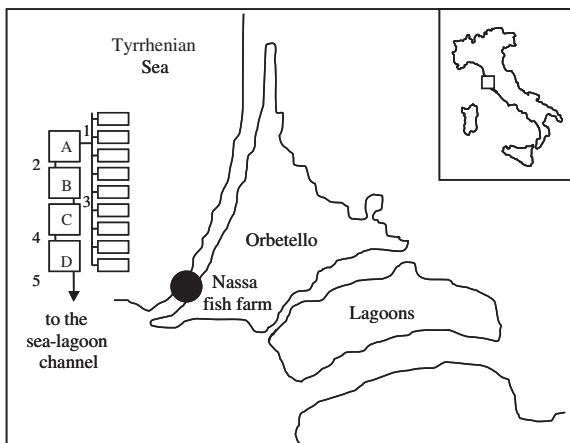


Fig. 1 Location of the study area and schematic representation of the fish farm and phytotreatment ponds. *Squares* represent phytotreatment ponds A–D; *numbers* indicate inlet and outlet sampling sites for water. Sediments were collected approximately at the center of each pond

sediment samples were collected. Water was sampled at the inlet and outlet of the phytotreatment plant according to the procedure described by Porrello et al. (2003) and filtered for the determination of dissolved and particulate nutrients.

Intact sediment cores (transparent plexiglass liners, 8 cm i.d., length 30 cm) were collected manually at each pond. Sediment was leveled to a 12-cm height, and the water column overlying sediments was approximately 14 cm. On each sampling date, six cores were collected from each pond for a total of 24 sediment cores. Immediately after collection, the cores were submerged in tanks containing in situ aerated water and were left overnight to allow for sediment stabilization. Sediment–water fluxes of O_2 , dissolved inorganic carbon (DIC), NH_4^+ , soluble reactive phosphorus (SRP), and S^{2-} were measured in 12 cores via dark incubations starting from the morning after the sampling. The remaining cores were left submerged in oxic water for later surface sediment characterization.

Incubated cores were sealed with floating lids; water stirring was ensured by Teflon-coated magnetic bars rotating at 40 rpm. A detailed description of core pre-incubation and incubation techniques employed in this study is reported in Dalsgaard et al. (2000). Compared to standard flux measurements, the dark incubation was prolonged up to the establishment of anoxic conditions in the water column. The incubation time was set on the basis of oxygen measurements to entirely deplete the oxygen reserve in the water column and was protracted under anoxic conditions for a maximum of $\sim 20 \text{ h}$. Following this rationale, the March incubation lasted 44 h, while the June incubation lasted 25 h. Water samples were collected with different timing under oxic and anoxic conditions for a total of eight samplings. Following each collection, sampled water (representing less than 10% of the water volume inside the core) was replaced with water from the incubation tank. Rates for the oxic part were estimated generally within the initial 2 h, while rates in the anoxic part were calculated after a few hours (2–3) from the disappearance of dissolved oxygen in the water column of each core.

Sediment characterization

The upper sediment horizon (0–2 cm depth) of the cores maintained in oxic conditions was sliced, mixed

thoroughly, and analyzed for the determination of sediment features and microbial activity. In addition, microbial activity and organic matter quality were measured in the incubated cores at the end of the incubations.

Total organic carbon (TOC) and total nitrogen (TN) were determined with a CHNS-O EA 1108 Elemental Analyzer (Carlo Erba); total phosphorus (TP) was determined spectrophotometrically after acid extraction from ashes (Aspila et al., 1976). Proteins (PRT) were determined according to Hartree (1972); carbohydrates (CHO) were analyzed according to Dubois et al. (1956).

The total labile iron pool (LFe, defined as LFe(III) + LFe(II)), acid volatile sulfide (AVS, which includes S^{2-} and FeS), and the chromium-reducible sulfur (CRS, which is mainly composed of FeS₂ and S⁰) were analyzed with methods reported by Azzoni et al. (2005).

Enzymatic activity

Analyses of aminopeptidase (L-Leucine-4-methylcoumarinyl-7-amide, Leu-MCA) and β -glucosidase (MUF- β -glucopyranoside, Glu-MUF) enzymatic activity were carried out by adding 1 ml of Leu-MCA and Glu-MUF, respectively (final concentration 200 μ M), as described by Hoppe (1993). Undisturbed sediment sub-samples (0.5 ml) were incubated for 1 h in the dark and at in situ temperature and then centrifuged (3,000 rpm, 1,000 \times g, 5 min). Enzymatic rates were assessed by fluorometric analysis at 380 nm excitation, 440 nm emission (Meyer-Reil, 1986) for MCA, and at 365 nm excitation; 455 nm emission (Hoppe, 1993) for MUF. Data were normalized to dry weight after desiccation (60°C, 24 h) and reported as nanomoles of Leu-MCA or Glu-MUF released per gram of sediment dry weight h⁻¹. Esterase activity was assayed with fluorescein diacetate (FDA, Sigma, St. Louis, USA) according to Battin (1997). Samples were incubated at a final concentration of 200 μ M FDA. After a 30-min incubation period, the hydrolysis was stopped with acetone, and the samples were stored in ice. Fluorescein was extracted by sonicating (45 s, 30 W output) the suspension, and particles were removed by centrifugation (8°C, 20 min, 5,000 rpm). The absorbance of the supernatant was measured spectrophotometrically at 490 nm. Enzymatic activity was

converted into equivalents of C mobilized assuming that 1 nmol of substrate hydrolyzed corresponded to 72 ng of mobilized C (Manini et al., 2003).

Heterotrophic bacterial production

Heterotrophic benthic bacterial production (HBP) was measured using ³H-leucine incorporation (Van Duyl & Kop, 1994). Sediment sub-samples (200 μ l) added to 26 μ l of ³H-leucine (6 μ Ci final concentration per sample) were incubated for 1 h in the dark at in situ temperature. After incubation, bacterial C incorporation was stopped with 1.7 ml of 80% v/v ethanol before scintillation counting. Sediment blanks were made by adding ethanol immediately before ³H-leucine addition. Data were normalized to sediment dry weight after desiccation (60°C, 24 h).

Total prokaryotic abundance (TPA) and actively respiring cells (ARC)

For a total direct count of prokaryote cells, sediment samples were sonicated three times (Sonifier Labor 2000, 195 W for 1 min). Sub-samples were diluted 250 times. The number of prokaryote cells in 0.2- μ m black Nucleopore filters was estimated using the Acridine Orange Direct Count technique (Hobbie et al., 1977) with epifluorescence microscopy (Zeiss Universal Microscope). Actively respiring cells in the sediment were determined following Proctor & Souza (2001). In brief, 1.3-mm depth of seawater was gently pipetted onto homogenized sediment slurries (0.5 ml). Under red light, 250 μ l of the fluorogenic redox dye 5-cyano-2,3-ditolyl tetrazolium chloride (CTC) working stock was injected into the 2-cm sediment slurries with a 26-G needle, lifting the needle upward from the bottom through the sediment section. The final concentration of CTC was 5.5 mM (Bhupathiraju et al., 1999). The assay was incubated in the dark for 4 h at room temperature, and the reaction was stopped with 3 ml of 20% formalin and stored at -20°C. An aliquot of the supernatant was filtered through a 0.2-mm pore-size black Nucleopore membrane filter. Then, microscope slides were prepared, and the numbers of actively respiring bacteria were counted by epifluorescence microscopy (Zeiss Universal Microscope).

Analyses of solutes and particulate carbon, nitrogen, and phosphorus

Dissolved oxygen was determined with the Winkler method (APHA, 1975) and with polarographic Clark-style microelectrodes (Unisense, DK). Total dissolved inorganic carbon was measured by titration with 0.1 N HCl (Talling, 1973). Ammonium (Bower & Holm Hansen, 1986), SRP (Valderrama, 1977), and sulfide (Cline, 1969) were determined spectrophotometrically. Water samples were filtered on GF/F Whatman filters for particulate organic carbon (POC) and particulate nitrogen (PN) and processed with a CHNS-O EA 1108 Elemental Analyzer (Carlo Erba). Particulate phosphorus (PP) was determined as soluble reactive phosphorus following persulfate oxidation and extraction from filters (Valderrama, 1981).

Statistical analyses

A two-way ANOVA was used to test the null hypothesis that solute flux values and microbial activity rates were the same in the oxic and anoxic cores following the induced oxic–anoxic transition. Factors in the model were “Transition [T]” (two levels, orthogonal, fixed) and “Pond [P]” (four levels, orthogonal, fixed). Three replicates were employed in the analysis. The effect of the oxic to anoxic transition was then investigated by testing whether the main factor [T] was statistically significant ($P < 0.05$). In order to interpret statistically significant interactions among the different treatments, a SNK post-hoc test was carried out. Prior to the analysis, the homogeneity of variance was tested by Cochran’s test and, when necessary, data were appropriately transformed. Pearson correlation analysis was carried out to test relationships among microbial and environmental variables. All statistical tests and correlation analyses were performed using the MATLAB Statistics Toolbox (Version 6.1; TheMathWorks).

Results

General features of effluents and surface sediments

Chemical analyses on effluents from the fish farms indicated an increase of nitrogen species

Table 1 Physicochemical features of the fish-farm effluents entering into phytotreatment pond A, and net daily balances of particulate carbon, nitrogen, and phosphorus in March and June 2004. For the latter, a negative balance indicates a retention of particulate matter within the phytotreatment ponds (= sedimentation)

| Variables | March | June |
|-----------------------------------|-------------|--------------|
| T (°C) | 14.6 ± 0.8 | 22.6 ± 0.9 |
| O ₂ (%) | 75.7 ± 11.4 | 66.3 ± 4.7 |
| NH ₄ ⁺ (μM) | 37.7 ± 5.1 | 61.4 ± 4.6 |
| SRP (μM) | 3.2 ± 0.5 | 2.8 ± 0.5 |
| TDN (μM) | 64.7 ± 5.6 | 113.1 ± 30.4 |
| TDP (μM) | 3.4 ± 0.6 | 3.1 ± 0.4 |
| POC (μM) | 61 ± 4 | 95.8 ± 11.8 |
| PON (μM) | 8 ± 0.2 | 14.8 ± 1.33 |
| PP (μM) | 0.7 ± 0.2 | 1.2 ± 0.3 |
| C-sed (mol d ⁻¹) | −367 | −1,095 |
| N-sed (mol d ⁻¹) | −46 | −163 |
| P-sed (mol d ⁻¹) | −2.7 | −13 |

T temperature, O₂ oxygen, NH₄⁺ ammonium, SRP soluble reactive phosphorus, TDN total dissolved nitrogen, TDP total dissolved phosphorus, POC particulate organic carbon, PON particulate organic nitrogen, PP particulate phosphorus, C-sed particulate carbon, N-sed particulate nitrogen, P-sed particulate phosphorus

concentrations from March to June (Table 1). The increase in suspended solids and development of algal mats within the phytotreatment ponds also resulted in higher retention of particulate matter within the four basins (Table 1).

Sediments collected in March were light brown in the upper horizon and bioturbated; benthic macrofauna displayed increasing abundances from pond A to pond D. The dominant species were represented by the Polychaeta *Neanthes caudata* (up to 19,500 ind m⁻², pond B), *Malacoceros fuliginosus* (up to 5,000 ind m⁻², ponds C and D), the Crustacea *Microdeutopus gryllotalpa* (up to 6,000 ind m⁻², pond B), and *Corophium insidiosum* (up to 12,000 ind m⁻², ponds B and C). In June, surface sediments were dark brown and defaunated (Tomassetti, pers. comm. Central Institute for the Scientific and Technologic Research applied to the Sea), with the exception of *Neanthes caudata* (up to 3,100 ind m⁻², pond B).

Total carbon, nitrogen, and phosphorus in surface sediments were similar in the two sampling periods; while proteins increased markedly from March to

Table 2 Main features of surface sediment (upper 2 cm) collected from the four phytotreatment ponds in March and June 2004

| Pond | Date | TOC (%) | TN (%) | TP (%) | PRT (mg g ⁻¹) | CHO (mg g ⁻¹) | LFe (μmol cm ⁻³) | AVS (μmol cm ⁻³) | CRS (μmol cm ⁻³) |
|------|-------|-----------|-----------|-----------|------------------------------|------------------------------|---------------------------------|---------------------------------|---------------------------------|
| A | March | 4.9 ± 1.3 | 0.8 ± 0.1 | 0.2 ± 0.0 | 0.56 ± 0.19 | 1.12 ± 0.56 | 16.2 ± 1.3 | 5.3 ± 2.8 | 57.7 ± 5.0 |
| | June | 4.9 ± 0.6 | 0.9 ± 0.1 | 0.2 ± 0.1 | 5.60 ± 0.27 | 6.14 ± 4.62 | 21.7 ± 6.5 | 26.7 ± 2.2 | 23.8 ± 3.4 |
| B | March | 4.7 ± 0.8 | 0.7 ± 0.1 | 0.2 ± 0.0 | 2.06 ± 0.40 | 5.84 ± 1.42 | 20.2 ± 3.7 | 13.2 ± 6.4 | 61.9 ± 11.8 |
| | June | 6.2 ± 0.4 | 0.9 ± 0.1 | 0.4 ± 0.1 | 8.58 ± 0.74 | 10.56 ± 4.42 | 59.5 ± 3.5 | 27.5 ± 1.8 | 57.4 ± 8.2 |
| C | March | 5.3 ± 0.5 | 0.7 ± 0.1 | 0.3 ± 0.0 | 0.31 ± 0.64 | 8.37 ± 1.24 | 34.6 ± 3.9 | 14.7 ± 3.4 | 105.1 ± 9.0 |
| | June | 3.8 ± 0.6 | 0.5 ± 0.1 | 0.3 ± 0.1 | 7.46 ± 1.51 | 7.73 ± 4.04 | 44.6 ± 9.5 | 25.5 ± 3.1 | 100.8 ± 13.0 |
| D | March | 4.3 ± 0.3 | 0.6 ± 0.0 | 0.2 ± 0.1 | 2.18 ± 0.73 | 7.04 ± 1.47 | 34.0 ± 3.2 | 22.0 ± 1.7 | 75.7 ± 14.5 |
| | June | 5.7 ± 0.6 | 0.9 ± 0.1 | 0.3 ± 0.0 | 6.99 ± 0.40 | 10.47 ± 7.00 | 52.4 ± 5.7 | 22.9 ± 4.0 | 85.0 ± 7.1 |

TOC total organic carbon, TN total nitrogen, TP total phosphorus, PRT proteins, CHO carbohydrates, LFe total labile iron, AVS acid volatile sulfide, CRS chromium-reducible sulfur

June (Table 2); the carbohydrate increase was significant only in ponds A and B. From March to June, total labile iron, which represents the more reactive iron pool, increased in all ponds. In parallel, a significant increase in AVS production was observed mainly in the first basins which receive the greater organic loads. CRS exhibited significant changes only in pond A where it decreased from 57.7 to 23.8 μmol S cm⁻³, while in the other three ponds, it did not undergo significant changes. The total reduced sulfur (TRS = AVS + CRS) remained almost constant from March to June, while an increase was observed from ponds A and B to ponds C and D, thus evidencing a better capacity of accumulating reduced sulfur in the final ponds.

Short-term effect of oxic to anoxic transition on the microbial activity

The oxic–anoxic transition induced a significant increase in esoenzymatic and microbial activities in March 2004, especially for esterase (FDA), which also increased from pond A to pond D (Fig. 2). Heterotrophic bacterial production displayed an even larger increase in all cores and shifted on average from 1,927 ± 432 ng C g⁻¹ h⁻¹ (oxic) to 3,566 ± 427 ng C g⁻¹ h⁻¹ (anoxic) (ANOVA, $p < 0.05$). A significant change in the composition of sedimentary organic matter was also observed following the transition, with an increase in carbohydrate and a decrease in the protein contributions to total biopolymeric carbon. No significant changes were observed in the total prokaryotic density, with 3.6 ± 1.9 cells × 10⁹ g⁻¹ under oxic conditions and

3.5 ± 1.3 cells × 10⁹ g⁻¹ under anoxia (ANOVA, $p > 0.05$). The actively respiring cells were slightly higher in the anoxic (average 37 ± 3%) than in oxic (average 31 ± 3%) cores.

In June, the microbial activity increase following the transition was not further observed (Fig. 2). The aminopeptidase activity displayed higher rates in the oxic cores (ANOVA, $p < 0.05$). No significant changes were observed for glucosidase and esterase rates, or for heterotrophic bacterial production (ANOVA, $p > 0.05$). Furthermore, the total prokaryotic density (19.0 ± 7.9 cells × 10⁹ g⁻¹ oxic and 17.3 ± 9.5 cells × 10⁹ g⁻¹ anoxic) and the number of actively respiring cells (32.0 ± 5.3% oxic and 31.0 ± 4.8% anoxic) displayed similar values in both oxic and anoxic cores.

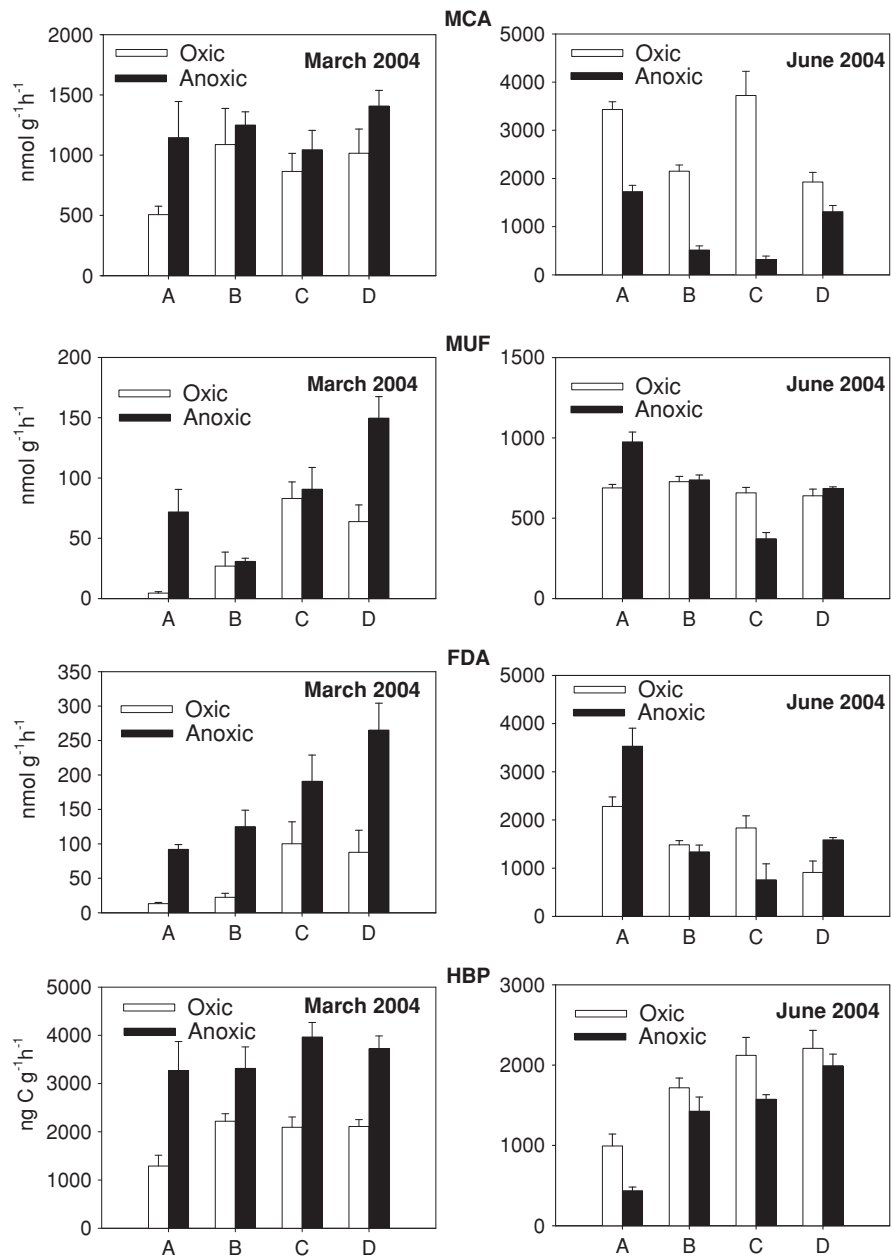
Short-term effect of oxic to anoxic transition on the solute fluxes

An example of the evolution of solute concentrations in the water overlying the sediments in the cores collected at pond A during the oxic–anoxic transition is given in Fig. 3. During the oxic part of the incubation, oxygen consumption was coupled to a net regeneration of inorganic carbon from pond sediments to the water column, while only in June, a great sulfide efflux was detected just after the onset of anoxia.

Rates of solute net exchange across the sediment–water interface calculated in the oxic and anoxic parts of the incubation changed following different patterns among ponds and seasons.

Sediment oxygen demand (SOD) was significantly higher in pond A in March and June (−6.06 ± 0.17

Fig. 2 Rates of esoenzymatic activity and heterotrophic bacterial production in oxic and anoxic sediment cores following the experiment in March and June 2004. *MCA* aminopeptidase, *MUF* glucosidase, *FDA* esterase, *HBP* heterotrophic bacterial production

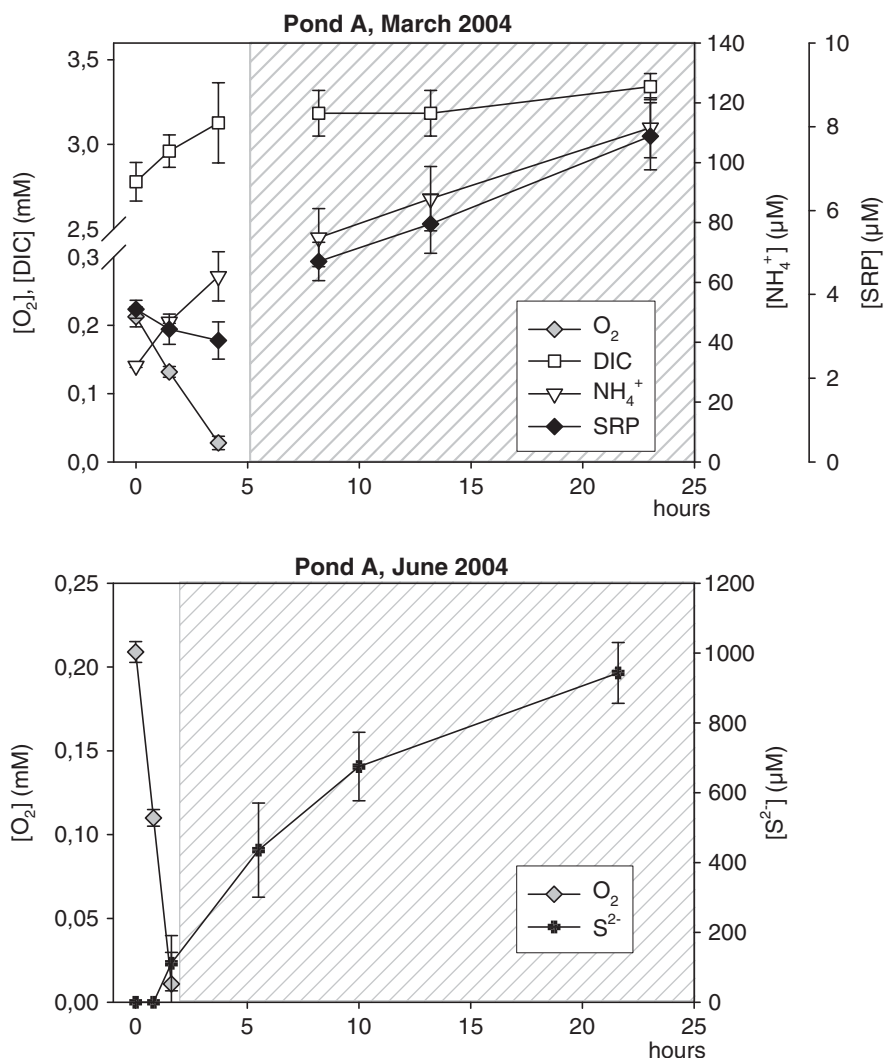


and $-13.89 \pm 1.03 \text{ mmol O}_2 \text{ m}^{-2} \text{ h}^{-1}$, respectively, ANOVA, $p < 0.01$); in ponds B, C, and D, rates were similar and ranged between -1.78 and $-4.21 \text{ mmol O}_2 \text{ m}^{-2} \text{ h}^{-1}$ (Fig. 4).

DIC efflux rates were significantly higher at pond A where they increased sharply from March ($5.78 \pm 1.48 \text{ mmol m}^{-2} \text{ h}^{-1}$) to June ($29.66 \pm 4.19 \text{ mmol m}^{-2} \text{ h}^{-1}$). Rates were not significantly different among ponds B, C, and D and the sampling

dates (ANOVA, $p > 0.05$) even though there was a general tendency toward higher DIC effluxes in June. The respiratory quotient, calculated as the ratio between DIC and $|\text{O}_2|$ dark fluxes (Dilly, 2003) averaged 1.0 in March and 2.1 in June. The shift between oxic and anoxic conditions resulted in a significant decrease of DIC efflux (ANOVA, $p < 0.01$) in the March incubation; the relative decrease was minimum at pond A (-39%) and maximum at pond D

Fig. 3 Evolution of O_2 , DIC, NH_4^+ , SRP, and S^{2-} in the water column during oxic-anoxic transitions in sediment cores collected in pond A in March (*upper panel*) and in June (*lower panel*) 2004. The *hatched area* indicates the anoxic part of the incubation. The March incubation lasted 44 h, but sulfide efflux to the water column was not detected; the June incubation was shorter, as sulfides migrated in the water column also under hypoxic conditions



(−82%). Induced anoxia led to a general decrease in DIC efflux (−7 to −35%) also in June, even though differences with oxic incubations were not significant (ANOVA, $p > 0.05$).

Sediments from all ponds regenerated large amounts of ammonium on both sampling dates (Fig. 5). Efflux rates measured in the oxic part of the March incubation were found between 794 (pond D) and 1,202 $\mu\text{mol m}^{-2} \text{h}^{-1}$ (pond A); differences between sampling sites were not significant. In June, the highest NH_4^+ aerobic flux was determined at pond A ($4,176 \pm 314 \mu\text{mol N m}^{-2} \text{h}^{-1}$), while at ponds B, C, and D rates were similar to those determined in March and occurred between 442 and 1,080 $\mu\text{mol m}^{-2} \text{h}^{-1}$. As anoxia established, the net

release of ammonium was significantly attenuated, with the exception of pond D, June incubation (Fig. 5). The relative decrease determined in March overlapped that just described for DIC with a minimum in pond A (−15%) and a maximum in pond D (−76%). In June, the picture was different, with a greater effect of induced anoxia in ponds A and B, where the anoxic flux was three times smaller than the corresponding oxic flux. At pond C, rates measured in anoxic conditions tended to be only somewhat smaller (−23%), while at pond D, rates were similar.

Surface sediments of ponds A, B, and C were a net sink for SRP during the aerobic phase of the March incubation, with fluxes varying between −48 and

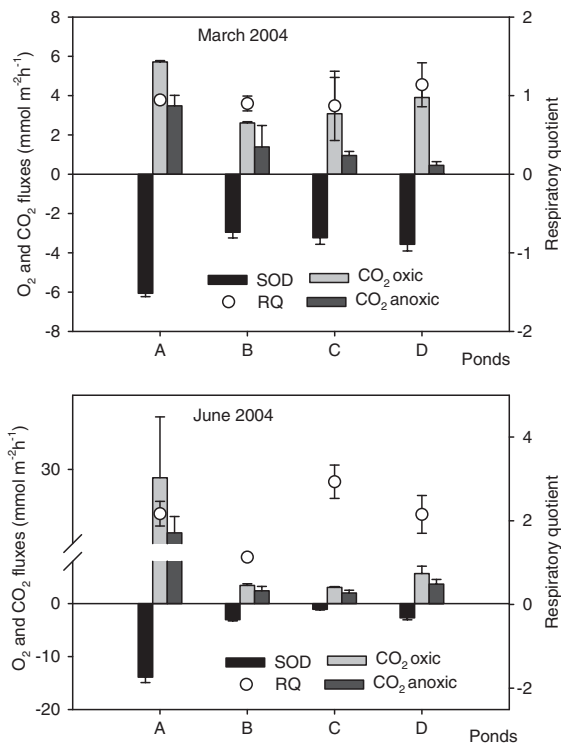
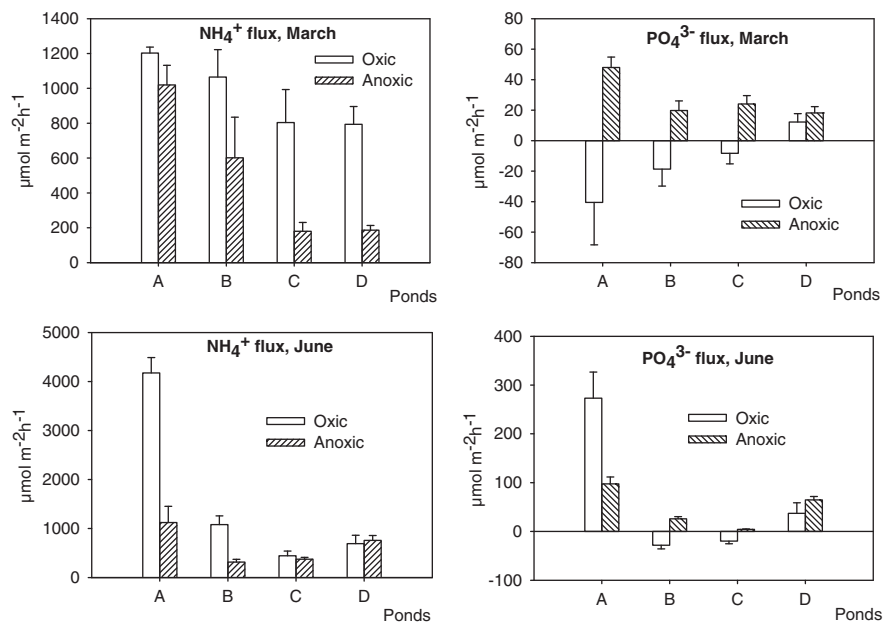


Fig. 4 Dark sediment O₂ demand (SOD), CO₂ net efflux (= DIC), and respiratory quotient (RQ = CO₂ oxic flux/O₂ flux) measured in March and June 2004 in the four phytotreatment ponds before and after the oxic–anoxic transition has occurred. Each bar is the average of three measurements; standard errors are reported

Fig. 5 Sediment–water fluxes of NH₄⁺ and PO₄³⁻ measured in March and June 2004 in the four phytotreatment ponds during the oxic and the anoxic parts of the incubation. Each bar is the average of three measurements; standard errors are reported



–8 μmol SRP m⁻² h⁻¹, while SRP regeneration was measured at pond D (12 ± 4 μmol m⁻² h⁻¹). This picture was similar during the oxic incubation of June, except for sediments from pond A that were a net source of inorganic phosphorus (273 ± 90 μmol m⁻² h⁻¹) (Fig. 5). Overall, the oxic–anoxic transition reversed SRP influx or enhanced SRP efflux to the water column; a significant effect was determined in March (ANOVA, *p* < 0.01) but not in June. In pond D where SRP flux was directed to the water column during the oxic part of the incubation, regeneration rates increased by 50% and 36% in March and June, respectively, after the transition to anoxia. In June, in pond A, SRP efflux decreased significantly (–62%), shifting from 273 ± 91 to 105 ± 31 μmol m⁻² h⁻¹.

Efflux of free sulfides in the water column was undetectable in March, even after more than 20 consecutive hours of anoxia. The situation was different in June, with marked differences among sites. In pond A, after just 1.6 h of incubation, the water column of the cores was anoxic and sulfides started migrating out of the sediments with a rate of 3,966 ± 1,293 μmol m⁻² h⁻¹ (Fig. 6); at ponds D and B, sulfides were detectable after 11 and 16 h, respectively, with rates of 1,747 ± 501 and 1,024 ± 1,040 μmol m⁻² h⁻¹. At pond C, sulfide efflux was much lower, and only few micromoles were determined in the water column during the last sampling, after 21 h of incubation (Fig. 6).

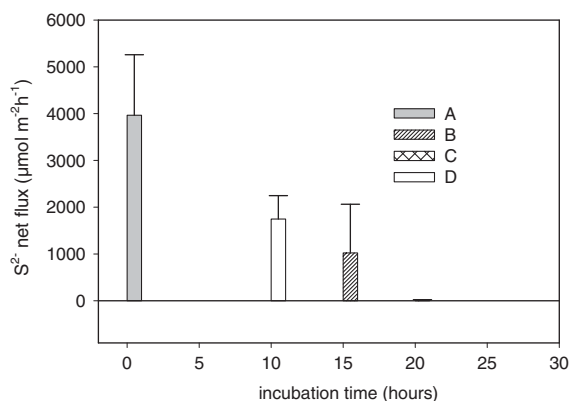


Fig. 6 Net sediment–water fluxes of free sulfides measured during the oxic–anoxic transition in the four fishponds. Rates were calculated from linear fitting of concentrations versus time, using the time at which S^{2-} was detectable in the water column at time zero

Discussion

Oxic to anoxic transition events are very common in aquatic sediments and may occur as sharp changes in environmental conditions within short spatial (mm) and temporal (h) scales. Although a number of studies have been carried out to investigate both the structure and activities of microbial populations at the oxic–anoxic interface and the long-term effects of anoxic events, little is known about the dynamic role of the OAI and the effect of short-term redox oscillations on sediment metabolism and flux rates across the interface. This is particularly true in benthic systems such as those investigated in this study that undergo large seasonal changes in sedimentary pools, the macromolecular quality of organic matter, and colonization by meio- and macrobenthos.

The phytotreatment ponds studied in this research are artificial systems that have been running for more than 10 years; their sediments host variable numbers of tolerant meio- and macrofauna species and are subject to different organic loads from winter to summer due to increased fish feed and fish biomass in the farm, which significantly alter benthic features and metabolism (Bartoli et al., 2005). Accordingly, the different effects of short-term oxic to anoxic transition on sediment metabolism and flux rates across the interface observed in the March and June incubations are a consequence of the change in sediment conditions which occurred between the two dates. If the total C, N, and P in surface sediments

were comparable in the two sampling dates, the total amount of particulate matter settling on the ponds' sediment was about three times higher in June. This input of labile organic matter resulted in significantly higher concentrations of carbohydrates and proteins that fed microbial metabolism. In turn, enhanced metabolic rates induced the reduction of electron acceptors, with a parallel redox decrease and lower sedimentary buffering capacity.

The respiratory quotient (RQ), the ratio of dark CO_2 production to oxygen uptake rates, was used as a proxy for the degree of coupling reduction and re-oxidation processes. RQ should be close to 1 under balanced conditions, that is, when all organic matter is metabolized aerobically or any reduced products of anaerobic respiration are completely re-oxidized (Andersen & Kristensen, 1988). Values substantially greater than 1 indicate incomplete re-oxidation and accumulation in the sediment of the end products of anaerobic respiration. Thus, the shift from 1 to 2.1 from March through June could indicate that the system accumulated reducing compounds. This interpretation conforms to AVS data which clearly demonstrated that there was an accumulation of reduced compounds in the June sampling, especially in ponds A and B which received the fish-farm effluent. From March to June, the AVS to AVS + CRS ratio, which is considered an index of potential risk of dystrophy (Azzoni et al., 2005), increased especially in pond A (by a factor 6) and to a less degree in ponds B (2) and C (1.6). Organic matter loads to the sediment, coupled to the onset of anaerobic metabolism, weakens or depletes the sedimentary buffering capacity against sulfides, thereby increasing the system vulnerability to dystrophic events. In March, sediments were incubated anaerobically for about 20 h, and no free sulfides accumulated in the water column, meaning that they were efficiently retained within sediments by ferrous iron (Rozan et al., 2002; Rickard & Morse, 2005 and references therein). In June, after the exhaustion of geochemical buffers, anoxia was followed by an immediate release of S^{2-} from reducing sediments of pond A, and to a lesser extent and with a longer time lag, from ponds D, B, and C.

Reducing conditions at the benthic level were responsible for the collapse of macrofauna communities (Diaz & Rosenberg, 1995; Gamelink et al., 1996; Hyland et al., 2005). Bioturbation by macrofauna has

a well-demonstrated effect on flux rates across the sediment–water interface and enhances the bacterial activity through particle reworking and transport in deep sediments of electron acceptors (Aller, 1980; Kristensen, 1988; Pelegri and Blackburn 1995; Nizzoli et al., 2007).

Experimental measurements following the induced oxic to anoxic transition within sediment cores displayed a significant increase of microbial enzymatic activity rates and bacterial production during the March experiment. A higher percentage of actively respiring cells was also observed in anoxic cores, indicating an increase of the active bacterial fraction notwithstanding the depletion of dissolved oxygen. These results are consistent with the findings of Holmer (1999) and Amtoft Neubauer et al. (2004) who reported a stimulation of sulfate reduction rates after the initial phase (6–48 h) of oxygen depletion events within marine sediments. Increased mineralization when an oxic condition changes to anoxic is also referred to as “nutrient flush” in soil studies, where the dynamic nature of OAI is known to stimulate decomposition of bacterial biomass caused by rapid environmental change (Aller, 1994).

Rapid environmental change due to redox oscillations may thus stimulate the release of labile organic substrates and promote microbial decomposition. The lowest protein-to-carbohydrate ratio observed in March after the transition reflected this event and suggested a rapid exploitation of the labile protein fraction by the microbial community. The breakdown of organic polymers by enzymatic attack contributes to the transformation of particulate organic carbon (POC) into dissolved organic carbon (DOC) which therefore becomes available for respiration (Unanue et al., 1999). In March 2004, the amount of DOC release was indirectly estimated assuming that the assimilation of $1 \text{ ng C g}^{-1} \text{ h}^{-1}$ corresponds to the processing of $4 \text{ ng C g}^{-1} \text{ h}^{-1}$ when Bacterial Growth Efficiency (BGE) is 0.2, with a BGE of 0.5 being the maximum in eutrophic waters (Del Giorgio & Cole, 1998). Assessing the total potential mobilized carbon (TPMC) from enzymatic activity measurements, we estimated that with a BGE of 0.2, $\sim 5\%$ of TPMC was converted into bacterial biomass, $\sim 20\%$ was respired, and the remaining $\sim 75\%$ was released as DOC by incomplete degradation of organic polymers. These results are in agreement with Kristensen et al. (1995), Andersen (1996), Holmer & Kristensen

(1996), and Holmer (1999), who demonstrated that under anoxic conditions, a lower CO_2 production was accompanied by an accumulation of dissolved organic carbon in pore waters.

Interestingly, the short-term increase in bacterial production following oxygen depletion was not coupled with an increase of the total mineralization rates; these, in contrast, underwent a rapid decrease as evidenced by DIC and NH_4^+ fluxes under anoxic conditions, which were about half of those measured under aerobic conditions. Such results could be explained in terms of less efficient or incomplete organic matter oxidation by the anaerobic microbial metabolism. Meio- and macrofauna can also exert an active role in the POC breakdown and are also responsible for a major fraction of solute exchange through the interface (Kristensen & Hansen, 1999; Kristensen & Holmer, 2001).

Oxygen depletion had an opposite effect on SRP dynamics, with phosphorus largely dependent on several biogeochemical factors, that is, ferric iron pool, carbonates, pore-water pH, and redox sulfur cycling (Heijs et al., 2000; Rozan et al., 2002). SRP influx during oxic incubation was the probable consequence of precipitation with ferric iron in bioturbated sediment, whereas SRP efflux under induced anoxia was a consequence of iron reduction and SRP mobilization.

The significant effect of the oxic–anoxic transition on microbial activity was not further observed during the June experiment. This was also true in terms of the percentage of actively respiring cells, which displayed similar values in both oxic and anoxic cores. Enzymatic activities showed higher rates at the start of the experiment in relation to the highest concentration of sedimentary organic matter, while bacterial production displayed lower values most probably linked to a low BGE (Del Giorgio & Cole, 1998). Nevertheless, delta values calculated as the arithmetic difference between anoxic and oxic rates in each experimental core for both enzymatic activity and bacterial production were lower when compared to those of the March experiment. Accordingly, the oxic–anoxic transition resulted in a lower reduction of NH_4^+ efflux and a not significantly different inorganic carbon production. It is likely that organic carbon oxidation in these sediments proceeds with comparable rates in aerobic and anaerobic conditions. This can be explained in terms of the lability of

settling particles (fish feces and not-ingested food together with fragments of macroalgae) and the nature of the microbial communities that are adapted to a reducing and sulfidic environment (Kristensen et al., 1995). In pond A, which received the highest load of particulate organic matter from the fish farm, DIC efflux was very high during oxic incubation ($29.7 \pm 4.2 \text{ mmol m}^{-2} \text{ h}^{-1}$), but it decreased by only 7% during the transition. Excluding results from this pond which can be biased by high solute concentrations in the closed experimental system, oxic–anoxic transition had a significant effect on SRP fluxes with the anaerobic flux higher than the corresponding aerobic fluxes.

Conclusion

These results all fit with the general hypothesis that when sediments characterized by a relatively low organic matter concentration (as is the case during the March experiment) are exposed to anoxia, the aerobic community will be damaged resulting in the sudden release of labile DOC. Under these conditions, the anaerobic community will respond positively to the DOC input. In contrast, the oxic zone in heavily loaded sediments (as is the case during the June experiment) is much narrower, which implies that oxic–anoxic transitions will only have a minor impact on the sediment biogeochemistry.

The short-term increase in microbial activity rates following induced anoxia suggests that repetitive short-term oscillations in both OAI and oxic–anoxic conditions may be a critical factor not only affecting rates of microbial mineralization but also the biological decomposition of organic polymers. The DOC which is released from enzymatic processing may accumulate within the sediment or diffuse upward into the upper sediment layers and efflux into the water column. This latter sequence of processes has been shown to fuel pelagic microbial mineralization, representing an additional important process involved in the sediment metabolism (Fabiano et al., 2003; Gallizia et al., 2004). As a general statement, the authors speculate that the dynamic nature of OAI may play a significant role in sediment biogeochemistry by stimulating microbial activities over the short term. This role may vary based on different sediment conditions and can be strongly affected by organic

matter loadings derived from direct sewage discharge and/or produced within the aquatic system as a consequence of eutrophication. Under changing conditions—for example, increased organic loads coupled with increased temperatures—the occurrence and frequency of oxic–anoxic transitions can be expected to saturate the sedimentary buffering capacity as well as to shift the benthic system toward azoic and rather unstable conditions. In other words, step-by-step the system would become more vulnerable to external stressors and less resilient until the shift would become irreversible.

Acknowledgements This research was carried out under the Contract D.R. 439 between DSA and ICRAM. We would like to thank an anonymous reviewer for helpful comments on the earlier version of this manuscript and for improving the language.

References

- Aller, R. C., 1980. Diagenetic processes near the sediment-water interface of Long Island Sound. I. Decomposition and nutrient element geochemistry (S, N, P). *Advances in Geophysics* 22: 237–350.
- Aller, R. C., 1994. Bioturbation and remineralization of sedimentary organic matter: effects of redox oscillation. *Chemical Geology* 114: 331–345.
- Amtoft Neubauer, A. T., A. G. Underlien Pedersen, K. Finster, R. A. Herbert, A. P. Donnelly, P. Viaroli, G. Giordani, R. De Wit & B. A. Lomstein, 2004. Benthic decomposition of *Zostera marina* roots: a controlled laboratory experiment. *Journal of Experimental Marine Biology and Ecology* 313: 105–124.
- Andersen, F. Ø., 1996. Fate of organic carbon added as diatom cells to oxic and anoxic marine sediment microcosms. *Marine Ecology Progress Series* 134: 225–233.
- Andersen, F. O. & E. Kristensen, 1988. The influence of macrofauna on estuarine benthic community metabolism: a microcosm study. *Marine Biology* 99: 591–603.
- APHA, 1975. *Standard Methods for the Examination of Water and Wastewaters*, 14th Edition. American Public Health Association, Washington, DC.
- Aspila, K. I., H. Aagemian & A. S. Y. Chau, 1976. A semi-automated method for the determination of inorganic, organic and total phosphate in sediments. *Analyst* 101: 187–197.
- Azzoni, R., G. Giordani & P. Viaroli, 2005. Iron-sulphur-phosphorus interactions: implications for sediment buffering capacity in a Mediterranean eutrophic lagoon (Sacca di Goro, Italy). *Hydrobiologia* 550: 131–148.
- Bartoli, M., D. Nizzoli, M. Naldi, L. Vezzulli, S. Porrello, M. Lenzi & P. Viaroli, 2005. Inorganic nitrogen control in wastewater treatment ponds from a fish farm (Orbetello, Italy): denitrification versus *Ulva* uptake. *Marine Pollution Bulletin* 50: 1386–1397.

- Battin, T. J., 1997. Assessment of fluorescein diacetate hydrolysis as a measure of total esterase activity in natural stream sediment biofilms. *Science of the Total Environment* 198: 51–60.
- Bernard, C. & T. Fenchel, 1995. Mats of colourless sulphur bacteria. II. Structure, composition of biota and successional patterns. *Marine Ecology Progress Series* 128: 171–179.
- Bhupathiraju, V. K., M. Hernandez, P. Krauter & L. Alvarez-Cohen, 1999. A new direct microscopy based method for evaluating in situ bioremediation. *Journal of Hazardous Materials* 67: 299–312.
- Bower, C. E. & T. Holm Hansen, 1986. A salicylate-hypochlorite method for determining ammonia in seawater. *Canadian Journal of Fisheries and Aquatic Science* 37: 794–798.
- Brune, A., P. Frenzel & H. Cypionka, 2000. Life at the oxic-anoxic interface: microbial activities and adaptations. *FEMS Microbiology Review* 24: 691–710.
- Canfield, D. E. & D. J. DesMarais, 1993. Biogeochemical cycles of carbon, sulphur, and free oxygen in a microbial mat. *Geochimica et Cosmochimica Acta* 57: 3971–3984.
- Castel, J., P. Caumette, & R. Herbert, 1996. Eutrophication gradients in coastal lagoons as exemplified by the bassin d'Arcachon and the étang du Prévost. *Hydrobiologia* 329: ix–xxviii.
- Cline, J. D., 1969. Spectrophotometric determination of hydrogen sulphide in natural waters. *Limnology and Oceanography* 14: 454–459.
- Crossland, C. J., H. H. Kremer, H. J. Lindeboom, J. I. Marshall Crossland, & M. D. A. Le Tissier, 2005. Coastal Fluxes in the Anthropocene. *The Land-Ocean Interactions in the Coastal Zone Project of the International Geosphere-Biosphere Programme. Global Change—The IGBP Series n XX. Springer.*
- Dalsgaard, T., L. P. Nielsen, V. Brotas, P. Viaroli, G. J. C. Underwood, D. B. Nedwell, K. Sundbäck, S. Rysgaard, A. Miles, M. Bartoli, L. Dong, D. C. O. Thornton, L. D. M. Ottosen, G. Castaldelli & N. Risgaard-Petersen, 2000. Protocol Handbook for NICE-Nitrogen Cycling in Estuaries: A Project Under the EU Research Programme. *Marine Science and Technology (MAST III). National Environmental Research Institute, Silkeborg, Denmark.*
- De Wit, R., L. J. Stal, B. A. Lomstein, R. A. Herbert, H. van Gemerden, P. Viaroli, V. U. Ceccherelli, F. Rodríguez-Valera, M. Bartoli, G. Giordani, R. Azzoni, B. Shaub, D. T. Welsh, A. Donnelly, A. Cifuentes, J. Anton, K. Finster, L. P. Nielsen, A. G. Underlien Pedersen, A. T. Neubauer, M. A. Colangelo & S. K. Heijs, 2001. ROBUST: The ROLE of Buffering capacities in STabilising coastal lagoon ecosystems. *Continental Shelf Research* 21: 2021–2041.
- Del Giorgio, P. A. & J. J. Cole, 1998. Bacterial growth efficiency in natural aquatic systems. *Annual Review of Ecology and Systematics* 29: 503–541.
- Diaz, R. J. & R. Rosenberg, 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: An Annual Review* 33: 245–303.
- Dilly, O., 2003. Regulation of the respiratory quotient of soil microbiota by availability of nutrients. *FEMS Microbiology Ecology* 43: 375–381.
- Duarte, C. M., 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41: 87–112.
- Dubois, M., K. Gilles, J. K. Hamilton, P. A. Rebers & F. Smith, 1956. Colorimetric method for determination of sugars and related substances. *Analytical Chemistry* 28: 350–356.
- Fabiano, M., D. Marrale & C. Mistic, 2003. Bacteria and organic matter dynamics during a bioremediation treatment of organic-rich harbour sediments. *Marine Pollution Bulletin* 46: 1164–1173.
- Fenchel, T., G. M. King, & T. H. Blackburn, 1998. *Bacterial Biogeochemistry: The Ecophysiology of Mineral Cycling*, 2nd Edition. Academic Press, London.
- Gallizia, I., L. Vezzulli & M. Fabiano, 2004. Oxygen supply for biostimulation of enzymatic activity in organic-rich marine ecosystems. *Soil Biology and Biochemistry* 36: 1645–1652.
- Gamelink, I., A. Jahn, K. Vopel & O. Giere, 1996. Hypoxia and sulphide as structuring factors in a macrozoobenthic community on the Baltic Sea shore: colonisation studies and tolerance experiments. *Marine Ecology Progress Series* 144: 73–85.
- Gee, J. M., R. M. W. Warwick, M. Schaanning, J. A. Berge & W. G. Ambrose Jr, 1985. Effects of organic enrichment on meiofaunal abundance and community structure in sublittoral soft sediments. *Journal of Experimental Marine Biology and Ecology* 91: 247–262.
- Hartree, E. F., 1972. Determination of proteins: a modification of the Lowry methods that give a linear photometric response. *Analytical Biochemistry* 48: 422–427.
- Heijs, S. K., R. Azzoni, G. Giordani, H. M. Jonkers, D. Nizzoli, P. Viaroli & H. Gemerden, 2000. Sulphide-induced release of phosphate from sediments of coastal lagoons and the possible relation to the disappearance of *Ruppia* sp. *Aquatic Microbial Ecology* 23: 85–95.
- Hobbie, J. E., R. J. Daley & S. Jasper, 1977. Use of Nuclepore filters for counting bacteria by fluorescence microscopy. *Applied Environmental Microbiology* 33: 1225–1228.
- Holmer, M., 1999. The effect of oxygen depletion on anaerobic organic matter degradation in marine sediments. *Estuarine, Coastal and Shelf Science* 48: 383–390.
- Holmer, M. & E. Kristensen, 1996. Seasonality of sulphate reduction and pore water solutes in a marine fish farm sediment: the importance of temperature and sedimentary organic matter. *Biogeochemistry* 32: 15–39.
- Hoppe, H. G., 1993. Significance of exoenzymatic activities in the ecology of brackish water: measurements by means of methylumbelliferyl-substrate. *Marine Ecology Progress Series* 80: 191–201.
- Hyland, J., L. W. Balthis, I. Karakassis, P. Magni, A. Petrov, J. R. Shine, O. Vestergaard & R. Warwick, 2005. Organic carbon content of sediments as an indicator of stress in the marine benthos. *Marine Ecology Progress Series* 295: 91–103.
- Jørgensen, B. B., 1996. Material flux in the sediment. In Richardson, K. & B. B. Jørgensen (eds.), *Eutrophication in Coastal Marine Ecosystems*. American Geophysical Union, Washington, DC: 115–135.
- Kristensen, E., 1988. Benthic fauna and biogeochemical processes in marine sediments: microbial activities and fluxes. In Blackburn, T. H. & J. Sørensen (eds), *Nitrogen Cycling in Coastal Marine Environments*. Wiley, New York: 276–299.

- Kristensen, E. & K. Hansen, 1999. Transport of carbon dioxide and ammonium in bioturbated (*Nereis diversicolor*) coastal, marine sediments. *Biogeochemistry* 45: 147–168.
- Kristensen, E. & M. Holmer, 2001. Decomposition of plant materials in marine sediment exposed to different electron acceptors (O_2 , NO_3^- and SO_4^{2-}), with emphasis on substrate origin, degradation kinetics and the role of bioturbation. *Geochimica et Cosmochimica Acta* 65: 419–434.
- Kristensen, E., S. I. Ahmed & A. H. Devol, 1995. Aerobic and anaerobic decomposition of organic matter in marine sediments: which is fastest? *Limnology and Oceanography* 40: 1430–1437.
- Lomstein, B., L. Bonne Guldberg, A. T. Amtoft Neubauer, J. Hansen, A. P. Donnelly, R. A. Herbert, P. Viaroli, G. Giordani, R. Azzoni, R. De Wit & K. Finster, 2006. Benthic decomposition of *Ulva lactuca*: a controlled laboratory experiment. *Aquatic Botany* 85: 271–281.
- Manini, E., C. Fiordelmondo, C. Gambi, A. Pusceddu & R. Danovaro, 2003. Benthic microbial loop functioning in coastal lagoons: a comparative approach. *Oceanologica Acta* 26: 27–38.
- Meyer-Reil, L. A., 1986. Measurement of hydrolytic activity and incorporation of dissolved organic substrate by microorganisms in marine sediments. *Marine Ecology Progress Series* 31: 143–149.
- Meyer-Reil, L. A. & M. Köster, 2000. Eutrophication of marine waters: effects on benthic microbial communities. *Marine Pollution Bulletin* 41: 255–263.
- Nizzoli, D., M. Bartoli, M. Cooper, G. Underwood & P. Viaroli, 2007. Implications for oxygen and nutrient fluxes and denitrification rates during sediment colonization by the polychaete *Nereis* spp. in four estuaries. *Estuarine, Coastal and Shelf Science* 75: 125–134.
- Pelegri, S. P. & T. H. Blackburn, 1995. Effect of bioturbation by *Nereis* sp., *Mya arenaria* and *Cerastoderma* sp. on nitrification and denitrification in estuarine sediments. *Ophelia* 42: 289–299.
- Porrello, S., M. Lenzi, E. Persia, P. Tomassetti & M. G. Finoia, 2003. Reduction of aquaculture wastewater eutrophication by phytotreatment ponds system. I. Dissolved and particulate nitrogen and phosphorus. *Aquaculture* 219: 515–529.
- Proctor, L. M. & A. C. Souza, 2001. Method for enumeration of 5-cyano-2,3-ditoyl tetrazolium chloride (CTC)-active cells and cell-specific CTC activity of benthic bacteria in riverine, estuarine and coastal sediments. *Journal of Microbiological Methods* 43: 213–222.
- Rabalais, N. N., R. E. Turner, B. K. Sen Gupta, E. Platon & M. L. Parson, 2007. Sediments tell the history of eutrophication and anoxia in the northern Gulf of Mexico. *Ecological Applications* 17: S129–S143.
- Rickard, D. & J. W. Morse, 2005. Acid Volatile Sulphide (AVS). *Marine Chemistry* 97: 141–197.
- Rozañ, T. F., M. Taillefert, R. E. Trouwborst, B. T. Glazer, S. Ma, J. Herszage, L. M. Valdes, K. S. Price & G. W. Luther III, 2002. Iron-sulphur-phosphorus cycling in the sediments of a shallow coastal bay: implications for sediment nutrient release and benthic macroalgal blooms. *Limnology and Oceanography* 47: 1346–1354.
- Schramm, W., 1999. Factors influencing seaweed responses to eutrophication: some results from EU-project EUMAC. *Journal of Applied Phycology* 11: 69–78.
- Sun, M. Y., S. G. Wakeham & C. Lee, 1997. Rates and mechanisms of fatty acid degradation in oxic and anoxic coastal marine sediments of Long Island Sound, New York, USA. *Geochimica et Cosmochimica Acta* 61: 341–355.
- Talling, J. F., 1973. The application of some electrochemical methods to the measurements of photosynthesis and respiration in fresh water. *Freshwater Biology* 3: 335–362.
- Unanue, M., B. Ayo, M. Agis, D. Slezak, G. J. Herndl & J. Iriberrí, 1999. Ecto-enzymatic activity and uptake of monomers in marine bacterioplankton described by a biphasic kinetic model. *Microbial Ecology* 37: 36–48.
- Valderrama, J. C., 1977. Methods used by the Hydrographic Department of National Board of Fisheries, Sweden. In Grasshof, K. (ed.), Report of the Baltic Intercalibration Workshop. Annex, Interim Commission for the Protection of the Environment of the Baltic Sea: 13–40.
- Valderrama, J. C., 1981. The simultaneous analysis of total nitrogen and total phosphorus in natural waters. *Marine Chemistry* 10: 109–122.
- Van Duyl, F. C. & A. J. Kop, 1994. Bacterial variation in North Sea sediments: clues to seasonal and spatial variations. *Marine Biology* 120: 323–337.
- Viaroli, P., M. Bartoli, C. Bondavalli, R. R. Christian, G. Giordani & M. Naldi, 1996. Macrophyte communities and their impact on benthic fluxes of oxygen, sulphides and nutrients in shallow eutrophic environments. *Hydrobiologia* 329: 105–119.

Denitrification efficiency for defining critical loads of carbon in shallow coastal ecosystems

Bradley D. Eyre · Angus J. P. Ferguson

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 137–146.
DOI: 10.1007/s10750-009-9765-1 © Springer Science+Business Media B.V. 2009

Abstract Denitrification efficiency [DE; $(N_2 - N / (DIN + N_2 - N) \times 100\%)$] as an indicator of change associated with nutrient over-enrichment was evaluated for 22 shallow coastal ecosystems in Australia. The rate of carbon decomposition (which can be considered a proxy for carbon loading) is an important control on the efficiency with which coastal sediments in depositional mud basins with low water column nitrate concentrations recycle nitrogen as N_2 . The relationship between DE and carbon loading is due to changes in carbon and nitrate (NO_3) supply associated with sediment biocomplexity. At the DE optimum ($500\text{--}1,000 \mu\text{mol m}^{-2} \text{h}^{-1}$), there is an overlap of aerobic and anaerobic respiration zones (caused primarily by the existence of anaerobic micro-niches within the oxic zone, and oxidized burrow structures penetrating into the anaerobic zone), which enhances

denitrification by improving both the organic carbon and nitrate supply to denitrifiers. On either side of the DE optimum zone, there is a reduction in denitrification sites as the sediment loses its three-dimensional complexity. At low organic carbon loadings, a thick oxic zone with low macrofauna biomass exists, resulting in limited anoxic sites for denitrification, and at high carbon loadings, there is a thick anoxic zone and a resultant lack of oxygen for nitrification and associated NO_3 production. We propose a trophic scheme for defining critical (sustainable) carbon loading rates and possible thresholds for shallow coastal ecosystems based on the relationship between denitrification efficiency and carbon loading for 17 of the 22 Australian coastal ecosystems. The denitrification efficiency “optimum” occurs between carbon loadings of about 50 and $100 \text{ g C m}^{-2} \text{ year}^{-1}$. Coastal managers can use this simple trophic scheme to classify the current state of their shallow coastal ecosystems and for determining what carbon loading rate is necessary to achieve any future state.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

B. D. Eyre (✉) · A. J. P. Ferguson
Centre for Coastal Biogeochemistry, Southern Cross
University, P.O. Box 157, Lismore, NSW 2480, Australia
e-mail: bradley.eyre@scu.edu.au

Keywords Carbon loading · Critical loads · Eutrophication · Nutrient enrichment · Trophic state · Indicator · Threshold

Introduction

Eutrophication associated with nutrient over-enrichment is a global problem in shallow coastal ecosystems

Present Address:

B. D. Eyre
Aquatic Biogeochemical and Ecological Research, P.O.
Box 409, Brunswick Heads, NSW 2483, Australia

(Nixon, 1995). General conceptual models are available that show the response of coastal systems to nutrient enrichment (Cloern, 2001; Kemp et al., 2005) and central to these models is the importance of benthic–pelagic coupling in shallow systems. In brief, coastal systems with small nutrient loads typically have low pelagic productivity and low light attenuation in the water column (Nielsen et al., 2002). These oligotrophic systems support a diverse range of benthic habitats and communities (e.g., seagrass, benthic microalgae) which retain nitrogen in microbial and metazoan biomass and grazing of these communities represents a flow of nitrogen up the food chain (Eyre & Ferguson, 2005). Little nitrogen is recycled from the benthic system to the water column, with most of the benthic nitrogen load lost through denitrification and/or passed up the food chain. As nutrient loads increase there is an associated increase in pelagic productivity and an increase in light attenuation in the water column (Nielsen et al., 2002). The decrease in light reaching the sediment surface results in a loss of benthic habitats (e.g., seagrass and benthic microalgae; Short & Wyllie-Echeverria, 1996) and less nitrogen flowing up the food chain. Increased organic loading from phyto-detritus leads to an increase in benthic carbon decomposition and oxygen consumption, with more nitrogen recycled to the water column as ammonium and less nitrogen lost to the atmosphere via denitrification (Kemp et al., 1990). The combination of an increased benthic flux of ammonium and less nitrogen flowing up the food chain results in a negative feedback loop with the stimulation of more pelagic production. These bottom-up changes have flow-on effects that alter the structure of higher trophic levels such as fish and invertebrate communities (Kemp et al., 2005).

To be able to effectively manage nutrient over-enrichment of coastal systems, environmental managers need criteria to define critical (sustainable) nutrient loads. Since excess nutrients disrupt the delicate balance between carbon production and metabolism in coastal systems, management criteria are probably best based on carbon loading rates rather than loadings of inorganic nutrients (Nixon, 1995). In shallow coastal systems, pelagic and benthic compartments are tightly coupled and much of the carbon production and metabolism occurs in the sediments. As such, sediments and benthic communities are probably the most sensitive part of the coastal ecosystem to nutrient enrichment, making them

ideally suited as an indicator of change and for defining management criteria such as critical (sustainable) carbon loading rates. Benthic denitrification is particularly important with respect to nutrient enrichment because it permanently removes fixed nitrogen and, as such, counteracts the processes of nitrogen over-enrichment (commonly the limiting nutrient). More important than the actual denitrification rate is the percentage of the total inorganic nitrogen released from the sediments as di-nitrogen gas (N_2), which can be expressed by the term denitrification efficiency [DE; $(N_2 - N)/(DIN + N_2 - N) \times 100\%$] (Berelson et al., 1998; Eyre & Ferguson, 2002). Because of the link among carbon loading and decomposition, oxygen consumption, and coupled nitrification–denitrification, the DE of shallow coastal systems should decrease as the carbon loading increases, making DE useful for defining critical loads (Eyre & Ferguson, 2002). The purpose of this article is to demonstrate how the DE of shallow coastal systems changes as carbon loading rates change and to use these changes in DE for defining critical loads.

Materials and methods

Data from 22 Australian coastal systems were used in this study (Table 1; Fig. 1). These systems covered a range of different trophic states from oligotrophic to hypertrophic, each of the seven major geomorphic types of Australian estuaries (Harris & Heap, 2003), as well as artificial ponds, and a range of climate zones from temperate to tropical. With the exception of Port Phillip Bay (Berelson et al., 1998), the methods used for the measurement of the benthic ΣCO_2 , dissolved inorganic nitrogen (DIN), and N_2 fluxes in all the systems were similar. Details of these methods are given in the published studies (Table 1) and similar methods were used in the unpublished studies. In brief, all fluxes were measured in plexi-glass cores (95 mm internal diameter), except for Heron Island Sands, Southern Moreton Bay Sands, and two sites in Southern Moreton Bay where chambers were used. Cores and chambers were pre-incubated for 24 h and then incubated over a 24-h dark–light cycle (only the dark data were used in this study). Typically, three samples were taken during the dark for ΣCO_2 (pH and alkalinity), DIN (flow

Table 1 Shallow coastal ecosystems used in this study and the total number of core/chamber incubations, the number of sites studied, and the frequency of study for each system

| Location | Total number of cores/chambers | Sites | Frequency | Source |
|---|--------------------------------|-------|-------------------------|--|
| Bramble Bay | 9 | 3 | Summer, Winter | Ferguson et al. (2007) |
| Bremer River | 7 | 4 | Summer | Cook et al. (2004) |
| Brunswick Estuary | 74 | 1 | Monthly for 2 years | Eyre & Ferguson (2005) |
| Cairns Prawn Effluent Pond | 6 | 2 | Summer | Unpubl. data |
| Cairns Prawn Grow-out Pond | 6 | 2 | Summer | Unpubl. data |
| Curl-Curl Lagoon | 8 | 4 | Summer, Winter | Eyre & Ferguson (2002) |
| Deception Bay | 216 | 6 | 3-Monthly for 18 months | Ferguson & Eyre (2007) |
| Dee Why Lagoon | 8 | 4 | Summer, Winter | Eyre & Ferguson (2002) |
| Edmunds Bay | 4 | 1 | Summer | Eyre & Ferguson (2002) |
| Fennell Bay | 4 | 1 | Summer | Eyre & Ferguson (2002) |
| Great Sandy Straits | 6 | 2 | Summer | Unpubl. data |
| Kooroora Bay | 4 | 1 | Summer | Eyre & Ferguson (2002) |
| Heron Island | 27 | 1 | Summer | Eyre et al. (2008); Glud et al. (2008) |
| Manly Lagoon | 8 | 4 | Summer, Winter | Eyre & Ferguson (2002) |
| Mary River | 9 | 3 | Summer | Unpubl. data |
| Mission Beach Prawn Effluent Pond | 6 | 2 | Summer | Unpubl. data |
| Mission Beach Prawn Mangrove Treatment Pond | 6 | 2 | Summer | Unpubl. data |
| Mud Pit, Moreton Bay | 9 | 3 | Summer, Winter | Ferguson et al. (2007) |
| Ord River | 15 | 5 | Summer | Unpubl. data |
| Port Phillip Bay | | | | Berelson et al. (1998) |
| Southern Moreton Bay | 40 | 8 | Summer, Winter | Ferguson et al. (2004) |
| Southern Moreton Bay Sands | 30 | 1 | Summer | Ferguson et al. (2004) |
| Waterloo Bay | 6 | 2 | Summer, Winter | Ferguson et al. (2007) |

injection analysis), and N_2 (membrane inlet mass spectrometer with oxygen removal; Eyre & Ferguson, 2002) analysis and the concentration changes over time were used to calculate flux rates. In Port Phillip Bay, N_2 was estimated using measured dissolved inorganic nitrogen and carbon efflux ratios and assuming Redfield stoichiometry, with some direct N_2 measurements (Berelson et al., 1998). Denitrification efficiency (DE) was expressed as the proportion of the total inorganic nitrogen released as N_2 ($N_2 - N / (DIN + N_2 - N) \times 100\%$) (Eyre & Ferguson, 2002). Only dark ΣCO_2 , DIN, and N_2 flux rates were used and flux rates less than zero were excluded. DE was calculated for individual cores before averaging. For individual sites and sampling times, the DE and ΣCO_2 efflux are typically the average of triplicate core or chamber incubations. Sites and

times were averaged for the DE of individual systems; the number of core or chamber incubations averaged is given in Table 1. A Loess curve was fitted to the data using SPSS v11. The Loess curve uses iterative weighted least squares to determine values that best fit the data. An Epanechinov kernel was used to smooth the data.

Results

The Loess curve fitted to the system average data indicates that the maximum DE occurred between ΣCO_2 efflux rates of about 500 and 1,500 $\mu mol m^{-2} h^{-1}$ (Fig. 2). DE then decreased rapidly to less than 10% at ΣCO_2 effluxes above about 6,000 $\mu mol m^{-2} h^{-1}$. Permeable quartz (Southern

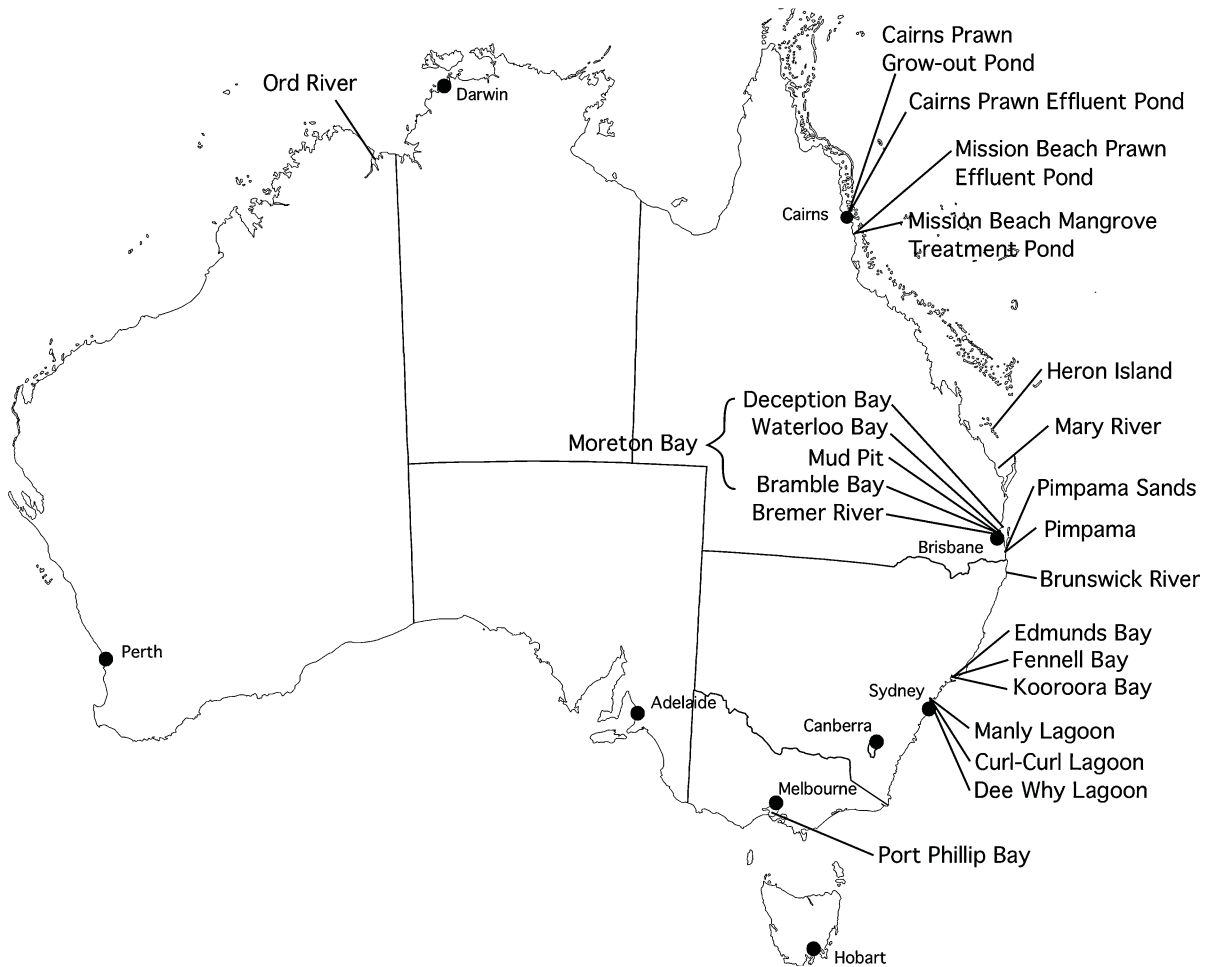


Fig. 1 Locations of the systems studied. The 22 systems studied covered a range of different trophic states from oligotrophic to hypertrophic, each of the seven major

geomorphic types of Australian estuaries, as well as artificial ponds, and a range of climate zones from temperate to tropical

Moreton Bay Sands) and carbonate sands (Heron Island), seagrass sites (Kooroora Bay, Waterloo Bay) and the Bremer River were outliers to this general relationship, with a much higher DE for a given ΣCO_2 efflux.

The Loess curve fitted to the individual site data was similar to the system average curve, showing that the maximum DE occurred between ΣCO_2 efflux rates of about 500 and 1,000 $\mu\text{mol m}^{-2} \text{h}^{-1}$ (Fig. 3). However, it also suggests that DE decreased on either side of the maximum, not just with increasing ΣCO_2 effluxes. The relationship between DE and ΣCO_2 efflux was quite variable between individual systems. In some systems, the relationship between DE and

ΣCO_2 efflux for individual sites (e.g., Port Phillip Bay) and sample times (e.g., Bramble Bay) followed the Loess curve (Fig. 3). On the other hand, in other systems, the relationship between DE and ΣCO_2 efflux for individual sites (e.g., Southern Moreton Bay) and sample times (e.g., Brunswick Estuary) showed little resemblance to the Loess curve (Fig. 3).

Discussion

Since much of the organic matter supplied to shallow coastal ecosystems is decomposed in the sediments, benthic ΣCO_2 efflux is a useful proxy for organic

Fig. 2 System average denitrification efficiency (DE) as a function of ΣCO_2 efflux for 22 shallow coastal Australia systems. A Loess curve has been fitted to the data which uses iterative weighted least squares to determine values that best fit the data. The mechanistic modeled DE versus respiration curves of Murray & Parslow (1997) and Webster (2001) are also shown. *Circled points* are outliers to the Loess curve

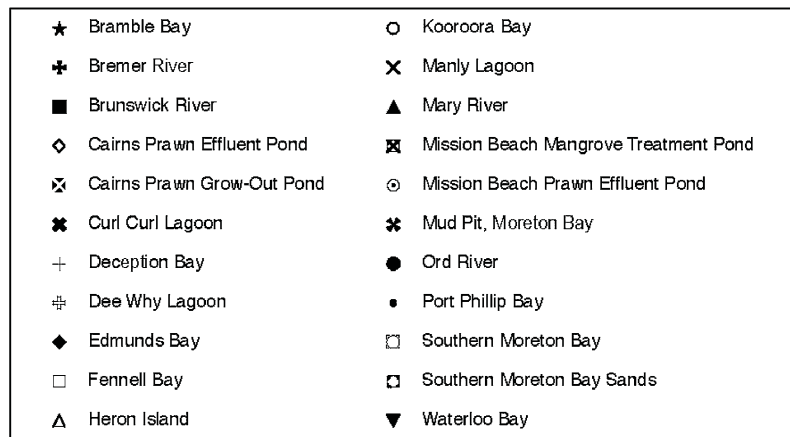
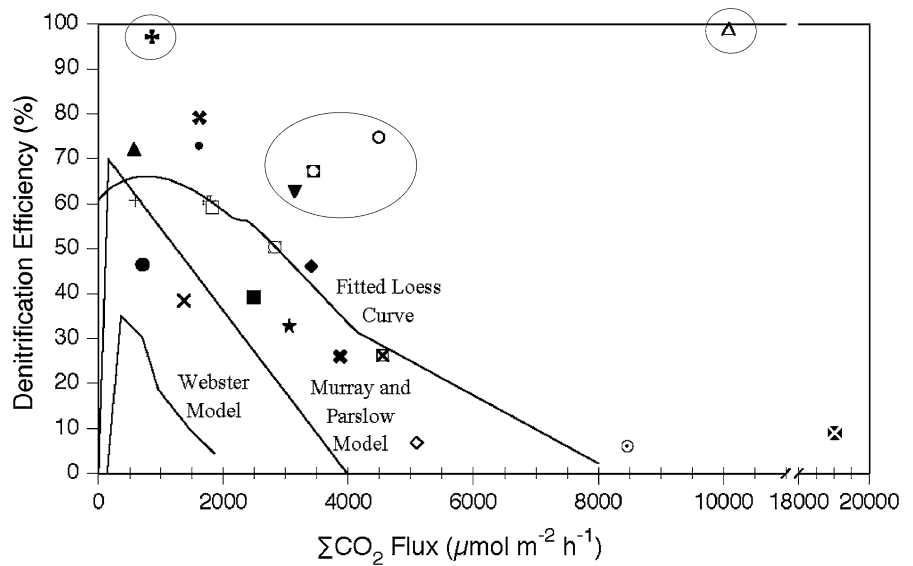
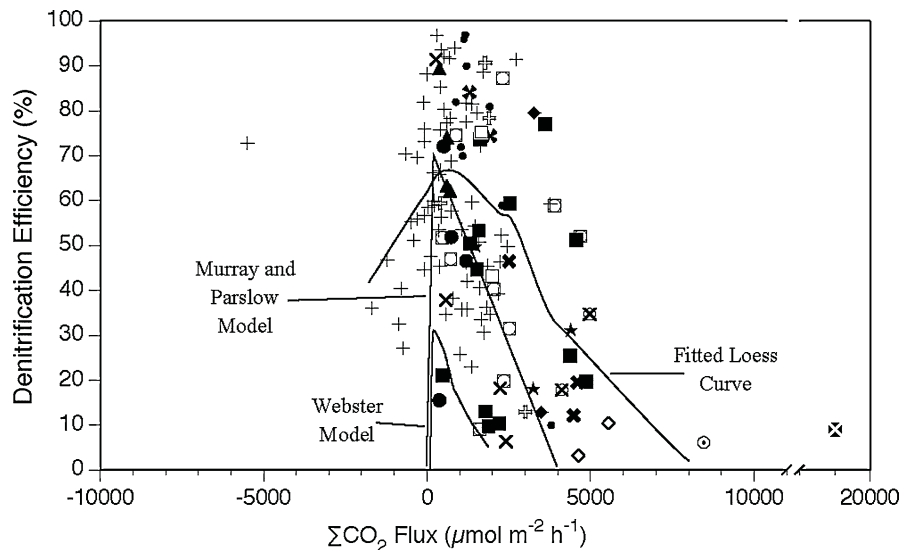


Fig. 3 Denitrification efficiency averaged for individual sites and sample times as a function of ΣCO_2 efflux for 17 shallow coastal Australia systems (outliers from Fig. 2 were excluded). A Loess curve has been fitted to the data. The mechanistic modeled DE versus respiration curves of Murray & Parslow (1997) and Webster (2001) are also shown. For symbols legend, see Fig. 2



carbon loading (both autochthonous and allochthonous), although part of the flux may be from the dissolution of CaCO_3 (see Exceptions to the Rule). In addition, benthic ΣCO_2 efflux is an easier measurement to make than quantifying both benthic and pelagic productivity and allochthonous carbon loads. The impact of carbon loading on DE in this study (i.e., above a carbon decomposition rate (threshold) of about 1,000 to 1,500 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$, there was a large decrease in DE) was similar to that in a number of previous studies. For example, in Port Phillip Bay sediments, DE decreased rapidly above a carbon decomposition rate of 1,250 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (Berelson et al., 1998), in the Bremer River sediments, above a carbon decomposition rate of 1,000 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (Cook et al., 2004), and in the sediments of warm temperate lagoons, above a carbon decomposition rate of 1,500 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (Eyre & Ferguson, 2002); all of these data sets are included in this study. Blackburn & Blackburn (1993) also noted a drop in coupled nitrification–denitrification rates at DOC loads above about 1,500 $\mu\text{mol m}^{-2} \text{ h}^{-1}$. In an experimental system with different organic loadings, the DE was highest at a carbon decomposition rate of 890 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ and decreased to near zero at a carbon decomposition rate of 1,350 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (Sloth et al., 1995). There clearly appears to be an optimum carbon loading that managers need to maintain to maximize DE in shallow coastal systems.

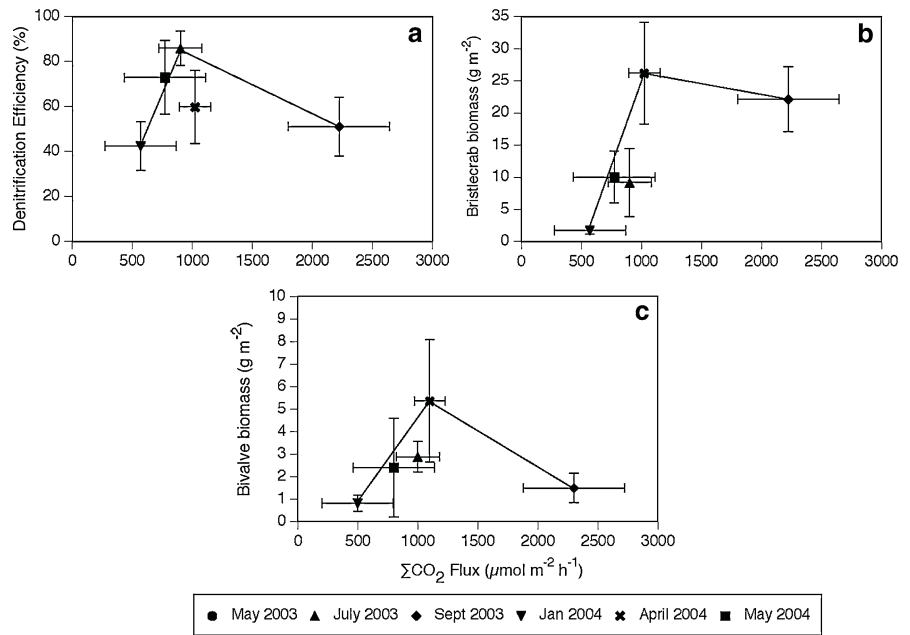
The DE versus ΣCO_2 efflux relationship of low DE at low carbon loadings, increasing to a maximum DE between a carbon loading of about 500 and 1,000 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (optimum) and then decreasing rapidly at higher carbon loadings can be explained by changes in carbon and NO_3 supply associated with sediment biocomplexity. At the DE optimum, there is probably an overlap of aerobic and anaerobic respiration zones, caused primarily by the existence of anaerobic micro-niches within the oxic zone, and oxidized burrow structures penetrating into the anaerobic zone, which enhances denitrification by improving both the organic carbon and NO_3 supply to denitrifiers (Middelburg et al., 1996; Kristensen, 2000; Ferguson & Eyre, 2007). Although there is little direct evidence of anoxic microsites, recent studies suggest that short-lived anoxic micro-niches can evolve in marine snow aggregates (Glud, 2008). On either side of the DE optimum zone (500–

1,000 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$), there is most likely a reduction in denitrification sites as the sediment loses its three-dimensional complexity. At low organic carbon loadings, a thick oxic zone (Blackburn & Blackburn, 1993) with low macrofauna biomass (Herman et al., 1999) would exist, resulting in limited anoxic sites for denitrification. In contrast, at high carbon loadings, there would be a thick anoxic zone and a resultant lack of oxygen for nitrification and associated NO_3 production (Kemp et al., 1990). The lack of coupled nitrification–denitrification sites at higher carbon loadings is exacerbated by the reduction or change in macrofauna biomass and species due to hypoxia (Gray et al., 2002) and possibly an inhibition of nitrification by exposure to sulfide (Joye & Hollibaugh, 1995). Increased organic loading has also previously been shown to decrease nitrification and denitrification in an experimental system, most likely also due to a decrease in oxygen supply and inhibition by sulfide (Caffrey et al., 1993). In addition, dissimilatory nitrate reduction to ammonium (DNRA) may become quantitatively more important as the organic matter loading increases (Kaspar et al., 1988; Gilbert et al., 1997; Christensen et al., 2000).

The influence of sediment biocomplexity on DE is further illustrated by changes in macrofauna biomass (Ferguson & Eyre, 2007; unpublished data). For example, the biomass of burrowing deposit-feeding macrofauna (e.g., bristle crabs and bivalves) in Deception Bay (Fig. 4b, c; Ferguson & Eyre, 2007; unpublished data) shows the same pattern as DE (Fig. 4a), increasing to a maximum at a carbon loading of about 1,000 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ (optimum) and then decreasing at higher carbon loadings. This results in a more two-dimensional sediment profile with less effective overlap between nitrification and denitrification zones and reduced areal extent of oxic-anoxic boundaries (Ferguson & Eyre, 2007; unpublished data).

The shape of our Loess curve of DE versus ΣCO_2 efflux (Figs. 2, 3) is consistent with the shape of simple mechanistic modeled relationships of DE versus respiration (Murray & Parslow, 1997; Webster, 2001; see Figs. 2, 3). However, the Webster model had a maximum DE of only 34% at a respiration rate of 18 $\text{mg N m}^{-2} \text{ day}^{-1}$ (ΣCO_2 efflux of 350 $\mu\text{mol m}^{-2} \text{ h}^{-1}$) and the Murray and Parslow model had a maximum DE of 70% at a respiration

Fig. 4 a Denitrification efficiency (site averages), b bristle crab biomass, and c bivalve biomass in Deception Bay as a function of ΣCO_2 efflux ($n = 6 \times 36$; Ferguson & Eyre, 2007; unpublished data)



rate of $10 \text{ mg N m}^{-2} \text{ day}^{-1}$ (ΣCO_2 efflux of $200 \mu\text{mol m}^{-2} \text{ h}^{-1}$). The maximum modeled DE occurred at lower respiration rates than our measured DE most likely because the models used net diel respiration, whereas our analysis only used dark respiration rates (Murray & Parslow, 1997). Some individual measurements of DE were also much higher than the modeled predicted maximum DE, most likely because the bioturbation and bioirrigation behavior of the macro- and meiofauna was poorly presented in the models and hence coupled nitrification–denitrification would be underestimated (Webster, 2001; also see above). Overall, the Murray and Parslow model average maximum DE of 70% compared well to the measured median DE of about 68%. The next step is to use our extensive field measurements to conceptualize, calibrate, and verify a more detailed mechanistic benthic model that more realistically includes the effects of macrofauna.

Exceptions to the rule

Despite moderate carbon decomposition rates in the Bremer River and high carbon decomposition rates in the permeable quartz sands (Southern Moreton Bay Sands), carbonate sands (Heron Island), and seagrass sites (Kooroora Bay, Waterloo Bay), they all had high DE (Fig. 2). Seagrass beds appear to be able to maintain high rates of coupled nitrification–

denitrification even under high carbon loadings (Eyre & Ferguson, 2002) probably due to the transport of oxygen down to the rhizosphere (Frederiksen & Glud, 2006). Denitrification rates as high as $412 \mu\text{mol N}_2 \text{ m}^{-2} \text{ h}^{-1}$ were measured in the seagrass beds. Similarly, high rates of porewater flushing enhance the downward transport of oxygen into permeable sands (Booij et al., 1991), and this probably enhances coupled nitrification–denitrification even under high respiration rates. In addition, some of the ΣCO_2 efflux in the carbonate sands would consist of CaCO_3 , resulting in an over-estimation of the respiration rates. In systems like the Bremer River with high water column nitrate concentrations, denitrification of water column nitrate is more important than coupled nitrification–denitrification (Cook et al., 2004) and carbon loading has little impact on DE. As such, DE as an indicator of change, and for defining critical carbon loads, is most appropriately measured in the depositional mud basins of coastal systems with low water column nitrate concentrations.

Spatial and temporal variability within an individual system

Application of the system-averaged DE versus ΣCO_2 efflux relationship to an individual system requires consideration of the controlling factors at each site and for each sample period. For example, sandy and

seagrass sites need to be excluded from any analysis (as discussed above) and some sample periods may also need to be excluded due to episodic increases in nitrate and an associated switch from coupled nitrification–denitrification to water column nitrate-driven denitrification (see Eyre & Ferguson, 2005, 2006). Some sites in depositional mud basins with low water column nitrate concentrations still have a low DE for a given carbon loading (e.g., Brunswick Estuary; Deception Bay; Fig. 3), suggesting that other factors may be limiting the nitrate supply (i.e., reduced coupled nitrification–denitrification). Possibilities include competition for DIN by benthic microalgae (Risgaard-Petersen, 2003) and bacterial assimilation of DIN during the decomposition of high C:N carbon (Eyre & Ferguson, 2005). In addition, the sites in the Brunswick Estuary with a low DE for a given carbon loading (Fig. 3) also had low respiratory quotients ($RQ = \text{dark } \Sigma\text{CO}_2 \text{ efflux/dark O}_2 \text{ consumption}$; Eyre & Ferguson, 2005), suggesting that coupled nitrification–denitrification may be inhibited due to oxygen consumption when reduced sulfides are oxidized. In the Brunswick Estuary, and other systems, low RQs typically occur in winter (see Eyre and Ferguson, 2005) under conditions of low carbon loading, suggesting that the DE versus ΣCO_2 efflux relationship may be best applied using spring and summer data when maximum carbon loadings occur.

Critical carbon loading rates

Based on the system relationship between DE and carbon loading for 17 of the 22 Australian coastal ecosystems (i.e., sands, seagrasses and Bremer River excluded; Fig. 2), the authors propose a trophic scheme for defining critical (sustainable) carbon loading rates for shallow coastal ecosystems (Fig. 5; Table 2). Oligotrophic systems have a carbon loading rate of less than about $200 \text{ g C m}^{-2} \text{ year}^{-1}$ ($2,000 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) and are defined by a median DE of 68%; although DE did start to decrease above a carbon loading of about $150 \text{ g C m}^{-2} \text{ year}^{-1}$ ($1,500 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$), the median DE is similar to the curve fitted and previously modeled maximum DE. Mesotrophic systems have carbon loading rates between about 200 and $400 \text{ g C m}^{-2} \text{ year}^{-1}$ ($2,000\text{--}4,000 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) and are defined by a median DE of 40%. Eutrophic systems have carbon loading rates between about 400

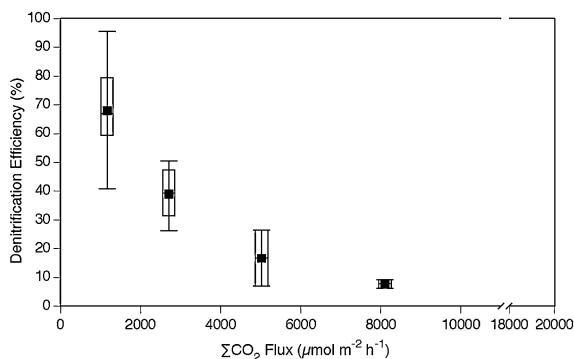


Fig. 5 Oligotrophic, mesotrophic, eutrophic, and hypertrophic box plots of system-averaged denitrification efficiencies

Table 2 A trophic scheme for shallow coastal systems based on the system relationship between denitrification efficiency (DE) and carbon loading

| Trophic type | Organic carbon loading ($\text{g C m}^{-2} \text{ year}^{-1}$) | Median denitrification efficiency (%) |
|--------------|---|--|
| Oligotrophic | <200 | 68 |
| Mesotrophic | 200–400 | 40 |
| Eutrophic | 400–600 | 18 |
| Hypertrophic | >600 | 8 |

and $600 \text{ g C m}^{-2} \text{ year}^{-1}$ ($4,000\text{--}6,000 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) with the median DE decreasing to 18%. Hypertrophic systems have carbon loading rates in excess of $600 \text{ g C m}^{-2} \text{ year}^{-1}$ ($6,000 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ h}^{-1}$) resulting in a very low median DE (8%). The cutoffs were arbitrarily chosen to give a systematic increase in the carbon loading ranges and a systematic decrease in DE. The cutoff for each trophic state is higher than those proposed by Nixon (1995), but Nixon's cutoffs were only based on pelagic production data and did not include benthic production and allochthonous sources.

Coastal managers can use this simple trophic scheme to classify the current state of their shallow coastal ecosystems and to determine what carbon loading rate is necessary to achieve any future state. Benthic ΣCO_2 fluxes can be used as a reasonably simple integrated measure of organic carbon loading rates (both autochthonous and allochthonous). More importantly, because the DE relationship is exponential, it suggests that managers will be rewarded with greater return (i.e., increased denitrification for natural removal of nitrogen) for effort (i.e., reducing

carbon loading rates) in the early stages of eutrophication (Eyre & Ferguson, 2002). We are currently measuring DE and ΣCO_2 effluxes in a coastal system where wastewater effluent has been removed to determine if there is any hysteresis in the DE/carbon loading relationship.

Acknowledgements We thank the many people (too many to name) who helped with the field work and Iain Alexander and Lyndon Brooks for assistance with the data analysis. The manuscript was improved by the comments of two anonymous reviewers. This work was funded by two ARC Discovery Grants (A00000938; DP0342956) and an ARC Linkage Grant (LP0212075).

References

- Berelson, W. M., D. Heggie, A. Longmore, T. Kilgore, G. Nickolson & G. Skyring, 1998. Benthic nutrient recycling in Port Phillip Bay, Australia. *Estuarine, Coastal and Shelf Science* 46: 917–934.
- Blackburn, T. H. & N. D. Blackburn, 1993. Rates of microbial processes in sediments. *Philosophical Transactions of the Royal Society of London* 344: 49–58.
- Booij, K., W. Helder & B. Sundby, 1991. Rapid redistribution of oxygen in a sandy sediment induced by changes in the flow velocity of the overlying water. *Netherlands Journal of Sea Research* 28: 149–165.
- Caffrey, J. M., N. P. Sloth, H. F. Kasper & T. H. Blackburn, 1993. Effect of organic loading on nitrification and denitrification in a marine sediment microcosm. *FEMS Microbiology and Ecology* 12: 159–167.
- Christensen, P. B., S. Rysgaard, N. P. Sloth, T. Dalsgaard & S. Schwaerter, 2000. Sediment mineralisation, nutrient fluxes, denitrification and dissimilatory nitrate reduction to ammonium in an estuarine fjord with sea cage trout farms. *Aquatic Microbiology Ecology* 21: 73–84.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Cook, P. L. M., B. D. Eyre, R. Leeming & E. C. V. Butler, 2004. Benthic fluxes of nitrogen in the tidal reaches of a turbid, high nitrate subtropical river. *Estuarine, Coastal and Shelf Science* 59: 675–685.
- Eyre, B. D. & A. J. P. Ferguson, 2002. Comparison of carbon production and decomposition, benthic nutrient fluxes and denitrification in seagrass, phytoplankton, benthic microalgal and macroalgal dominated warm temperate Australian lagoons. *Marine Ecology Progress Series* 229: 43–59.
- Eyre, B. D. & A. J. P. Ferguson, 2005. Benthic metabolism and nitrogen cycling in a sub-tropical east Australian estuary (Brunswick): temporal variability and controlling factors. *Limnology and Oceanography* 50: 81–96.
- Eyre, B. D. & A. J. P. Ferguson, 2006. Impact of a flood event on benthic and pelagic coupling in a sub-tropical east Australian estuary (Brunswick). *Estuarine, Coastal and Shelf Science* 66: 111–122.
- Eyre, B. D., R. N. Glud & N. Pattern, 2008. Coral mass spawning—a natural large-scale nutrient enrichment experiment. *Limnology and Oceanography* 53: 997–1013.
- Ferguson, A. J. P. & B. D. Eyre, 2007. Seasonal discrepancies in denitrification measured by isotope pairing and N_2 : Ar techniques. *Marine Ecology Progress Series* 350: 19–27.
- Ferguson, A. J. P., B. D. Eyre, A. Webb & D. Maher, 2004. Sediment biogeochemistry. In *Pimpama River Estuary Ecological Study: Chapter 11. Report Prepared for Gold Coast Water, SKM, Brisbane*.
- Ferguson, A. J. P., B. D. Eyre, J. Gay, N. Emtage & L. Brooks, 2007. Benthic metabolism and nitrogen cycling in a subtropical embayment: spatial and seasonal variability and controlling factors. *Aquatic Microbiology Ecology* 48: 175–195.
- Frederiksen, M. S. & R. N. Glud, 2006. Oxygen dynamics in the rhizosphere of *Zostera marina*: a two dimensional planar optode study. *Limnology and Oceanography* 51: 1072–1083.
- Gilbert, F., P. Souchu, M. Bianchi & P. Bonin, 1997. Influence of shellfish farming activities on nitrification, nitrate reduction to ammonium and denitrification at the water-sediment interface of the Thau lagoon, France. *Marine Ecology Progress Series* 151: 143–153.
- Glud, R. N., 2008. Oxygen dynamics of marine sediments. *Marine Biology Research* 4: 243–289.
- Glud, R. N., B. D. Eyre & N. Pattern, 2008. Biogeochemical responses to coral mass-spawning on the Great Barrier Reef: effects on respiration and primary production. *Limnology and Oceanography* 53: 1014–1024.
- Gray, J. S., R. S. Wu & Y. Y. Or, 2002. Effects of hypoxia and organic enrichment on the coastal marine environment. *Marine Ecology Progress Series* 238: 249–279.
- Harris, P. T. & A. D. Heap, 2003. Environmental management of clastic coastal depositional environments: inferences from an Australian geomorphic database. *Ocean and Coastal Management* 46: 457–478.
- Herman, P. M. J., J. J. Middleburg, J. Van De Koppel & C. H. R. Heip, 1999. Ecology and estuarine macrobenthos. *Advances in Ecological Research* 29: 195–240.
- Joye, S. B. & J. T. Hollibaugh, 1995. Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments. *Science* 270: 623–625.
- Kaspar, H. F., G. H. Hall & A. J. Holland, 1988. Effects of sea cage salmon farming on sediment nitrification and dissimilatory nitrate reduction. *Aquaculture* 70: 333–334.
- Kemp, W. M., P. Sampou, J. Caffery & M. Mayer, 1990. Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnology and Oceanography* 35: 1545–1563.
- Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M. Glibert, J. D. Hagy, L. M. Harding, E. D. Houde, D. G. Kimmel, W. D. Miller, R. I. E. Newell, M. R. Roman, E. M. Smith & J. C. Stevenson, 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303: 1–29.
- Kristensen, E., 2000. Organic matter diagenesis at the oxic/anoxic interface in coastal marine sediments, with emphasis on the role of burrowing animals. *Hydrobiologia* 426: 1–24.

- Middelburg, J. J., K. Soetaert & P. M. J. Herman, 1996. Evaluation of the nitrogen isotope-pairing method for measuring benthic denitrification: a simulation analysis. *Limnology and Oceanography* 41: 1839–1844.
- Murray, A. & J. Parslow, 1997. Port Phillip Bay Integrated Model: Final Report. Port Phillip Bay Environmental Study Technical Report No. 44, CSIRO, Canberra.
- Nielsen, S. L., K. Sand-Jensen, J. Borum & O. Geertz-Hansen, 2002. Phytoplankton, nutrients and transparency in Danish coastal waters. *Estuaries* 25: 930–937.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Risgaard-Petersen, N., 2003. Coupled- nitrification-denitrification in autotrophic and heterotrophic estuarine sediments: on the influence of benthic microalgae. *Limnology and Oceanography* 48: 93–105.
- Short, F. T. & S. Wyllie-Echeverria, 1996. Natural and human-induced disturbance of seagrass. *Environmental Conservation* 23: 17–27.
- Sloth, N. P., H. Blackburn, L. S. Hansen, N. Risgaard-Petersen & B. A. Lomstein, 1995. Nitrogen cycling in sediment with different organic loading. *Marine Ecology Progress Series* 116: 163–170.
- Webster, I., 2001. Modelling denitrification in aquatic sediments. In Hart, B. T. & M. R. Grace (eds), *Nitrogen Workshop 2000: Sources, Transformations, Effects and Management of Nitrogen in Freshwater Ecosystems*. Land and Water Australia Occasional Paper 10/00: 109–113.

Responses of benthic–pelagic coupling to climate change in a temperate estuary

Robinson W. Fulweiler · Scott W. Nixon

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 147–156.
DOI: 10.1007/s10750-009-9766-0 © Springer Science+Business Media B.V. 2009

Abstract This article reports the first demonstration of the impact of climate change on benthic–pelagic coupling and the biogeochemical cycles of a coastal marine system. Over the last 30 years Narragansett Bay, a 328-km² temperate estuary on the east coast of the United States, has undergone a variety of ecological changes. Building on a robust data set that spans three decades, we present a link between warming (+1.7°C in annual mean water temperature) in the bay and a marked decrease in sediment oxygen consumption, in the fluxes of ammonium and phosphate from sediments to the overlying water, and in sediment denitrification. We attribute this reduction in biogeochemical exchange to a dramatic drop in the standing crop of water-column chlorophyll as the system has shifted from one characterized by a dominant winter–spring bloom to one supported by

more ephemeral and less intense summer–autumn blooms. The recent climate-induced oligotrophication of the bay will be further exacerbated by forthcoming nitrogen reductions due to tertiary sewage treatment.

Keywords Climate change · Benthic–pelagic coupling · Oligotrophication · Denitrification · Nutrient regeneration · Phenology

Introduction

Even if the climate change—eutrophication linkage is well established for the pelagic system...it remains poorly defined for benthic–pelagic coupling and the benthos.

Grall & Chauvaud (2002)

Coastal ecosystems around the world have been subjected to a variety of locally driven anthropogenic changes (e.g., nutrient enrichment: Valiela et al., 1992; Kinney & Roman, 1998; eutrophication: de Jonge et al., 1994; overfishing: Jackson et al., 2001; toxic pollutant loading: Kot-Wasik et al., 2004). Further complicating these local issues are regional and global forces, most notably climate change. Obvious possible effects of climate change on coastal systems include sea level rise, changes in precipitation patterns, and the associated pulsing of freshwater, nutrients, and sediment (Scavia et al., 2002), and perhaps increased

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

R. W. Fulweiler (✉)
Department of Earth Sciences, Boston University, Boston, MA 02215, USA
e-mail: rwf@bu.edu

S. W. Nixon
Graduate School of Oceanography, University of Rhode Island, Narragansett, RI 02882, USA
e-mail: swn@gso.uri.edu

intensity of storms (Emanuel, 2005). Climate change may also affect coastal systems in more subtle ways by altering the phenology or sequencing of events (Farnsworth et al., 1995; Inouye et al., 2000; Walther et al., 2002; Edwards & Richardson, 2004). To date, the interactions between climate change, sediment biogeochemical cycling, and benthic–pelagic coupling have largely been ignored.

Over the last decade, various ecological changes in Narragansett Bay (Rhode Island, USA) have been reported, including a decrease in mean annual water column chlorophyll concentrations (Li & Smayda, 1998), replacement of boreal demersal fish with demersal decapods (Oviatt, 2004), and earlier development and larger populations of the major predator zooplankton, *Mnemiopsis leidyi* (Sullivan et al., 2001). These changes are thought to be mainly the result of climate variability (North Atlantic Oscillation) and a long-term warming trend (Nixon et al., 2003, 2004; Oviatt, 2004). There is a long history of work documenting the strong linkages between the benthos and the water column in Narragansett Bay, including benthic oxygen uptake and nutrient regeneration (Nixon et al., 1976, 1980), denitrification (Seitzinger et al., 1984; Nowicki & Oviatt, 1990), benthic responses to organic additions of ^{15}N -labeled (Garber, 1982) and ^{14}C -labeled (O'Reilly, 1984; Rudnick & Oviatt, 1986) and unlabeled (Kelly & Nixon, 1984; Grassle et al., 1985) organic matter, and predation pressure by the benthos on the water column (Doering et al., 1986; Doering, 1989; Oviatt, 2004). Building upon this study, we recently revisited the most well-studied site in the bay to measure benthic oxygen uptake, nutrient regeneration, and sediment denitrification over an annual cycle. Our main motivation for this study was to see whether it would be possible to document any changes in the biogeochemical cycling of Narragansett Bay benthos over the last 30 years. Also, the major wastewater treatment plants discharging to the bay are moving to tertiary treatment. This research will provide baseline data for future studies of denitrification and benthic fluxes after tertiary wastewater treatment is implemented. While this management intervention may improve the water quality in the upper bay, the consequences for the much larger mid- and lower bay are unknown.

The purpose of this article is twofold. The main goal is to report, we believe for the first time, a link

between climate-induced changes in benthic–pelagic coupling and sediment biogeochemical cycles. Second, to highlight the possible interactions between climate change and management mitigation of nutrient enrichment.

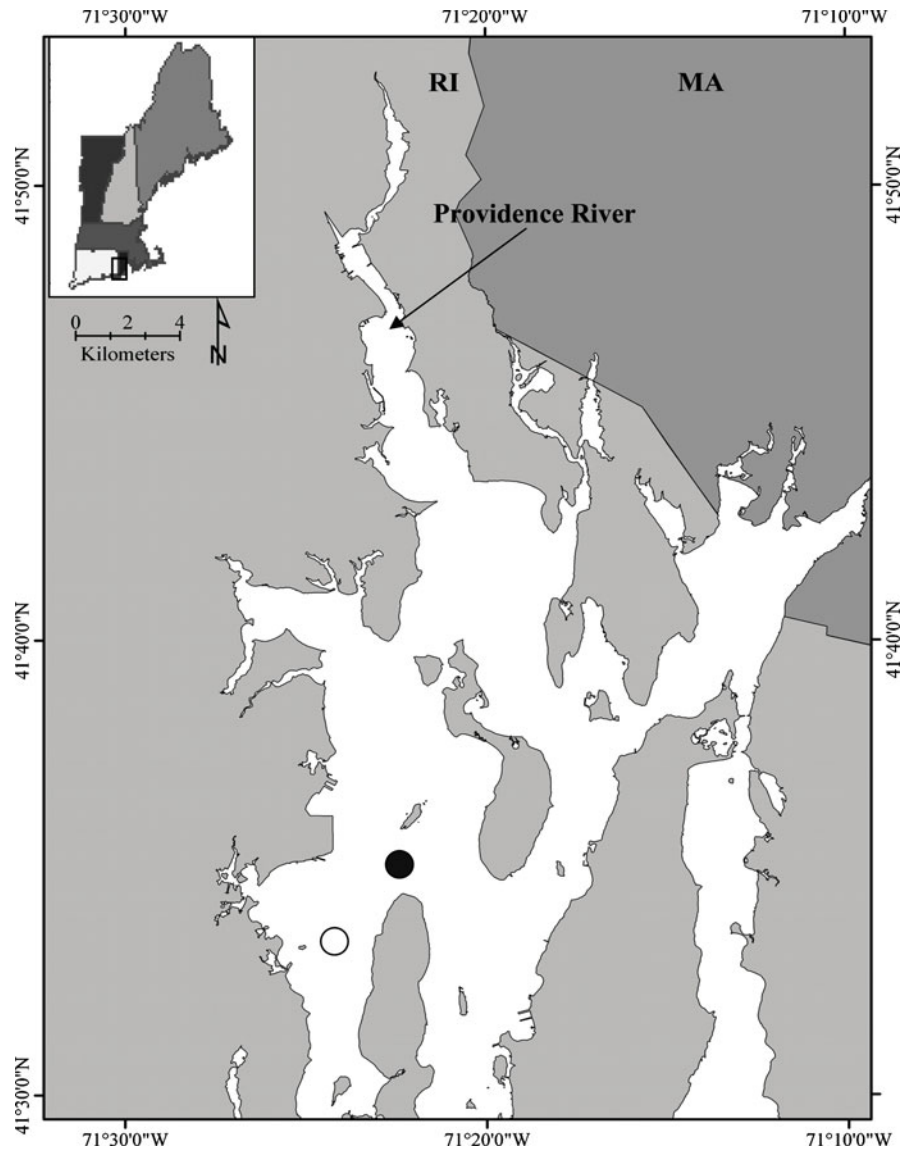
Materials and methods

Site description

Narragansett Bay, including Mount Hope Bay, is a 328-km² phytoplankton-based temperate ecosystem (latitude 41°N) with a mean depth of 8.6 m and a mean flushing rate of 26 days (Pilson, 1985; Nixon et al., 1995). Freshwater input is relatively low (100 m³ s⁻¹), with the result that the mid-bay is generally well mixed and there is only occasional weak vertical stratification (Nixon et al., 2005). Salinity follows a down-bay gradient from ~20 psu at the head to ~32 psu at the mouth of Narragansett Bay. The annual temperature varies from about 0 to 24°C.

Clayey silt and sand-silt-clay sediments comprise the majority of Narragansett Bay sediments (McMaster, 1960). For this study, sediment cores were collected from a station ~7 m deep near the middle of Narragansett Bay (Fig. 1). With an annual mean vertical light attenuation coefficient of -0.55 m^{-1} (Oviatt et al., 2002), only about 2% of surface light reaches the heterotrophic bottom. This mid-bay station has been the site of many studies ranging from in situ experiments on sediment oxygen demand and nutrient regeneration (Nixon et al., 1976) to denitrification measurements using extracted cores (Seitzinger et al., 1984) and larger mesocosms (Nowicki, 1994). Sediment from this site has also been used for various experiments (i.e., assessments of the benthic community (Rudnick et al., 1985; Frithsen, 1989; Ellis, 2002); effects of organic enrichment on the benthos (Garber, 1982; Grassle et al., 1985); benthic predation on the water column (Doering et al., 1986; Oviatt, 2004) in the large mesocosms of the marine ecosystems research laboratory (MERL). Previous study has shown that sediments at this station are predominantly silt-clay (73%) with an organic content of 4.5% (Hale, 1974). At this mid-bay station, the dominant macrofauna include *Mediomastus ambiseta*, *Nephtys incisa*, *Amplisca*, and *Nucula annulata* (Seitzinger et al., 1984;

Fig. 1 Location of the long-term sediment sampling site (*closed circle*) in mid-Narragansett Bay and the location of the long-term water quality site (*open circle*)



Rudnick et al., 1985). This site has been and is currently being monitored for benthic macro infauna and meiofauna species composition and abundance as part of the Environmental Protection Agency northeast coastal assessment program (C.A. Oviatt pers. comm., see <http://www.gso.uri.edu/merl/data.htm>).

Sediment collection and incubation

Over the annual cycle in 2005 and 2006, triplicate cores (78.5 cm² and ~15 cm deep) were collected by SCUBA divers using a round PVC corer. Cores were

maintained in the field at in situ temperatures and transported to an environmental chamber at the Graduate School of Oceanography where they were held in the dark in a water bath at field temperature. The cores were then left uncapped with oxygen bubbling gently through the overlying water overnight (8–12 h).

Throughout this study, the benthic flux measurements were conducted in the dark and separated into two incubations. During the first incubation, water samples were collected for dissolved gas analysis to measure denitrification using the N₂/Ar technique

(Kana et al., 1994; Giblin et al., 1995; Eyre et al., 2002). Then, after a 24-h period of no measurements, the second incubation for nutrient fluxes (NH_4^+ , PO_4^-) was initiated. Sediment oxygen consumption was measured during both incubations. Before each set of flux measurements, the overlying water in the cores was carefully replaced with filtered (1 μm) Narragansett Bay water. Cores were then sealed with gas-tight lids containing magnetic stirrers and the overlying water was continuously stirred (40 rpm). Initial and final samples for Winkler analysis were collected from each core and oxygen was monitored over the course of the incubation using a Hach HQ10 LDO probe. N_2/Ar samples were collected in duplicate at five times during the incubation and preserved with HgCl_2 . Dissolved inorganic nitrogen and phosphorus samples were collected at three points during the second incubation and filtered through glass fiber filters (Whatman GF/F 0.70 μm) using a 60-ml acid-washed polypropylene syringe. The filtrate was collected and stored in 60-ml acid-washed and deionized water-leached polyethylene bottles and stored at -15°C until analysis. An additional core container with no sediments was incubated as a control. When necessary, corrections were made for any changes that occurred, but the water column respiration was usually negligible.

Analytical methods

N_2/Ar was measured using MIMS (Kana et al., 1994; Giblin et al., 1995) including a Pfeiffer Quadrupole Mass Spectrometer and a flow-through silicone capillary membrane inlet (Bay Instruments, Easton, Maryland). Dissolved inorganic nitrogen and phosphorus concentrations were analyzed using a Lachat Instrument QuikChem 8000 flow injection analyzer (Table 1). Oxygen concentrations were determined using the Winkler method with manual titration.

Data analysis

The N_2 flux for each core was determined by the rate of change in N_2 concentration (Groffman et al., 2006). For the N_2/Ar method, the change in N_2 concentration is determined from the change in N_2/Ar multiplied by the Ar concentration at air saturation (Colt, 1984). N_2 production was then determined from a five-point linear regression (Giblin et al., 1995). Sediment oxygen consumption was determined by subtracting the final concentration from the initial concentration and dividing by the total time of the incubation. Rates were then prorated for the volume and area of the core. Nutrient fluxes were initially calculated as above for oxygen. However, upon further examination, it was noticed that fluxes decreased over time, most likely because of a decrease in the concentration gradient between the sediments and the overlying water. To avoid underestimating the nutrient fluxes, the flux between the initial and middle time points of the incubation was calculated. Differences between the historical and most recently collected data and their relationship with temperature were examined using a two-way ANOVA and least-squared difference (LSD) multiple comparison test.

Results

Oxygen uptake and nutrient fluxes

Sediment oxygen consumption over the annual cycle ranged from a low of 3 $\text{mg m}^{-2} \text{h}^{-1}$ at 6°C to a high of 27 $\text{mg m}^{-2} \text{h}^{-1}$ at 23°C (Fig. 2). The low-temperature measurements (4 $\text{mg m}^{-2} \text{h}^{-1}$ at 5°C) in this study were similar to the historical data, but the annual mean in the past (43 $\text{mg m}^{-2} \text{h}^{-1}$) was much higher than what was measured here (14 $\text{mg m}^{-2} \text{h}^{-1}$). Overall, the oxygen consumption measurements in this study showed no significant ($P < 0.01$) relationship with

Table 1 Analytical methods used in this study

| Parameter | Method reference | Detection limit |
|------------------------------|--|---------------------------------|
| N_2/Ar ratio | Kana et al., 1994 | 0.03% |
| Ammonium | US EPA, 1983a; Grasshoff, 1976 | 0.07 μM |
| Nitrite + Nitrate | US EPA, 1997; Grasshoff, 1976 | 0.02 μM |
| Orthophosphate | US EPA, 1983b; Grasshoff, 1976 | 0.01 μM |
| Oxygen | Lambert & Oviatt, 1983; Carrit & Carpenter, 1966 | 0 $\text{mg O}_2 \text{l}^{-1}$ |

Fig. 2 Sediment oxygen uptake over the annual temperature cycle for mid-Narragansett Bay sediments. Historical data (*closed circles*) from Nixon et al. (1976, 1980) and unpublished data. Most recent 2005/2006 data (*open circles*) from this study

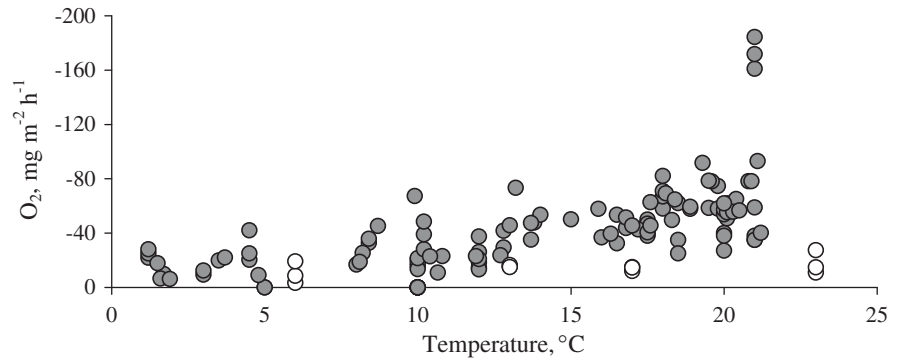


Fig. 3 Ammonia fluxes over the annual temperature cycle for mid-Narragansett Bay sediments. Historical data (*closed circles*) from Nixon et al. (1976, 1980) and unpublished data. Most recent 2005/2006 data (*open circles*) from this study

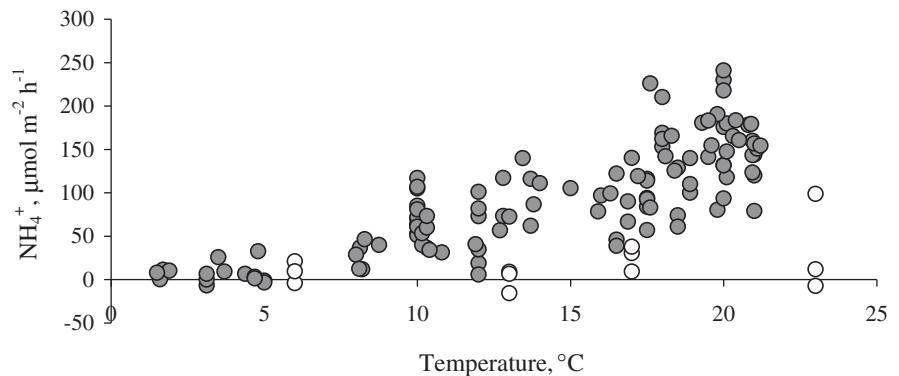
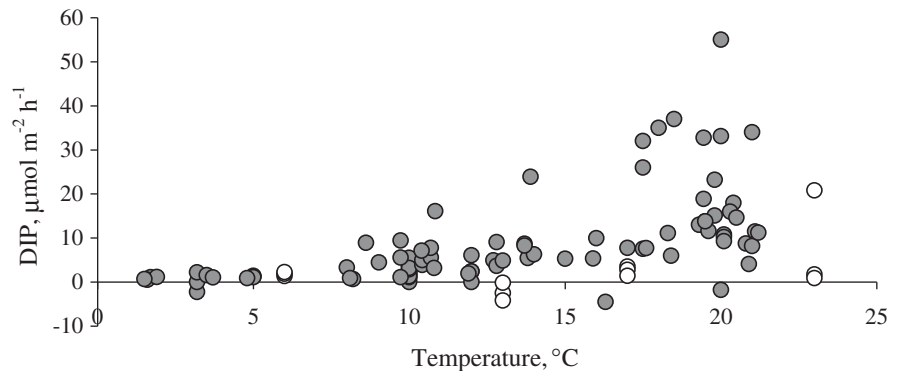


Fig. 4 Dissolved inorganic phosphate (DIP) fluxes over the annual temperature cycle for mid-Narragansett Bay sediments. Historical data (*closed circles*) from Nixon et al. (1976, 1980) and unpublished data. Most recent 2005/2006 data (*open circles*) from this study

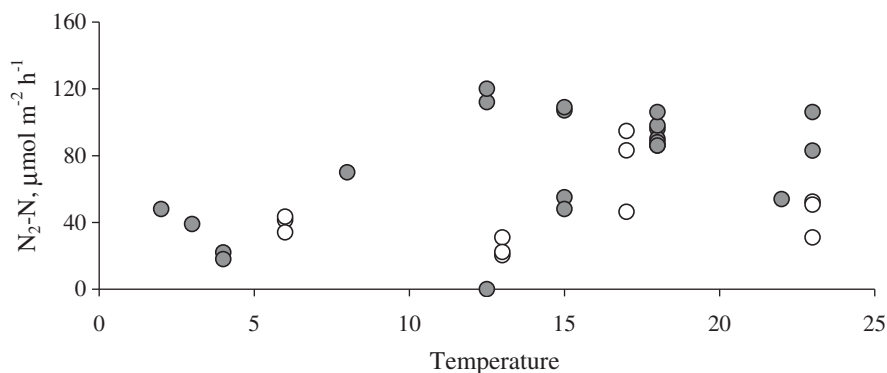


temperature. This is surprising since all of the historical data sets for mid-Narragansett Bay, including measurements made in situ, in extracted cores, and in the large (13 m^3) MERL mesocosms (Oviatt et al., 1993) showed an exponential relationship between oxygen uptake and temperature (Nixon et al., 1976). In the most recent measurements, the rates were also significantly ($P < 0.01$) lower than the historical rates.

The measurements of ammonium flux over the annual cycle in this study also showed small uptakes and releases (Fig. 3). With an influx of $-15 \mu\text{mol}$

$\text{m}^{-2} \text{h}^{-1}$ and an efflux of $98 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$, the mean ammonium flux ($14 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$) was significantly ($P < 0.01$) lower than the historical mean ($94 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$) (Nixon et al., 1980). Previously, ammonium regeneration has exhibited a marked increase with higher temperatures, but this no longer appears to be true. Dissolved inorganic phosphate (DIP) flux ranged from -4 to $21 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$ with a mean of $2.5 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$ (Fig. 4). This range is significantly ($P < 0.01$) lower than the historical range (-5 to $55 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$) and mean ($8 \mu\text{mol} \text{m}^{-2} \text{h}^{-1}$).

Fig. 5 Denitrification over the annual temperature cycle for mid-Narragansett Bay sediments. Historical data (*closed circles*) from Seitzinger et al. (1984) and Nowicki (1994). Most recent 2005/2006 data (*open circles*) from this study



Remarkably, as seen with oxygen and ammonia, DIP fluxes also showed no increase with temperature as they had in the past.

Denitrification rates

Using the N₂/Ar technique, we found denitrification rates ranging from 22 to 100 μmol N₂-N m⁻² h⁻¹. Seitzinger et al. (1984) reported seasonal denitrification rates measured for 1979 at the mid-bay station that ranged from 40 to 109 μmol N₂-N m⁻² h⁻¹. Additional denitrification measurements made by Nowicki (1994) using sediment from this site and a slightly modified technique varied from 18 to 120 μmol N₂-N m⁻² h⁻¹. While the rates in this study are similar to those measured previously, the mean annual rate has significantly ($P < 0.01$) declined from 74 μmol N₂-N m⁻² h⁻¹ in 1979 to 40 μmol N₂-N m⁻² h⁻¹ in 2005/2006. Neither the present nor the historic denitrification rates exhibit a positive relationship with increasing temperature (Fig. 5).

Discussion

The analytical techniques used to measure the concentrations of oxygen, ammonia, and phosphate have not changed substantially between the earlier work and the current study and therefore analytical changes do not confound these comparisons. Unfortunately, it is possible that the denitrification decline may be an artifact of changing methodology. No direct comparisons between the N₂ technique, used by Seitzinger et al. (1984) and Nowicki (1994), and the N₂/Ar technique have yet been reported. However, the oxygen and nutrient regeneration measurements

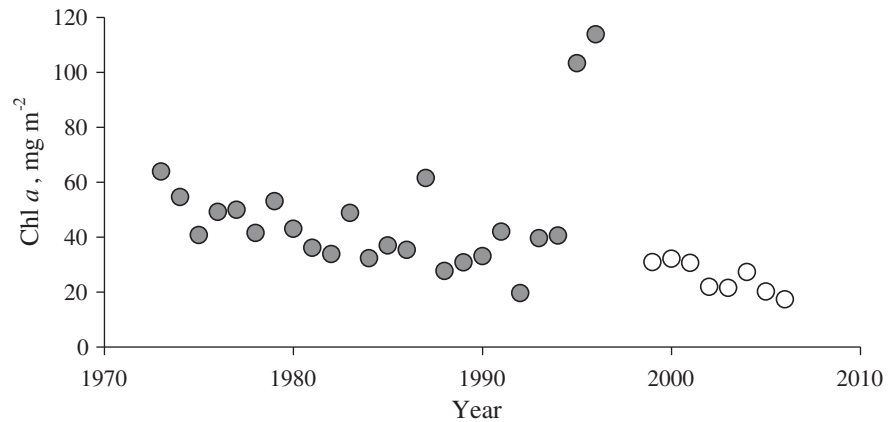
reported previously are not confounded by changing analytical techniques, and their large declines suggest that the decline in denitrification is also real.

Ecological responses to climate change in Narragansett Bay

Narragansett Bay has traditionally been characterized by a strong winter–spring diatom bloom that usually begins in mid-November and reaches peaked concentrations at the end of December or early January (Pratt, 1959). However, over the last three decades the timing and magnitude of the phytoplankton bloom in Narragansett Bay have decreased dramatically. Long-term monitoring of the phytoplankton at a station near the middle of the bay (Fig. 1) shows that mean annual water-column chlorophyll concentrations have decreased by almost 70% since 1970 (Fig. 6). Most of the decline in the annual means is associated with changes in the traditional winter–spring bloom which has arrived later or not at all (Oviatt et al., 2002). The abundance of diatoms during the bloom has also declined (Karentz & Smayda, 1998; Li & Smayda, 1998). Since chlorophyll is tightly coupled with the rate of ¹⁴C uptake in this system (Keller & Riebesell, 1989) as well as in many others (see recent review by Brush et al., 2002), this decline has almost certainly been associated with a marked decline in the rate of primary production.

The cause of the oligotrophication of Narragansett Bay is not well understood, but it cannot be attributed to a reduction in nitrogen inputs as they have remained essentially unchanged for at least 25 years (Nixon et al., 2008). There is some correlative evidence that climate variability (North Atlantic Oscillation) and long-term warming trends may be

Fig. 6 Mean annual water column chlorophyll *a* concentrations at the mid-bay station (see Fig. 1). Historical data (*closed circles*) from Li & Smayda (1998). Most recent data (*open circles*) from the Graduate School of Oceanography/URI (see <http://www.gso.uri.edu/phytoplankton>)



responsible (Hawk, 1998; Borkman, 2002; Oviatt et al., 2002; Oviatt, 2004). Winter warming in the bay is highly significant. Warming has amounted to an increase of about 1.7°C during the 1990s compared with the period between 1890 and 1970 at nearby Woods Hole, MA, where it is known that water temperature is closely correlated with that in Narragansett Bay (Nixon et al., 2003, 2004). Townsend & Cammen (1988) found that year-to-year variation in the spring bloom in the Gulf of Maine was controlled by light and not temperature. Working in Narragansett Bay, however, Borkman (2002) found that warmer winters tended to be cloudier, thus making it difficult to separate temperature and light effects. Experiments with the large MERL mesocosms have shown that slightly warmer temperatures reduce or eliminate winter–spring phytoplankton blooms in Narragansett Bay, presumably because of higher grazing pressure (Keller et al., 1999). Field evidence also confirms that warmer winters are associated with a delayed or complete elimination of the winter–spring bloom (Oviatt et al., 2002).

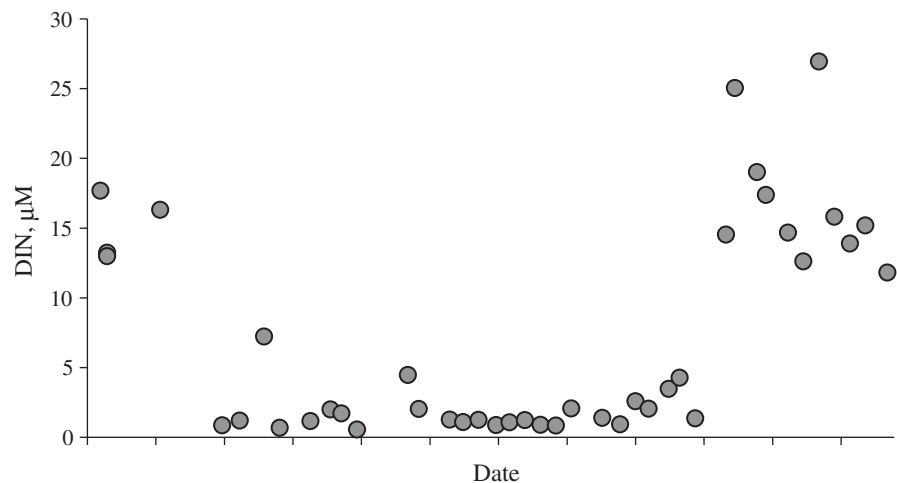
Regardless of the reason, the loss of the winter–spring bloom is particularly alarming for a benthic community that is already food-limited by the end of summer. Tracer and mass balance studies, also carried out in the MERL mesocosms, have shown that a smaller fraction of the summer blooms is deposited on the bottom compared with winter–spring blooms (Rudnick & Oviatt, 1986; Keller et al., 1999). From the field (Rudnick et al., 1985; Craig, 1989; Ellis, 2002), from mesocosm study (Grassle et al., 1985; Beatty, 1991), and from a numerical model (McKenna 1987), there is considerable evidence that certain groups of the meio- and macro-benthic infauna have

historically been food-limited during summer in Narragansett Bay.

Implications for management

There is a common impression among environmental groups, some managers, and some scientists that nuisance blooms of macroalgae (especially *Ulva* sp.) are increasing and, more importantly, that bottom waters in upper Narragansett Bay and the Providence River (actually an estuary at the head of the bay, Fig. 1) are developing more frequent, intense, and widespread hypoxia (e.g., Rhode Island Department of Environmental Management, 2003). Unfortunately, historical data are not really available to address the question of change in either of these conditions. Because of this, many of the major wastewater treatment plants that discharge directly to the bay are in the process of upgrading to tertiary treatment, thus reducing total nitrogen loading to the bay. Nixon et al. (2005) estimated that by 2010 summer nitrogen inputs to Narragansett Bay may decrease between 30% and 40%. Clearly, these reductions will have significant impacts on the new production of the bay because ¹⁴C uptake and bloom formation are strongly limited by nitrogen availability during summer (Furnas et al., 1976; Oviatt et al., 1995), when concentrations of dissolved inorganic nitrogen are very low or undetectable in the surface water of the mid- and lower bay (Fig. 7). It is anticipated that with less phytoplankton production there will be less organic matter deposition to the sediments, less benthic respiration, and therefore fewer hypoxic/anoxic events. However, we have shown here that there have already been large

Fig. 7 Annual surface water-column dissolved inorganic nitrate (DIN) concentrations at the mid-bay station for 2005. Data available at <http://www.gso.uri.edu/phytoplankton/>



reductions in pelagic chlorophyll in the mid-bay and that respiration and nutrient regeneration by the benthos of mid-Narragansett Bay have declined greatly over the past three decades. Others have shown a positive effect of nitrogen loading on benthic biomass (Josefson & Rasmussen, 2000) and a positive relationship between primary and secondary production (Nixon et al., 1986). Since climate change has already begun the oligotrophication of a once highly productive bay, it is possible that reductions of nitrogen input during the growing season will have a significant negative impact on secondary production throughout the mid- and lower bay, a much larger area than the regions currently exposed to frequent hypoxia.

Acknowledgments We thank the organizing committee and sponsors for the International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, where the material contained in this manuscript was first presented. We also thank B. Buckley for analyzing the most recent nutrient samples and S. Granger for his help in field core collection. Others who helped in core collection are: P. Dimilla, K. Hanks, M. Horn, J. Krumholz, C. Mueller, A. Pimenta, and M. Traber. Many thanks to A. Giblin and J. Tucker of the Ecosystems Center, Marine Biological Laboratory, Woods Hole, MA, for their help in denitrification analysis. J. Cornwell and M. Owens of the University of Maryland, Horn Point Laboratory helped with core design and construction, and sample analysis. The historic data were collected and/or analyzed by B. Nowicki, S. Hale, R. Chinman, and J. Kelly in collaboration with S. Nixon and C. Oviatt. R.W. Fulweiler was funded in part to complete this study by the Rhode Island Coastal Resources Management Council. The Graduate School of Oceanography Alumni Association and the Robert and Patricia Switzer Foundation provided travel support for R.W. Fulweiler to participate in the symposium.

References

- Beatty, L. L., 1991. The response of benthic suspension feeders and their grazing impact on phytoplankton in eutrophied coastal ecosystems. Ph.D. Thesis, University of Rhode Island, Graduate School of Oceanography, Narragansett, RI, 352 pp.
- Borkman, D. G., 2002. Analysis and simulation of *Skeletonema costatum* (Grev.) Cleve annual abundance patterns in lower Narragansett Bay 1959–1996. Ph.D. Thesis, University of Rhode Island, Graduate School of Oceanography, Narragansett, RI, 395 pp.
- Brush, M. J., J. W. Brawley, S. W. Nixon & J. N. Kremer, 2002. Modeling phytoplankton production: problems with the Eppley curve and an empirical alternative. *Marine Ecological Progress Series* 238: 31–45.
- Carrit, D. E. & J. H. Carpenter, 1966. Recommendation procedure for Winkler analyses of sea water for dissolved oxygen. *Journal of Marine Research* 24: 313–318.
- Colt, J., 1984. Computation of dissolved gas concentrations in water as functions of temperature, salinity, and pressure. *American Fisheries Society Special Publication* 14.
- Craig, N. I., 1989. Age and growth history characteristics of *Nucula annulata* in undisturbed and nutrient enhanced environments. M.S. Thesis, University of Rhode Island, 93 pp.
- de Jonge, V. N., W. Boynton, C. F. D'Elia, R. Elmgren & B. L. Welsh, 1994. Responses to developments in eutrophication in four different North Atlantic estuarine systems. In Dyer, K. R. & R. J. Orth (eds), *Changes in Fluxes in Estuaries*. Olsen & Olsen, Fredensberg: 179–196.
- Doering, P. H., 1989. On the contribution of the benthos to pelagic production. *Journal of Marine Research* 47: 371–383.
- Doering, P. H., C. A. Oviatt & J. R. Kelly, 1986. The effects of the filter-feeding clam *Mercenaria mercenaria* on carbon cycling in experimental marine mesocosms. *Journal of Marine Research* 44: 839–861.
- Edwards, M. & A. J. Richardson, 2004. Impact of climate change in marine pelagic phenology and trophic mismatch. *Nature* 430: 881–884.

- Ellis, G., 2002. An examination of the benthic macrofauna of Narragansett Bay and the possible implications of winter–spring bloom intensity on population size. M.S. Thesis, University of Rhode Island, Narragansett, RI.
- Emanuel, K. A., 2005. Increasing destructiveness of tropical cyclones over the past 30 years. *Nature* 436: 686–688.
- Eyre, B. D., S. Rysgaard, T. Dalsgaard & P. B. Christensen, 2002. Comparison of isotope pairing and N₂:Ar methods for measuring sediment denitrification—assumptions, modifications, and implications. *Estuaries* 25: 1077–1087.
- Farnsworth, E. J., J. Núñez-Farfán, S. A. Careaga & F. A. Bazzaz, 1995. Phenology and growth of three temperate forest life forms in response to artificial soil warming. *Journal of Ecology* 83: 967–977.
- Frithsen, J. B., 1989. The benthic communities within Narragansett Bay: an assessment completed for the Narragansett Bay Project. Rhode Island Department of Environmental Management, Providence, RI, 28 pp.
- Furnas, M. J., G. L. Hitchcock & T. J. Smayda, 1976. Nutrient-phytoplankton relationships in Narragansett Bay during the 1974 summer bloom. In Wiley, M. (ed.), *Estuarine Processes: Uses, Stresses, and Adaptation to the Estuary*, Vol. 1. Academic Press, New York: 118–133.
- Garber, J. H., 1982. ¹⁵N-tracer and other laboratory studies of nitrogen remineralization in sediments and waters from Narragansett Bay, Rhode Island. Ph.D. Thesis, University of Rhode Island, Graduate School of Oceanography, Narragansett, RI, 276 pp.
- Giblin, A., C. Hopkinson, J. Tucker, B. Nowicki & J. Kelly, 1995. Metabolism, nutrient cycling and denitrification in Boston harbor and Massachusetts Bay sediments in 1994. MWR Environmental Quality Report Series No. 95-13. MA Water Resources Authority, Boston, 56 pp.
- Grall, J. & L. Chauvaud, 2002. Marine eutrophication and benthos: the need for new approaches and concepts. *Global Change Biology* 8: 813–830.
- Grasshoff, K., 1976. *Methods of Seawater Analysis*, 2nd Edition. Verlag Chemie, Weinheim.
- Grassle, J. F., J. P. Grassle, L. S. Brown-Leger, R. F. Petrecca, N. J. Copley, 1985. Subtidal macrobenthos of Narragansett Bay. Field and mesocosm studies of the effects of eutrophication and organic input on benthic populations. In Gray, J. S. & M. E. Christiansen (eds), *Marine Biology of Polar Regions and Effects of Stress on Marine Organisms*. John Wiley, New York: 421–434.
- Groffman, P. M., M. Altabet, J. K. Bohlke, K. Butterbach-Bahl, M. B. David, M. K. Firestone, A. E. Giblin, T. M. Kana, L. P. Nielsen & M. A. Voytek, 2006. Methods for measuring denitrification: diverse approaches to a difficult problem. *Ecological Applications* 16: 2091–2122.
- Hale, S., 1974. The role of benthic communities in the nutrient cycles of Narragansett Bay. Ph.D. Thesis, University of Rhode Island, Narragansett, RI.
- Hawk, J. D., 1998. The role of the North Atlantic oscillation in winter climate variability as it relates to the winter–spring bloom in Narragansett Bay. M.S. Thesis, University of Rhode Island, Narragansett, RI.
- Inouye, D. W., B. Barr, K. B. Armitage & B. D. Inouye, 2000. Climate change is affecting altitudinal migrants and hibernating species. *Proceedings of the National Academy of Sciences* 97: 1630–1633.
- Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner & R. R. Warner, 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629–637.
- Josefson, A. B. & B. Rasmussen, 2000. Nutrient retention by benthic macrofaunal biomass of Danish estuaries: importance of nutrient load and residence time. *Estuarine, Coastal and Shelf Science* 50: 205–216.
- Kana, T. M., C. Darkangelo, M. D. Hunt, J. B. Oldham, E. Bennett & J. C. Cornwell, 1994. Membrane inlet mass spectrometer for rapid high-precision determination of N₂, O₂, and Ar in environmental water samples. *Analytical Chemistry* 66: 4166–4170.
- Karentz, D. & T. J. Smayda, 1998. Temporal patterns of variations in phytoplankton community organization and abundance in Narragansett Bay during 1959–1980. *Journal of Plankton Research* 20: 145–168.
- Keller, A. A. & U. Riebesell, 1989. Phytoplankton carbon dynamics during a winter–spring diatom bloom in an enclosed marine ecosystem: primary production, biomass, and loss rates. *Marine Biology* 103: 131–142.
- Keller, A. A., C. A. Oviatt, H. A. Walker & J. D. Hawk, 1999. Predicted impacts of elevated temperature on the magnitude of the winter–spring phytoplankton bloom in temperate coastal waters: a mesocosm study. *Limnology and Oceanography* 44: 344–356.
- Kelly, J. R. & S. W. Nixon, 1984. Experimental studies of the effect of organic deposition on the metabolism of a coastal marine bottom community. *Marine Ecology Progress Series* 17: 157–169.
- Kinney, E. H. & C. T. Roman, 1998. Response of primary producers to nutrient enrichment in a shallow estuary. *Marine Ecological Progress Series* 163: 89–98.
- Kot-Wasik, A., J. Dębska & J. Namieśnik, 2004. Monitoring of organic pollutants in coastal waters of the Gulf of Gdansk, Southern Baltic. *Marine Pollution Bulletin* 49: 264–276.
- Lambert, C. E. & C. A. Oviatt, 1983. *Manual of Biological and Geochemical Techniques in Coastal Areas*. MERL Publication Series, Report No. 1, 281 pp.
- Li, Y. & T. J. Smayda, 1998. Temporal variability of chlorophyll in Narragansett Bay 1973–1990. *ICES Journal of Marine Science* 55: 661–667.
- McKenna, J. E., 1987. An analysis of food limitation in the benthic communities of Narragansett Bay using a numerical simulation model. Ph.D. Thesis, University of Rhode Island, Graduate School of Oceanography, Narragansett, RI, 293 pp.
- McMaster, R. L., 1960. Sediments of Narragansett Bay system and Rhode Island Sound, Rhode Island. *Journal of Sedimentary Petrology* 30: 249–274.
- Nixon, S. W., C. A. Oviatt & S. S. Hale, 1976. Nitrogen regeneration and the metabolism of coastal marine bottom communities. In Anderson, J. M. & A. Macfayden (eds), *The Role of Terrestrial and Aquatic Organisms in Decomposition Processes*. Blackwell, Oxford: 269–283.
- Nixon, S. W., J. R. Kelly, B. N. Furnas, C. A. Oviatt & S. S. Hale, 1980. Phosphorus regeneration and the metabolism of coastal marine bottom communities. In Tenore, K. R. &

- B. C. Coull (eds), *Marine Benthic Dynamics*. University of South Carolina Press, Columbia: 219–242.
- Nixon, S. W., C. A. Oviatt, J. Frithsen & B. Sullivan, 1986. Nutrients and the productivity of estuarine and coastal marine ecosystems. *Journal of the Limnological Society of Southern Africa* 12: 43–71.
- Nixon, S. W., S. L. Granger & B. L. Nowicki, 1995. An assessment of the annual mass balance of carbon, nitrogen, and phosphorus in Narragansett Bay. *Biogeochemistry* 31: 15–61.
- Nixon, S. W., S. Granger & B. Buckley, 2003. The warming of Narragansett Bay. 41°N A Publication of Rhode Island Sea Grant and Land Grant Programs 2: 18–20. Available at: <http://www.seagrant.gso.uri.edu/41N/Vol2No1/baywarming.html>.
- Nixon, S. W., S. Granger, B. A. Buckley, M. Lamont & B. Rowell, 2004. A one hundred and seventeen year coastal water temperature record from Woods Hole, Massachusetts. *Estuaries* 27: 397–404.
- Nixon, S. W., B. Buckley, S. Granger, L. Harris, A. Oczkowski, L. Cole & R. Fulweiler, 2005. Anthropogenic nutrients to Narragansett Bay: a twenty-five year perspective. A Report to the Narragansett Bay Commission and Rhode Island Sea Grant. Available at http://www.narrabay.com/Documents/PDFs/Bay_Commission_Report%20.pdf.
- Nixon, S. W., B. Buckley, S. Granger, L. Harris, A. Oczkowski, R. Fulweiler & L. Cole, 2008. Nutrient inputs to Narragansett Bay: past, present, and future. In Desbonnet, A., A. B. Colt & B. A. Costa-Pierce (eds), *Science of Ecosystem-Based Estuary Management: Narragansett Bay in the 21st Century*. Springer, New York: 101–175.
- Nowicki, B. L., 1994. The effect of temperature, oxygen, salinity, and nutrient enrichment on estuarine denitrification rates measured with a modified nitrogen gas flux technique. *Estuarine, Coastal and Shelf Science* 38: 137–156.
- Nowicki, B. L. & C. A. Oviatt, 1990. Are estuaries traps for anthropogenic nutrients? Evidence from estuarine mesocosms. *Marine Ecology Progress Series* 66: 131–146.
- O'Reilly, J. E., 1984. Carbon flow in a coastal marine bottom community. Ph.D. Thesis, University of Rhode Island, 199 pp.
- Oviatt, C. A., 2004. The changing ecology of temperate coastal waters during a warming trend. *Estuaries* 27: 895–904.
- Oviatt, C. A., P. Doering, B. L. Nowicki & A. Zoppini, 1993. Net system production in coastal waters as a function of eutrophication, seasonality, and benthic macrofaunal abundance. *Estuaries* 16: 247–254.
- Oviatt, C. A., P. Doering, B. Nowicki, L. Reed, J. Cole & J. Frithsen, 1995. An ecosystem level experiment on nutrient limitation in temperate coastal marine environments. *Marine Ecology Progress Series* 116: 171–179.
- Oviatt, C., A. Keller & L. Reed, 2002. Annual primary production in Narragansett Bay with no bay-wide winter-spring phytoplankton bloom. *Estuarine, Coastal and Shelf Science* 54: 1013–1026.
- Pilson, M. E. Q., 1985. On the residence time of water in Narragansett Bay. *Estuaries* 8: 2–14.
- Pratt, D. M., 1959. The phytoplankton of Narragansett Bay. *Limnology and Oceanography* 4: 425–440.
- Rhode Island Department of Environmental Management, 2003. The Greenwich Bay Fish Kill—August 2003. Causes, Impacts and Responses Report. 32 pp.
- Rudnick, D. T. & C. A. Oviatt, 1986. Seasonal lags between organic carbon deposition and mineralization in marine sediments. *Journal of Marine Research* 44: 815–837.
- Rudnick, D. T., R. Elmgren & J. B. Frithsen, 1985. Meiofaunal prominence and benthic seasonality in a coastal marine ecosystem. *Oecologia* 67: 157–168.
- Scavia, D., J. C. Field, D. F. Boesch, R. W. Buddemeier, V. Burkett, D. R. Cayan, M. Fogarty, M. A. Harwell, R. W. Howarth, C. Mason, D. J. Reed, T. C. Royer, A. H. Salenger & J. G. Titus, 2002. Climate change impacts on U.S. Coastal and Marine Ecosystems. *Estuaries* 25: 149–164.
- Seitzinger, S., S. W. Nixon & M. E. Q. Pilson, 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. *Limnology and Oceanography* 29: 73–83.
- Sullivan, B. K., D. VanKeuren & M. Clancy, 2001. Timing and size of blooms of the ctenophore *Mnemiopsis leidyi* in relation to temperature in Narragansett Bay, Rhode Island. *Hydrobiologia* 451: 113–120.
- Townsend, D. W. & L. M. Cammen, 1988. Potential importance of the timing of spring plankton blooms to benthic-pelagic coupling and recruitment of juvenile demersal fishes. *Biological Oceanography* 5: 215–229.
- US EPA, 1983a. Methods for chemical analysis of water and wastes, EPA-600/4-79-020, Revised March 1983, Method 365.3. U.S. Environmental Protection Agency, Ohio.
- US EPA, 1983b. Methods for chemical analysis of water and wastes. EPA-600/4-79-020, Revised March 1983, Method 365.5. U.S. Environmental Protection Agency, Ohio.
- US EPA, 1997. Determination of nitrate and nitrite in estuarine and coastal waters by gas segmented continuous flow colorimetric analysis. In Ortner, P. B. & C. J. Fischer (eds), *Methods for Chemical Analysis of Water and Wastes, Revision 2.0, Method 353.4*. U.S. Environmental Protection Agency, Ohio.
- Valiela, I., K. Foreman, M. LaMontagne, D. Hersh, J. Costa, P. Peckol, B. DeMeo-Anderson, C. D'Avanzo, M. Babione, C. Sham, J. Brawley & K. Lajtha, 1992. Couplings of watersheds and coastal waters: sources and consequences of nutrient enrichment in Waquoit Bay, Massachusetts. *Estuaries* 15: 443–457.
- Walther, G., E. Post, P. Convey, A. Menzel, C. Parmesan, J. C. Beebee, J. Fromentin, O. Hoegh-Guldberg & F. Bairlein, 2002. Ecological response to recent climate change. *Nature* 416: 389–395.

Potential phytoplankton indicator species for monitoring Baltic coastal waters in the summer period

Andres Jaanus · Kaire Toming · Seija Hällfors ·
Kaire Kaljurand · Inga Lips

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 157–168.
DOI: 10.1007/s10750-009-9768-y © Springer Science+Business Media B.V. 2009

Abstract There are very few time series documenting clear trends of change in the biomass of total phytoplankton or single taxa that coincide with trends of increasing nutrient concentrations. Weekly or biweekly monitoring since 1997 on a cross section of the central Gulf of Finland (NE Baltic Sea) with similar climatic and hydrographic conditions, but different nutrient levels, provided a uniform dataset. In order to evaluate seasonal (June–September) patterns of phytoplankton succession, more than 1,200 samples were statistically analyzed by selecting 12 dominant taxa using wet weight biomass values. In addition, the continuously measured hydrographic parameters on board the ships of opportunity, and simultaneous

nutrient analyses gave high frequency information on the water masses. The objective of this study was to identify the taxa that may prove indicative in the assessment of eutrophication in the appropriate monitoring time periods. None of the most common blooming species (*Aphanizomenon* sp., *Nodularia spumigena*, and *Heterocapsa triquetra*) showed reliable correlations with enhanced nutrient concentrations. The species we suggest as reliable eutrophication indicators—oscillatorial cyanobacteria and the diatoms *Cyclotella choctawhatcheeana* and *Cylindrotheca closterium*—showed the best relationships with total phosphorus concentrations. Their maxima appear toward the end of July or in August–September when phytoplankton community structure is more stable, and less frequent observations may give adequate results. Another diatom, *Skeletonema costatum*, exhibited stronger correlations with dissolved inorganic and total nitrogen in June, during the period of the summer phytoplankton minimum.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

A. Jaanus (✉) · K. Toming · K. Kaljurand
Estonian Marine Institute, University of Tartu, Mäealuse
10a, 12618 Tallinn, Estonia
e-mail: andres.jaanus@ut.ee

S. Hällfors
Marine Research Centre, Finnish Environment Institute,
Erik Palménin aukio 1, 00561 Helsinki, Finland

I. Lips
Marine Systems Institute, Tallinn University
of Technology, Akadeemia 21, 12618 Tallinn, Estonia

Keywords Phytoplankton species composition ·
Nutrients · Monitoring · Baltic Sea

Introduction

Understanding and modeling an aquatic ecosystem is not possible without knowledge of the species

composition and biomass of phytoplankton, representing the base of the pyramid of productivity. According to the EU Water Framework Directive (WFD), phytoplankton is also determined as one of the biological quality elements for the classification of the ecological status of surface waters (European Communities, 2000). Increases in nutrient inputs lead directly to enhanced primary productivity, and phytoplankton may serve as an indicator of the trophic state.

Phytoplankton biomass in estuarine and coastal marine systems has been considered to be strongly dependent on the mean concentrations of total nitrogen (TN) and total phosphorus (TP) in the water column and generally predictable from water column TN:TP ratios (Hoyer et al., 2002; Smith, 2006). There are also more circumspect opinions concerning the impact of N loading on algal production rate and biomass (Borum, 1996; Guildford & Hecky, 2000) and on the availability of nutrients in proportion to their concentrations (Söderström, 1996). Moreover, Paerl et al. (2006) recalled that seasonal hydrologic perturbations can overwhelm nutrient controls on the floral composition, underscoring potential difficulties in predicting the responses of phytoplankton production and species composition to nutrient input reductions.

During the last decade, some new methods have been introduced in phytoplankton monitoring such as pigment analysis (Millie et al., 1993) and optical remote sensing (e.g., Gordon et al., 1983). In the Baltic Sea area, satellite remote sensing has been used for detecting cyanobacterial blooms (e.g., Kahru et al., 1994; Kutser, 2004). However, despite the effectiveness of new techniques in covering large areas in a short time period, microscopy is still the standard method for detailed qualitative assessment. Moreover, phytoplankton pigment composition usually gives adequate information only at the class level, which is not sufficient for ecological assessment, as different species even from the same genus may occupy different ecological niches.

Phytoplankton usually form communities which are highly complex and variable in terms of diversity and dynamics. As the communities can change fundamentally on a weekly scale, proper assessment of phytoplankton community composition requires large investments in offshore sampling and experienced personnel for microscopic identification and

quantification of phytoplankton. Recent phytoplankton monitoring in the Baltic Sea has directed emphasis toward intensive (weekly to bi-weekly) sampling at a few locations instead of extensive, but less frequent sampling covering larger sea areas. Previous monitoring programs have focused mainly on open-sea areas and assumed a uniform horizontal distribution of phytoplankton biomass. However, it is the coastal areas that are primarily subjected to eutrophication (Wasmund et al., 2001). The authors have applied a less expensive and higher frequency sampling method using ships-of-opportunity. Taking into account spatial and temporal dimensions, it should give more adequate information on plankton communities and dynamics than traditional monitoring.

Although the Baltic Sea is well investigated, there are very few published relationships between the trends in nutrient concentrations and the biomass of total phytoplankton or single taxa. The impact of nutrient enrichment on the phytoplankton community structure in the northern Baltic Sea has been observed by Lagus et al. (2004) and Vuorio et al. (2005). Gasiùnaite et al. (2005) tried to delineate a set of phytoplankton community indicators in the Baltic Sea that are independent of season and salinity, but indicative of the trophic status of different coastal areas.

In this article, the authors first aim to determine seasonal and spatial patterns for selected dominant phytoplankton species and to evaluate the relationships of selected taxa to external nutrient levels. High-frequency phytoplankton sampling should also enable the collection of information on a reasonable and cost-effective monitoring design.

Materials and methods

All measurements and sampling were conducted onboard passenger ferries (*Wasa Queen*, *Finnjet*, and *Romantika*) plying between Tallinn and Helsinki, in the central Gulf of Finland during the summer period (June–September) in 1997–2005. Water was pumped through an inlet from a depth of about 5 m onboard the moving ship. Temperature and salinity were recorded quasi-continuously with a spatial resolution of about 150–200 m using an Aanderaa thermosalinograph. Water samples for the analysis of nutrients and phytoplankton were obtained from an

automated sampler (ISCO); water was sampled weekly at nine fixed locations (Fig. 1) and kept refrigerated (4°C; 2–12 h depending on the schedule) in the dark before analysis. Altogether 1,296 samples were analyzed during the investigation period.

Nutrient concentrations (TN, NO₂-N + NO₃-N, TP, and PO₄-P) were obtained as described by HELCOM (<http://sea.helcom.fi/Monas/CombineManual2/PartC/CFrame.htm>). Until 2003, inorganic nitrogen and phosphates were analyzed according to standard methods of the Finnish Standards Association SFS (SFS-EN ISO 13395, SFS 3025).

Phytoplankton analysis

The authors assumed that the sampling depth (~5 m) represents the upper surface layer, as the ship creates turbulence when moving. The depth of the euphotic layer in the Gulf of Finland varies from 5 to 20 m and, consequently, the water sampled from the upper surface layer was considered to be representative for the phytoplankton in the research area. Phytoplankton samples were preserved with acid Lugol solution (0.5–1 ml per 200 ml sample) and analyzed using the inverted microscope technique (Utermöhl, 1958) within 1 or 2 weeks after sampling. The sedimented volume was mainly 25 ml except some samples from

the Helsinki sea area, where 10 ml chambers were used due to the large number of cells. No replicates were analyzed.

Wet weight biomass of phytoplankton was calculated from cell geometry (HELCOM, 1988; Hillebrand et al., 1999) and using cell volumes (PhytoWin, Software Kahma Ky) or after original measurements. All taxa were identified to the level achievable in fixed material using light microscopes (Olympus IM, Olympus IMT-2 and Leitz Fluovert FU). Most taxa were identified and counted using 200× magnification, except for small flagellates (<20 µm) and small centric diatoms, where 400× magnification was used. Autotrophic picoplankton were not included in the analysis. Dominant taxa constituting >10% of total nano- and microphytoplankton biomass in any sample were selected for further analysis (Table 1). In addition to the taxa listed in Table 1, *Chrysochromulina* spp. occurred in masses, especially in 1999, 2002, and 2004. Due to the probable misidentification in some samples, this genus was not included in the statistical analyses.

Data analysis

The software package PRIMER, version 5 developed at the Plymouth Marine Laboratory was used in statistical analyses (Clarke & Warwick, 2001). Variation in biomass of different species among sampling stations and weeks was subsequently tested for significance with an ANOSIM (analysis of similarity) procedure. A Spearman rank correlation (r) was computed between the similarity matrices of community and environmental data to examine the ecological significance of environmental variables in the dynamics of selected species (BIO-ENV). Non-metric multidimensional scaling (MDS) was used to examine temporal variation in phytoplankton, as well as to modify sampling frequency in the future according to stability in summer communities. For univariate analysis, the statistical program “Statistica” version 6 (StatSoft Inc., 2004) and MS Excel were used. Correlation and linear regression analyses were used to describe the relationships between nutrient concentrations and biomass variables. Earlier studies in the Gulf of Finland have indicated that the time lag between a nutrient pulse and the response of phytoplankton is a few days (Rantajarvi et al., 1997). Therefore, dissolved inorganic nitrogen (DIN) and

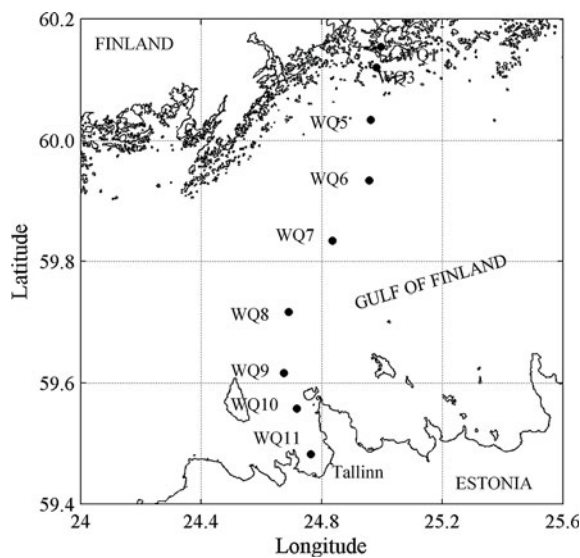


Fig. 1 Chart of the study area in the Gulf of Finland, eastern Baltic Sea with sampling stations (WQ1–WQ11) along the ferry route between Tallinn and Helsinki

Table 1 The dominant phytoplankton taxa in the Gulf of Finland used for statistical analysis (classification according to Hällfors, 2004)

| Division | Class | Order | Species |
|--------------|--|-----------------|--|
| Cyanophyta | <i>Nostocophyceae</i> (<i>Cyanophyceae</i>) | Oscillatoriales | Mainly from genera <i>Pseudanabaena</i> and <i>Planktolyngbya</i> |
| | | Nostocales | <i>Anabaena</i> spp. <i>Aphanizomenon</i> sp. <i>Nodularia spumigena</i> Mertens |
| | | | |
| Dinophyta | <i>Dinophyceae</i> | Prorocentrales | <i>Prorocentrum minimum</i> Schiller |
| | | Peridinales | <i>Heterocapsa triquetra</i> Stein |
| Chrysophyta | <i>Diatomophyceae</i> | Eupodiscales | <i>Cyclotella choctawhatcheeana</i> Prasad <i>Skeletonema costatum</i> Greville |
| | | Bacillariales | <i>Cylindrotheca closterium</i> Reimann and J. Lewin |
| Euglenophyta | <i>Euglenophyceae</i> | Euglenales | <i>Eutreptiella gymnastica</i> Throndsen |
| Chlorophyta | <i>Chlorophyceae</i> | Chlorococcales | <i>Monoraphidium contortum</i> Komárková-Legnerová |
| | <i>Prasinophyceae</i> | Chlorodendrales | <i>Pyramimonas</i> spp. |

dissolved inorganic phosphorus (DIP) were taken with 1-week differences compared with biotic variables for statistical analyses. Other abiotic variables used in the analysis had the same spatial and temporal coverage as the phytoplankton biomass. Significance was accepted at $p < 0.001$ throughout this article.

Results

Hydrological and nutrient conditions

Usually, the minima of the water temperatures were recorded at the beginning of the summer in weeks 22–26 (minimum 5.2°C; 29 May 2002) and maxima toward the end of July or in August in weeks 29–34 (maximum 23.2°C, 5 August 2003). The seasonal average water temperature varied only slightly (from 15.0 to 15.8°C) at the sampling stations for biological measurements in 1997–2005.

The mean water salinity ranged between 5.1 and 5.8 and increased slowly along the study transect from the Finnish to the Estonian coast. Although water salinity varied between weeks, no regular changes in the temporal pattern of salinity were observed.

Winter (December–February) $\text{NO}_3 + \text{NO}_2\text{-N}$ (DIN) and $\text{PO}_4\text{-P}$ (DIP) concentrations were higher near the Finnish coastal waters (stations WQ1 and WQ3; 9–15 μM DIN and 1.1–1.3 μM DIP). There

was no marked difference in inorganic nutrient concentrations between the open Gulf of Finland and the Estonian coastal waters (Fig. 2). According to winter nutrient values, the entire area was N limited with an N:P ratio of 10.1–13.4 near the Finnish coast, and 7.7 and 8.7 on average in the central and southern parts of the route, respectively.

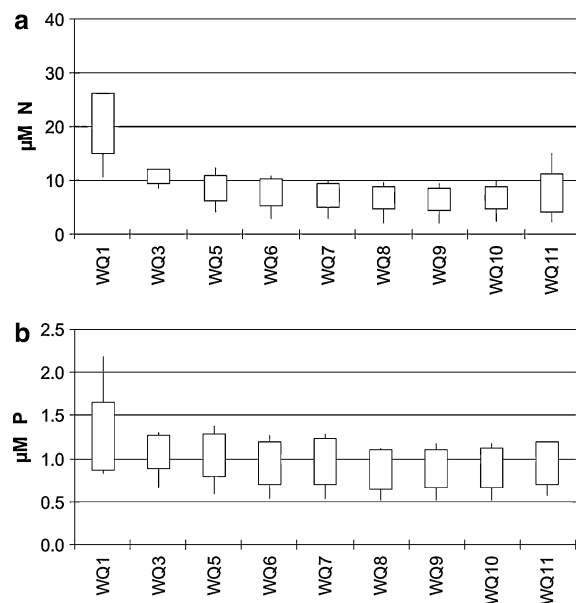


Fig. 2 Winter concentrations (Dec–Feb) of inorganic nitrogen ($\text{NO}_3 + \text{NO}_2\text{-N}$; **a**) and phosphorus ($\text{PO}_4\text{-P}$; **b**) (μM) at sampling stations. Mean \pm standard deviation in boxes; whiskers indicate maxima and minima

The mean concentrations of TP and TN were markedly higher at the sampling stations WQ1 (1.17 μM TP; 33.64 μM TN) and WQ3 (0.89 μM TP; 27.28 μM TN) and slightly higher at the sampling stations WQ10 (0.74 μM TP; 26.10 μM TN) and WQ11 (0.77 μM TP; 26.52 μM TN) (Fig. 3). The mean concentrations of TP measured across the Gulf of Finland varied between weeks and peaked at the end of May and the beginning of June (weeks 22–23). The mean concentrations of TN were somewhat higher in weeks 28–32 (mid-July to the beginning of August).

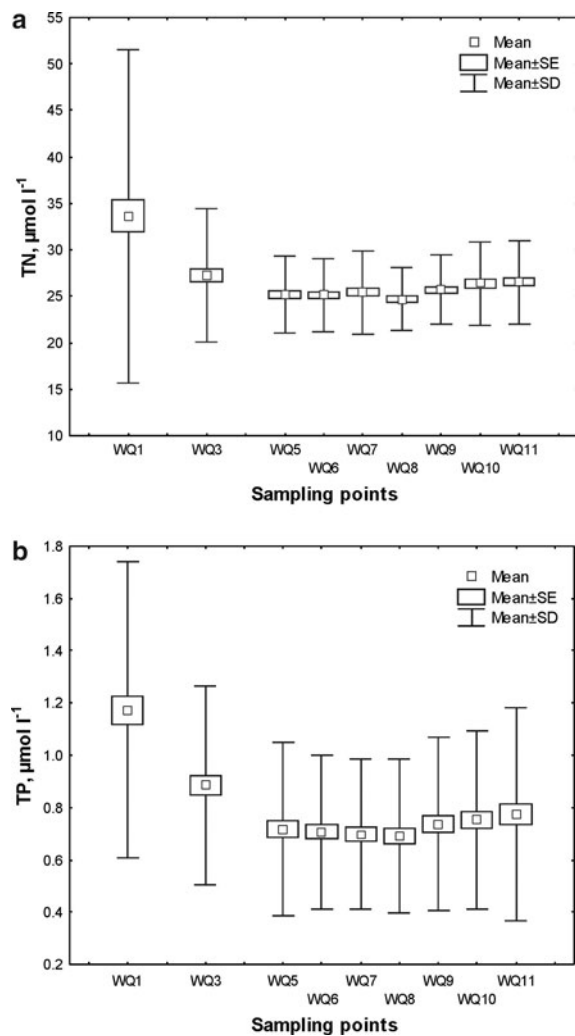


Fig. 3 Mean concentrations (μM), standard error (SE), and standard deviation (SD) of total nitrogen (TN; **a**) and total phosphorus (TP; **b**)

Dynamics of phytoplankton

The maximum biomass values of the diatom *Skeletonema costatum* and the green alga *Monoraphidium contortum* were observed mainly at the end of May or the beginning of June. The biomass of the cyanobacterium *Aphanizomenon* sp. was higher in June–July, while biomasses of *Anabaena* spp., *Nodularia spumigena*, the euglenophyte *Eutreptiella gymnastica*, and the prasinophyte *Pyramimonas* spp. were higher in July. Oscillatorial cyanobacteria and the dinoflagellate *Heterocapsa triquetra* reached their biomass peak generally in July–August. The diatoms *Cyclotella choctawhatcheeana* and *Cylindrotheca closterium* showed the highest biomass in August. The dinophyte *Prorocentrum minimum* appeared among dominant species only at the end of August and in September 1999 and 2003.

Similar to nutrient concentrations, biomasses of *H. triquetra*, *C. choctawhatcheeana*, *S. costatum*, *C. closterium*, and *E. gymnastica* were usually higher at the sampling stations situated near the coast (mainly in Finnish coastal waters) than in the central Gulf of Finland. However, the ANOSIM test showed only slight statistical differences between coastal and open-sea stations.

Relationships between abiotic and biotic variables

The relationships between abiotic and biotic variables as analyzed by linear correlation, regression, and multivariate BIO-ENV analyses are shown in Table 2. The biomass of the cyanobacteria *Aphanizomenon* sp., *Anabaena* spp., and *Nodularia* did not show a significant positive correlation with nutrients. *N. spumigena* and Oscillatoriales exhibited a positive correlation with temperature. Oscillatorial biomass increased with elevated TP. The biomass of the dinophyte *Prorocentrum minimum* did not show any correlation with abiotic variables at the determined significance level; nonetheless, multivariate BIO-ENV analyses gave the best match with TP and DIN. The dinophyte *Heterocapsa triquetra* correlated positively with DIP, TP, and DIN, but negatively with temperature.

The diatoms *Cyclotella choctawhatcheeana*, *Cylindrotheca closterium*, and *Skeletonema costatum* showed the strongest relationships with nutrients. The biomass of *C. choctawhatcheeana* gave a good

Table 2 Coefficients of determination (r^2), correlations (r), and Spearman rank correlations (BIO-ENV) between phytoplankton biomass and abiotic variables

| | Temperature | Salinity | TP | PO ₄ -P | TN | NO ₃ + NO ₂ -N |
|--|--------------|--------------|--------------|--------------------|-------------|--------------------------------------|
| <i>Oscillatoriales</i> , Weeks 28–35, $n = 575$ | | | | | | |
| r | 0.24 | -0.17 | 0.38 | 0.02 | 0.11 | -0.01 |
| r^2 | 0.06 | 0.03 | 0.15 | 0.00 | 0.01 | 0.00 |
| BIO-ENV | 0.01 | 0.01 | 0.07 | 0.00 | 0.06 | 0.04 |
| <i>Anabaena</i> spp., Weeks 27–30, $n = 288$ | | | | | | |
| r | -0.06 | -0.21 | 0.01 | -0.15 | 0.02 | -0.08 |
| r^2 | 0.00 | 0.04 | 0.00 | 0.02 | 0.00 | 0.01 |
| BIO-ENV | 0.03 | 0.14 | 0.03 | 0.08 | 0.04 | 0.11 |
| <i>Aphanizomenon</i> sp., Weeks 25–31, $n = 501$ | | | | | | |
| r | 0.03 | -0.11 | -0.08 | -0.21 | 0.03 | -0.07 |
| r^2 | 0.00 | 0.01 | 0.01 | 0.04 | 0.00 | 0.01 |
| BIO-ENV | 0.09 | -0.01 | 0.05 | 0.14 | -0.01 | 0.06 |
| <i>Nodularia spumigena</i> , Weeks 27–31, $n = 359$ | | | | | | |
| r | 0.23 | -0.05 | -0.22 | -0.19 | 0.05 | -0.07 |
| r^2 | 0.05 | 0.00 | 0.05 | 0.04 | 0.00 | 0.00 |
| BIO-ENV | 0.09 | 0.03 | 0.06 | 0.09 | 0.03 | 0.07 |
| <i>Prorocentrum minimum</i> , Weeks 34–35, $n = 144$ | | | | | | |
| r | -0.04 | -0.05 | 0.24 | -0.08 | 0.00 | -0.16 |
| r^2 | 0.00 | 0.00 | 0.06 | 0.01 | 0.00 | 0.03 |
| BIO-ENV | 0.01 | -0.07 | 0.08 | 0.00 | -0.08 | 0.06 |
| <i>Heterocapsa triquetra</i> , Weeks 28–35, $n = 575$ | | | | | | |
| r | -0.19 | 0.00 | 0.23 | 0.23 | 0.03 | 0.26 |
| r^2 | 0.04 | 0.00 | 0.05 | 0.05 | 0.00 | 0.07 |
| BIO-ENV | 0.02 | 0.09 | 0.03 | 0.03 | 0.04 | 0.07 |
| <i>Cyclotella choctawhatcheeana</i> , Weeks 31–35, $n = 360$ | | | | | | |
| r | 0.10 | -0.34 | 0.34 | 0.16 | 0.19 | 0.18 |
| r^2 | 0.01 | 0.11 | 0.12 | 0.03 | 0.04 | 0.03 |
| BIO-ENV | -0.02 | 0.07 | 0.16 | 0.09 | 0.06 | 0.05 |
| <i>Skeletonema costatum</i> , Weeks 22–26, $n = 288$ | | | | | | |
| r | -0.12 | -0.16 | 0.14 | 0.03 | 0.31 | 0.31 |
| r^2 | 0.01 | 0.02 | 0.02 | 0.00 | 0.10 | 0.10 |
| BIO-ENV | 0.11 | 0.02 | 0.10 | 0.05 | 0.11 | 0.03 |
| <i>Cylindrotheca closterium</i> , Weeks 32–35, $n = 288$ | | | | | | |
| r | 0.06 | -0.12 | 0.44 | 0.21 | 0.04 | -0.01 |
| r^2 | 0.00 | 0.01 | 0.20 | 0.05 | 0.00 | 0.00 |
| BIO-ENV | 0.00 | 0.08 | 0.20 | 0.01 | -0.03 | 0.08 |
| <i>Eutreptiella gymnastica</i> , Weeks 25–28, $n = 288$ | | | | | | |
| r | 0.04 | 0.01 | 0.01 | 0.01 | 0.00 | 0.00 |
| r^2 | 0.19 | -0.10 | -0.10 | -0.10 | 0.07 | -0.03 |
| BIO-ENV | -0.01 | 0.03 | 0.06 | 0.02 | 0.00 | 0.04 |
| <i>Pyramimonas</i> spp., Weeks 27–35, $n = 597$ | | | | | | |
| r | 0.21 | -0.19 | -0.02 | -0.13 | 0.21 | 0.00 |
| r^2 | 0.04 | 0.04 | 0.00 | 0.02 | 0.04 | 0.00 |

Table 2 continued

| | Temperature | Salinity | TP | PO ₄ -P | TN | NO ₃ + NO ₂ -N |
|---|--------------|--------------|-------------|--------------------|-------------|--------------------------------------|
| BIO-ENV | 0.06 | 0.09 | 0.01 | 0.05 | 0.08 | 0.02 |
| <i>Monoraphidium contortum</i> , Weeks 22–26, $n = 288$ | | | | | | |
| r | -0.20 | -0.32 | 0.02 | 0.04 | -0.16 | -0.04 |
| r^2 | 0.04 | 0.10 | 0.00 | 0.00 | 0.03 | 0.00 |
| BIO-ENV | 0.04 | 0.11 | 0.13 | 0.02 | 0.10 | -0.05 |

Values with $p < 0.001$ are marked in bold

n number of samples

positive correlation with TP, TN, and DIN, *C. closterium* with TP and DIP, and *S. costatum* with TN and DIN. In addition, *C. choctawhatcheeana* correlated negatively with salinity. The biomass of the euglenophyte *Eutreptiella gymnastica* did not show any significant correlation with abiotic variables; however, multivariate BIO-ENV analyses showed a positive correlation with TP, DIN, and salinity.

The biomass of *Pyramimonas* spp. showed a positive and *Monoraphidium contortum* a negative correlation with TN. *M. contortum* correlated negatively with salinity and temperature and *Pyramimonas* spp. with salinity.

Temporal variation in phytoplankton communities

Sampling frequency is determined by the variability, and optional increased sampling frequency in seasons with the main bloom events is recommended. Since traditional water quality monitoring is quite costly, it is very important to design properly the monitoring network so that the maximum amount of information can be extracted and forwarded to decision makers with moderate effort.

The authors conducted frequent sampling with weekly resolution and carried out 16–18 sampling events per station during the summer period from June to September. The results of ANOSIM analysis indicated that in the Gulf of Finland and probably in the whole northern Baltic Sea the most rapid changes, namely, mass development of phytoplankton caused mainly by the cyanobacterial species *Aphanizomenon* sp., *Anabaena* spp., and *Nodularia spumigena*, occur toward the end of June and in July—weeks 25–29 (Fig. 4). In this figure, the distance between the week

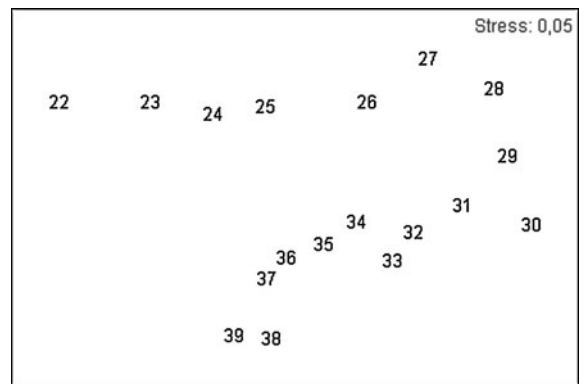


Fig. 4 MDS ordination of square-root transformed (June–September, weeks 22–39) phytoplankton biomass data of weekly resolution from the ferry route between Tallinn and Helsinki in 1998–2004. The distances between the points (week numbers on the calendar) are proportional to dissimilarities and correspond to the rate of change in the phytoplankton community structure

numbers of the calendar corresponds to dissimilarities in community structure. A longer distance between the numbers indicates a rapid change in community, while a shorter distance indicates more stability in community structure. In addition, weeks 22–24 at the beginning of June, usually known as the period of summer biomass minimum, may need more frequent surveys. In our study, those weeks were characterized by a rather regular biomass maximum of the diatom *Skeletonema costatum* especially in Finnish coastal waters. In August and September, phytoplankton communities were more stable, and less frequent monitoring (1–2 times per month) could provide adequate information on both species composition and biomass. The authors suggest that for surveillance and operative monitoring, 7–8 observations would be a minimum for the period from June to

September. In the period of the most probable bloom events (July in the Gulf of Finland), the recommended frequency is 3–4 times per month.

Discussion

No single indicator provides adequate information concerning the multiple, interrelated components of the ecosystem (Turner et al., 2006). On the other hand, whole-ecosystem experiments are not really possible in the study of coastal eutrophication, and thus, there is incomplete understanding of how the various components of the ecosystem interact (Cloern, 2001; Rogers & Greenaway, 2005). Therefore, the more realistic procedure has been the collection and interpretation of time series of observations.

The short duration of this study (years 1997–2005) precludes any assessment of long-term changes in the biomass of phytoplankton species. However, nine years of monitoring data still provide important information for the assessment of the inter-seasonal biomass variability of phytoplankton and the general biomass pattern for each species during the investigation period.

The authors were able to define the periods of biomass peaks for all selected dominant taxa and could follow the change on a community level at a weekly resolution. This allowed us to conclude that in the northern Baltic Sea, the most rapid changes in summer phytoplankton community structure take place in June–July and the shifts slow down in August–September. According to our data, July is the period of mass development of filamentous cyanobacteria (*Aphanizomenon* sp., *Nodularia spumigena* and *Anabaena* spp.), often forming surface accumulations. Some earlier data from the Gulf of Finland and northern Baltic Sea associate cyanobacteria-dominated blooms more with the late summer period (Niemi, 1979; Kononen, 1992). Our data also call into question the indicative value of these species concerning the trophic status of coastal waters. We could not find any relationships supporting their predominance in more eutrophic conditions, that is, in the Finnish coastal area in our study. Moreover, as long as the Baltic Sea has been brackish, cyanobacterial blooms have been recurrent (e.g., Bianchi et al., 2000). Therefore, the blooming of blue-green algae

may not, by itself, be a sign of high trophic conditions, as also suggested by Gasiùnaite et al. (2005). In addition, the nutrient enrichment experiments conducted in the northern Baltic Sea by Vuorio et al. (2005) did not indicate any effect on the total biomass of N₂-fixing cyanobacteria. Ships-of-opportunity data collected in the Gulf of Finland and interpreted by Rantajärvi et al. (1997) have shown a negative correlation of *Aphanizomenon* sp. to total nitrogen. It is likely that the dynamics and amplitude of filamentous diazotrophic cyanobacteria blooms are governed mainly by hydrodynamic factors and are natural phenomena in the Baltic Sea (e.g., Bianchi et al., 2000; Finni et al., 2001a).

Other cyanobacteria from the order Oscillatoriales exhibited a positive relationship with TP. Oscillatoriales also showed a good positive correlation with temperature and a negative correlation with salinity, indicating its freshwater origin and preference for warm water. Similar results were published by Rantajärvi et al. (1997). Based on recent plankton data and concerning the relationship between plankton and trophic level, Finni et al. (2001b) attributed the dominance of oscillatorean cyanobacteria to considerably eutrophic or hypereutrophic conditions. Nixdorf et al. (2003) also stated that the dominance of Oscillatoriales in lakes is a result of an anthropogenically induced eutrophication process. However, after analyzing phytoplankton data from shallow hypertrophic lakes, they concluded that turbid mixed layers with very light deficient conditions seem to be the key factor favoring the dominance of some oscillatorean genera such as *Pseudanabaena* and out-competing N₂-fixing cyanobacteria.

Many of the common dinoflagellates may potentially be mixotrophic species, partly independent of ambient nutrient conditions. Nonetheless, multivariate BIO-ENV analyses showed positive correlations of the dinophyte *Prorocentrum minimum* biomass to TP and DIN. A positive correlation between the success of *P. minimum* in the Baltic Sea and high concentrations of TP and TN have previously also been observed by Hajdu et al. (2004) and Pertola et al. (2004). In the Chesapeake Bay, blooms of *P. minimum* appeared to be stimulated by the addition of dissolved organic nitrogen including urea and humic acids (Heil, 2005). Nevertheless, the mass occurrence of *P. minimum* in the northern Baltic areas is probably related to the intrusion of water masses

from southern basins under favorable meteorological and hydrodynamic conditions. In our material, *P. minimum* was found in large numbers only in the late summer and autumn of 1999 and 2003.

The dinophyte *Heterocapsa triquetra* showed a moderate relationship with TP, DIP, and DIN. Lindholm & Nummelin (1999) related the development of red tides (discolored water) to a high phosphorus level and low TN:TP ratio with earlier indications of the success of *H. triquetra* in quite eutrophic conditions (e.g., Niemi & Hällfors, 1974). However, Olli & Seppälä (2001) demonstrated apparent vertical movements of *H. triquetra* (diurnal surface and nocturnal deep aggregations) in a mesocosm experiment, which make the species very independent of the availability of a nutrient supply in the mixed surface layer. Lindholm & Nummelin (1999) also suggested that this species is probably favored by frequent mixing with a more continuous supply of nutrients, for example, by the stimulating effect of water movements caused by ferries. In our area, the depths of the deepest sampling stations near the Finnish and Estonian coasts (stations WQ1 and WQ11) are 15 and 30 m, respectively. In the shallow Finnish archipelago, all ships are required to reduce their speed to 10–12 knots to avoid the destructive and disturbing effect of waves on the coast, sediments and bottom biota. It is unlikely that ship movements can mix the water column below the thermocline or nutricline and create additional nutrient supply for phytoplankton.

Diatoms generally prevail during the first phase of the succession (spring bloom) in temperate and boreal coastal areas and are generally adapted to more turbulent environments characterized by high nutrient concentrations. Some diatom species are still a significant part of the summer and autumn phytoplankton communities as well and may form secondary biomass peaks. Biomasses of the diatoms *Cyclotella choctawhatcheeana*, *Cylindrotheca closterium* and *Skeletonema costatum* showed good correlation with the nutrients and, as was the case for nutrient concentrations, the highest values of biomasses were observed at the sampling stations situated near the coast (WQ1 and WQ3). *C. choctawhatcheeana* and *C. closterium* showed the strongest relationship with TP and *S. costatum* with TN and DIN. A nutrient enrichment experiment conducted by Lagus et al. (2004) showed that *S. costatum* is P limited.

The biomass of the euglenophyte *Eutreptiella gymnastica* was positively correlated with TP and DIN. Olli et al. (1996) characterized this species as having a particular combination of environmental adaptations such as a high growth potential in environments with elevated nutrient levels and a flexible behavior (vertical migrations, cyst formation). In our material, *E. gymnastica* reached bloom-like concentrations throughout almost the entire investigation period and showed a rather irregular shape of spatiotemporal distribution. While it is true that the absolute maximum biomass values were recorded in coastal sites, the opportunistic character of such short-term peaks does not allow relating *E. gymnastica* (or also *S. costatum*) blooms to the background biogenic level unambiguously. Nonetheless, we do not exclude the possibility that the elevated biomass of opportunistic species may be a response to impairment of the ecological status in coastal waters.

Monoraphidium contortum has previously been observed at both low (Kononen, 1988; Kuosa, 1988) and high (Andersson et al., 1996) inorganic N:P ratios in the Baltic Sea. Andersson et al. (1996) suggested that this species could be a good indicator for eutrophication due to its increased abundance with increasing phosphorus concentrations in their nutrient addition experiment. Our dataset gave negative correlations to salinity and temperature, but no relationship to enhanced nutrients.

Eutrophication assessments are often based on the calculation of mean or median values over the assessment period (year, season). This assumes that the monitoring data should be distributed approximately equidistantly over the period considered. Averaged values are mainly applicable for physicochemical parameters, including, for example, chlorophyll *a* (e.g., Painting et al. 2005). We provide information here that for a single phytoplankton species, seasonal mean or median values are not relevant, as the period for their occurrence in the water column may be limited to only some weeks. Many species can be found in samples all year round, but their biomass values remain inconsiderable and peak only very briefly. For example, the diatoms *Cyclotella choctawhatcheeana* and *Cylindrotheca closterium* were mostly absent in plankton in June–July, but *Skeletonema costatum* disappeared after the middle of July. The dinophyte *Prorocentrum*

minimum was found only in September. This means that for some potential indicator species, the arguments for averaging biomass values over the whole assessment period (usually June–September in the northern Baltic Sea) are poor and may lead to inadequate conclusions. On the other hand, considering the labor expended in analyzing phytoplankton, the number of samples investigated can be increased by making selections in counting. This is a reason why the authors conducted analyses on data from selected periods.

According to our results, the diatoms *C. choctawhatcheeana*, *C. closterium*, and *S. costatum* and the oscillatorean cyanobacteria are the most reliable indicators in terms of eutrophication in the northern Baltic Sea. The biomass of diatoms did not correlate with temperature or salinity with the exception of the biomass of the diatom *C. choctawhatcheeana* that correlated negatively with salinity. This may set some limits on using this species as a water quality indicator in larger, more than basin-wide scales or it will require salinity adjustments when giving numerical values for certain quality classes. However, in the estuarine-type Chesapeake Bay this species has been found to be widely tolerant to salinity changes and nutrient inputs (Marshall & Alden, 1990). According to sediment core analysis, Cooper (1995) and Clarke et al. (2006) associated *C. choctawhatcheeana* with perturbations by anthropogenic activities. *C. closterium* is considered a cosmopolitan species (Tomas, 1997). We did not find any literature data about this species in the Baltic Sea area; however, in the eutrophic Haapsalu Bay (West Estonian coast), *C. closterium* formed a biomass peak in August 2000 similar to the spring maximum (Jaanus, 2003).

Various coastal environments experiencing increased nutrient loading have exhibited general trends of shifts in phytoplankton community dominance from larger to smaller cells (Cederwall & Elmgren, 1990; Cadee, 1992). Finni et al. (2001b) found that small or fragile forms of diatoms (*Skeletonema* spp., *Chaetoceros minimus*, small centrales, etc.) as well as small flagellated forms (Cryptophyceae, Prymnesiophyceae, Chrysophyceae, Prasinophyceae, and euglenoid *Eutreptiella* spp.) may be well represented in areas with considerable eutrophication. The investigation period was too short to follow any shifts in dominant plankton species.

However, in Finnish coastal waters with higher nutrient concentrations, some small-sized species (*C. choctawhatcheeana*, *S. costatum*, *E. gymnastica*) were more abundant than in the open Gulf of Finland or in Estonian coastal waters.

Concerning small diatoms, fragile lightly silicified species do not preserve well in the sediments leading to the under-representation of planktonic diatoms in sediment cores (Clarke et al. 2006) and making them unreliable as paleo-indicators. In addition, small diatom blooms cannot be detected by pigment analysis (Wänstrand & Snoeijs, 2006). This confirms the necessity of microscopic determination of phytoplankton not only as a supporting measure, but also as the most used and qualitative tool in assessing both ecological status and as an early warning indicator. However, the seasonal changes and interannual variations require a fairly extensive sampling frequency. Many plankton species show substantial year-to-year fluctuations that may be due to varying weather conditions or other circumstances not related to wastewater load. Another disadvantage may be a lack of taxonomic competence or subjective interpretation of the researchers.

Conclusions

Frequent sampling in the central Gulf of Finland, north-eastern Baltic Sea over nine years (1997–2005) using ships-of-opportunity provided important information for the assessment of the inter-seasonal variability and the general biomass pattern of dominant phytoplankton taxa. We defined the periods of biomass peaks for single species and examined the rates of change in phytoplankton community structure during the summer period (June–September). The most rapid changes occurred at the end of June and in July, while in August and September, phytoplankton communities were more stable. Knowledge of the stability in summer communities allowed modifications to be made to the sampling frequency and greater focus to be placed on certain periods and species that are important in terms of assessing water quality. As the period of their occurrence in the water column is often limited to a few weeks, calculation of seasonal means of physicochemical parameters is not relevant for potential phytoplankton indicator species.

According to our results, the diatoms *Cyclotella choctawhatcheeana* and *Cylindrotheca closterium* as well as the oscillatorean cyanobacteria with maxima in August and the diatom *Skeletonema costatum* with a maximum in June are the most reliable indicators in terms of eutrophication in the northern Baltic Sea.

Acknowledgments We thank two anonymous reviewers for helpful comments on the manuscript. This work was financially supported by the Estonian and Finnish Ministries of Environment, and the EU project FerryBox (EVK2-2001-00230). The cooperation of Silja Line and Tallink shipping companies made data collection possible. The nutrient analyses were conducted by the City of Helsinki Environment Centre and Uusimaa Regional Environmental Centre laboratories in 1997–2003.

References

- Andersson, A., S. Hajdu, P. Haecy, J. Kuparinen & J. Wikner, 1996. Succession and growth limitation of phytoplankton in the Gulf of Bothnia (Baltic Sea). *Marine Biology* 126: 791–801.
- Bianchi, T. S., P. Westman, C. Rolff, E. Engelhaupt, T. Andr n & R. Elmgren, 2000. Cyanobacterial blooms in the Baltic Sea: natural or human induced? *Limnology and Oceanography* 45: 716–726.
- Borum, J., 1996. Shallow waters and land/sea boundaries. In J rgensen, B. B. & K. Richardson (eds.), *Eutrophication in Coastal Marine Ecosystems*. American Geophysical Union, Washington: 179–203.
- Cadee, G. C., 1992. Trends in Marsdiep phytoplankton. *Netherlands Journal of Sea Research* 20: 143–149.
- Cederwall, H. & R. Elmgren, 1990. Biological effects of eutrophication in the Baltic Sea, particularly in the coastal zone. *Ambio* 19: 109–112.
- Clarke, K. R. & R. M. Warwick, 2001. *Change in Marine Communities: an Approach to Statistical Analysis and Interpretation*, 2nd Edn. Plymouth Marine Laboratory.
- Clarke, A. L., K. Weckstr m, D. J. Conley, & et al., 2006. Long-term trends in eutrophication and nutrients in the coastal zone. *Limnology and Oceanography* 51: 385–397.
- Cloern, J. E., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Cooper, S. R., 1995. Chesapeake Bay historical land use: impact on water quality and diatom communities. *Ecological Applications* 5: 703–723.
- European Communities, 2000. Directive 2000/60/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. Legislative acts and other instruments. ENV221 CODEC 513.
- Finni, T., K. Kononen, R. Olsonen & K. Wallstr m, 2001a. The history of cyanobacterial blooms in the Baltic Sea. *Ambio* 30: 172–178.
- Finni, T., S. Laurila & S. Laakkonen, 2001b. The history of eutrophication the sea area of Helsinki in the 20th century. Long-term analysis of plankton assemblages. *Ambio* 30: 264–271.
- Gasi naite, Z. R., A. C. Cardoso, A.-S. Heiskanen, P. Henriksen, P. Kauppila, I. Olenina, R. Pilkaityte, I. Purina, A. Razinkovas, S. Sagert, H. Schubert & N. Wasmund, 2005. Seasonality of coastal phytoplankton in the Baltic Sea: influence of salinity and eutrophication. *Estuarine, Coastal & Shelf Science* 65: 239–252.
- Gordon, H. R., D. K. Clark, J. W. Brown, O. B. Brown, R. H. Evans & W. W. Broenkow, 1983. Phytoplankton pigment concentrations in the Middle Atlantic Bight: comparison of ship determinations and CZCS estimates. *Applied Optics* 22: 20–36.
- Guildford, S. J. & R. E. Hecky, 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: is there a common relationship? *Limnology and Oceanography* 45: 1213–1223.
- Hajdu, S., S. Pertola & H. Kuosa, 2004. *Prorocentrum minimum* (Dinophyceae) in the Baltic Sea: morphology, occurrence—a review. *Harmful Algae* 4: 471–480.
- H llfors, G., 2004. Checklist of Baltic Sea Phytoplankton Species (including some heterotrophic protistan groups). *Baltic Sea Environmental Proceedings* 95, 210 pp.
- Heil, C. A., 2005. Influence of humic, fulvic and hydrophilic acids on the growth, photosynthesis and respiration of the dinoflagellate *Prorocentrum minimum* (Pavillard) Schiller. *Harmful Algae* 4: 603–618.
- HELCOM, 1988. Guidelines for the Baltic Monitoring Programme for the third stage: Part D. Biological Determinands. *Baltic Sea Environmental Proceedings* 27D, 161 pp.
- Hillebrand, H., C.-D. D rselen, D. Kirschtel, U. Pollinger & T. Zohary, 1999. Biovolume calculation for pelagic and benthic microalgae. *Journal of Phycology* 35: 403–424.
- Hoyer, M. V., T. K. Frazer, S. K. Notestein & D. E. Canfield Jr., 2002. Nutrient, chlorophyll, and water clarity relationships in Florida’s nearshore coastal waters with comparisons to freshwater lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 1024–1031.
- Jaanus, A., 2003. Water environment of Haapsalu Bay in retrospect (1975–2000). *Proceedings of the Estonian Academy of Sciences*. *Ecology* 52: 91–111.
- Kahru, M., U. Horstmann & O. Rud, 1994. Increased cyanobacterial blooming in the Baltic Sea detected by satellites: natural fluctuation or ecosystem change? *Ambio* 23: 469–472.
- Kononen, K., 1988. Phytoplankton summer assemblages in relation to environmental factors at the entrance to the Gulf of Finland during 1972–1985. *Kieler Meeresforschungen* 6(Sonderheft): 281–294.
- Kononen, K., 1992. Dynamics of the toxic cyanobacterial blooms in the Baltic Sea. *Finnish Marine Research* 261: 1–36.
- Kuosa, H., 1988. Observations on the taxonomy and ecology of *Monoraphidium* (Chlorophyceae, Chlorococcales) and *Koliella* (Chlorophyceae, Ulotrichales) species in the Tv rminne Sea area, SW coast of Finland. *Archiv f r Protistenkunde* 135: 45–53.

- Kutser, T., 2004. Quantitative detection of chlorophyll in cyanobacterial blooms by satellite remote sensing. *Limnology and Oceanography* 49: 2179–2189.
- Lagus, A., J. Suomela, G. Weithoff, K. Heikkilä, H. Helminen & J. Sipura, 2004. Species-specific differences in phytoplankton responses to N and P enrichments and the N:P ratio in the Archipelago Sea, northern Baltic Sea. *Journal of Plankton Research* 26: 779–798.
- Lindholm, T. & C. Nummelin, 1999. Red tide of the dinoflagellate *Heterocapsa triquetra* (Dinophyta) in a ferry-mixed coastal inlet. *Hydrobiologia* 393: 245–251.
- Marshall, H. G. & R. W. Alden, 1990. A comparison of phytoplankton assemblages and environmental relationships in three estuarine rivers of the lower Chesapeake Bay. *Estuaries* 13: 287–300.
- Millie, D. F., H. W. Pearl & J. P. Hurley, 1993. Microalgal pigment assessments using high-performance liquid chromatography: a synopsis of organismal and ecological applications. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 2513–2527.
- Niemi, Å., 1979. Blue-green algal blooms and N:P ratio in the Baltic Sea. *Acta Botanica Fennica* 110: 57–61.
- Niemi, Å. & G. Hällfors, 1974. Some phytoplankton species from Baltic waters. *Memoranda Societatis pro Fauna et Flora Fennica* 49: 77–93.
- Nixdorf, B., U. Mischke & J. Rücker, 2003. Phytoplankton assemblages and steady state in deep and shallow eutrophic lakes—an approach to differentiate the habitat properties of Oscillatoriales. *Hydrobiologia* 502: 111–121.
- Olli, K. & J. Seppälä, 2001. Vertical niche separation of phytoplankton: large-scale mesocosm experiments. *Marine Ecology Progress Series* 217: 219–233.
- Olli, K., A.-S. Heiskanen & J. Seppälä, 1996. Development and fate of *Eutreptiella gymnastica* bloom in nutrient-enriched enclosures in the coastal Baltic Sea. *Journal of Plankton Research* 18: 1587–1604.
- Paerl, H. W., L. M. Valdes, B. L. Peierls, J. E. Adolf & L. W. Harding Jr., 2006. Anthropogenic and climatic influences on the eutrophication of large estuarine ecosystems. *Limnology and Oceanography* 51: 448–462.
- Painting, S. J., M. J. Devlin, S. I. Rogers, D. K. Mills, E. R. Parker & H. L. Rees, 2005. Assessing the suitability of OSPAR EcoQOs for eutrophication vs ICES criteria for England and Wales. *Marine Pollution Bulletin* 50: 1569–1584.
- Pertola, S., H. Kuosa & R. Olsonen, 2004. Is the invasion of *Prorocentrum minimum* (Dinophyceae) related to the nitrogen enrichment of the Baltic Sea? *Harmful Algae* 4: 481–492.
- Rantajärvi, E., V. Gran, S. Hällfors & R. Olsonen, 1997. Effects of environmental factors on the phytoplankton community in the Gulf of Finland—unattended high frequency measurements and multivariate analyses. *Hydrobiologia* 363: 127–139.
- Rogers, S. I. & B. Greenaway, 2005. A UK perspective on the development of marine ecosystem indicators. *Marine Pollution Bulletin* 50: 9–19.
- Smith, V. H., 2006. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51: 377–384.
- Söderström, J., 1996. The significance of observed nutrient concentrations in the discussion about nitrogen and phosphorus as limiting nutrients for the primary carbon flux in coastal water ecosystems. *Sarsia* 81: 81–96.
- Tomas, C. (ed.), 1997. *Identifying Marine Phytoplankton*. Academic Press, London.
- Turner, R. E., N. N. Rabalais, B. Fry, N. Atilla, C. S. Milan, J. M. Lee, C. Normandeau, T. A. Oswald, E. M. Swenson & D. A. Tomasko, 2006. Paleo-indicators and water quality change in the Charlotte Harbor estuary (Florida). *Limnology and Oceanography* 51: 518–533.
- Utermöhl, H., 1958. Zur Vervollkommnung der quantitativen Phytoplanktonmethodik. *Mitteilungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* 9: 1–38.
- Vuorio, K., A. Lagus, J. M. Lehtimäki, J. Suomela & H. Helminen, 2005. Phytoplankton community responses to nutrient and iron enrichment under different nitrogen to phosphorus ratios in the northern Baltic Sea. *Journal of Experimental Marine Biology and Ecology* 322: 39–52.
- Wänstrand, I. & P. Snoeijs, 2006. Phytoplankton community dynamics assessed by ships-of-opportunity sampling in the northern Baltic Sea: a comparison of HPLC pigment analysis and cell counts. *Estuarine, Coastal and Shelf Science* 66: 135–146.
- Wasmund, N., A. Andrushaitis, E. Łysiak-Pastuszek, B. Müller-Karulis, G. Nausch, T. Neumann, H. Ojaveer, I. Olenina, L. Postel & Z. Witek, 2001. Trophic status of the South-eastern Baltic Sea: a comparison of coastal and open areas. *Estuarine, Coastal and Shelf Science* 53: 849–864.

Using bio-optics to investigate the extent of coastal waters: A Swedish case study

Susanne Kratzer · Paul Tett

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 169–186.
DOI: 10.1007/s10750-009-9769-x © Springer Science+Business Media B.V. 2009

Abstract In order to develop an optical model to map the extent of coastal waters, the authors analyzed variations in bio-optical constituents and submarine optical properties along a transect from the nutrient-enriched coastal bay, Himmerfjärden, out into the open Baltic Sea. The model is a simple implementation of the “ecosystem approach,” because the optical constituents are proxies for important components of ecosystem state. Yellow substance or colored dissolved organic matter (CDOM) is often a marker for terrestrial freshwater or decay processes in the littoral zone. Phytoplankton pigments, especially chlorophyll *a*, are used as a proxy for phytoplankton

biomass that may be stimulated by fluvial or coastal inputs of anthropogenic nutrients. Suspended particulate matter (SPM) is placed in suspension by tidal or wind-wave stirring of shallow seabeds, and is therefore an indicator for physical forcing. It is the thesis of this article that such constituents, and the optical properties that they control, can be used to provide an ecological definition of the extent of the coastal zone. The spatial distribution of the observations was analyzed using a steady-state model that assumes diffusional transport of bio-optical variables along an axis perpendicular to the coast. According to the model, the resulting distribution along this axis can be described as a low-order polynomial (of order 1–3) when moving from a “source” associated with land to the open-sea “sink.” Order 1 implies conservative mixing, and the higher orders imply significant biological or chemical processes within the gradient. The analysis of the transect data confirmed that the trend of each optical component could be described well using a low-order polynomial. Multiple regression analysis was then used to weigh the contribution of each optical component to the spectral attenuation coefficient $K_d(490)$ along the transect. The results showed that in this Swedish Baltic case study, the inorganic fraction of the SPM may be used to distinguish between coastal and open-sea waters, as it showed a clear break between coastal and open-sea waters. Alternative models may be needed for coastal waters in which fronts interrupt the continuity of mixing.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from
the Second International Symposium on Research and
Management of Eutrophication in Coastal Ecosystems, 20–23
June 2006, Nyborg, Denmark

S. Kratzer (✉)
Department of Systems Ecology, Stockholm University,
106 91 Stockholm, Sweden
e-mail: susanne.kratzer@ecology.su.se

S. Kratzer
Department of Physical Geography and Quaternary
Geology, Stockholm University, 106 91 Stockholm,
Sweden

P. Tett
School of Life Sciences, Napier University, 10 Colinton
Road, Edinburgh EH10 5DT, Scotland, UK

Keywords Bio-optics · Coastal zone · Ecosystem approach · Spectral diffuse attenuation coefficient · Diffusion mixing · Optical case 2 waters

Introduction

The coastal zone forms the transition between land and the open sea. Management of the health of ecosystems in this region is important because they are the parts of the sea first impacted by pollution from land and most visible to humans, as well as providing nurseries for young fish and sites for mariculture. States define the width of the coastal zone in different ways: during early modern times, the width was, typically, 3 miles that could be commanded by a land-based, muzzle-loaded, cannon. The twentieth century saw territorial waters expand wider and fisheries limits move offshore, reaching 200 nautical miles or the outer limits of the continental slope in the 1982 UN Convention on the Law of the Sea (UNCLOS, 1982).

During the second half of the century, the quality of coastal water began to seem important, but was initially regulated through the control of discharges only, with little attention paid to the management of the coastal zone, in general. This too has begun to change.

The EU Water Framework Directive (WFD; European Communities, 2000) requires that coastal waters of the European Union be maintained at high or good ecological quality status. It defines the coastal zone as reaching from the land to “a distance of one nautical mile on the seaward side from the nearest point of the baseline from which the breadth of territorial waters is measured, extending where appropriate up to the outer limit of transitional waters.” Borja (2005) estimated that only about 20% of the continental shelf areas in Europe are covered by the Water Framework Directive.

In some countries or regions, a greater breadth than one nautical mile is assumed: for example, out to 3 miles from a headland–headland baseline in Scotland. In the United States, there are currently several definitions of coastal waters in use. The U.S. Army Corps of Engineers also defines U.S. coastal waters as extending out to a three nautical mile line, but The H. John Heinz III Center for Science, Economics, and the Environment (2002) uses a distance within

25 miles off shore to delineate coastal from open-sea waters.

All these examples show that the breadth of the coastal zone is defined in different ways, but they all have in common that the coastal zone is defined by a fixed distance to the shore. The grounds on which these widths have been set are unclear, and can be taken as largely pragmatic. However, how broad is the coastal zone in principle and which variables can be used as an indicator of coastal waters? In this article, the authors provide a case study that demonstrates how bio-optical measurements and remote sensing can be used to determine the breadth of the coastal zone on the east coast of Sweden with respect to water quality in the Baltic Sea.

Area description

The Baltic Sea may be regarded as a large fjord of the Atlantic Ocean or a large estuary with weak tides (<20 cm) and, in most places, broad shallow margins. The Baltic Sea is characterized by permanent salinity stratification with a brackish surface layer caused by the high freshwater input from rivers and more saline deep and bottom waters coming in from the North Sea. In the Baltic Proper, the halocline ranges between 40 and 70 m depth. During spring and summer, a seasonal thermocline at depths between 15 and 20 m develops in most parts of the Baltic Sea, providing another density barrier for vertical exchange (Voipio, 1981). The winter water formed during the preceding winter remains as a layer on top of the primary halocline. When the thermocline rises toward the surface, local upwelling can be observed: cold water from below the thermocline is lifted upward and eventually reaches the surface, where it replaces the well-mixed and warmer upper layer (Gidhagen, 1987). In the Baltic Sea, upwellings are usually found within 10–20 km of the coast, and sometimes spread out in finger-like filaments. Gidhagen (1987) and Krężel et al. (2005) found horizontal temperature differences of up to 10 and 14°K, respectively, within particular upwelling events in the Baltic Sea. Upwelling plays an important role for the coastal plankton communities by transporting nutrients from the deep layer to the euphotic zone with major temperature variations taking place.

In addition to the vertical density stratification, the large fluvial input from the north and the saline input of water from the North Sea also produce a strong horizontal salinity gradient across the whole of the Baltic Sea. The surface salinity decreases progressively from 6 to 8 in the Baltic Sea proper, to 5 to 6 in the Bothnian Sea, down to 2–3 in the Bothnian Bay. The salinity is therefore low compared to other seas, and the large freshwater content is associated with a high content of CDOM. Due to the low tidal range in the Baltic Sea, and the strong salinity stratification, there is relatively little resuspension of sediment. Furthermore, Milliman (2001) pointed out that Scandinavian rivers have extremely low sediment loads compared to other European rivers. This is because most rocks are old (pre-Mesozoic) and difficult to erode.

The typical long-term mean horizontal current field in the surface layers has a weak cyclonic pattern, that is, with anticlockwise rotation (Kullenberg, 1981; Stigebrandt, 2001). This leads to currents from the north along the Swedish coast, and from the south along the Finnish coast. The flow gives rise to many horizontal eddies, which are visible in satellite images (Kahru et al., 1995; Victorov, 1996). From the point of view of remote sensing, the Baltic Sea comprises optical case 2 waters (Morel & Prieur, 1977), in which CDOM and SPM play an important part in light attenuation, in addition to the part played by water itself and phytoplankton pigments.

Himmerfjärden is a fjord-like bay situated in the southern Stockholm Archipelago, just south of 60°N, opening into the Baltic Sea (Fig. 1). With a mean depth of about 17 m, Himmerfjärden is rather shallow and consists of a sequence of basins divided by several sills. The bay and its adjacent waters have been well studied for many years, in part because of concern about nutrient enrichment by urban wastewater (Engqvist, 1996; Savage et al., 2002). Due to the low freshwater input (flushing rate 0.025 d^{-1}) and the presence of several sills, Himmerfjärden has a weak circulation and, as observed generally in the Baltic Sea, there is virtually no tidal influence. Applying the CSST (Comprehensive Studies Task Team) model to Himmerfjärden, the inner basins of Himmerfjärden were shown as potentially phosphorus limited, and may be regarded as “potentially eutrophic,” despite comparatively low nutrient loading relative to their volume (Tett et al., 2003). The

area is subject to frequently occurring summer blooms of filamentous cyanobacteria, dominated by *Aphanizomenon* sp. and *Pseudanabaena limnetica* (Hajdu et al., 1997), as well as occasional surface blooms of *Nodularia spumigena*, which are often more frequent and more intense in the open Baltic Proper, where they may cover large areas that can be monitored by remote sensing (Kahru, 1997; Subramaniam et al., 2000). The filamentous cyanobacteria are able to fix nitrogen, and are limited by phosphorus in their production. Over the whole year, the total phytoplankton biomass tends to be higher inside Himmerfjärden, whereas from June to August, the total cyanobacteria biomass used to be higher outside Himmerfjärden (Hajdu et al., 1997). The recent introduction of nitrogen removal in the sewage treatment plant (STP, Fig. 1) at the head of the bay, however, has reduced this difference, and stimulated the growth of nitrogen-fixing bacteria inside the bay (Elmgren & Larsson, 2001).

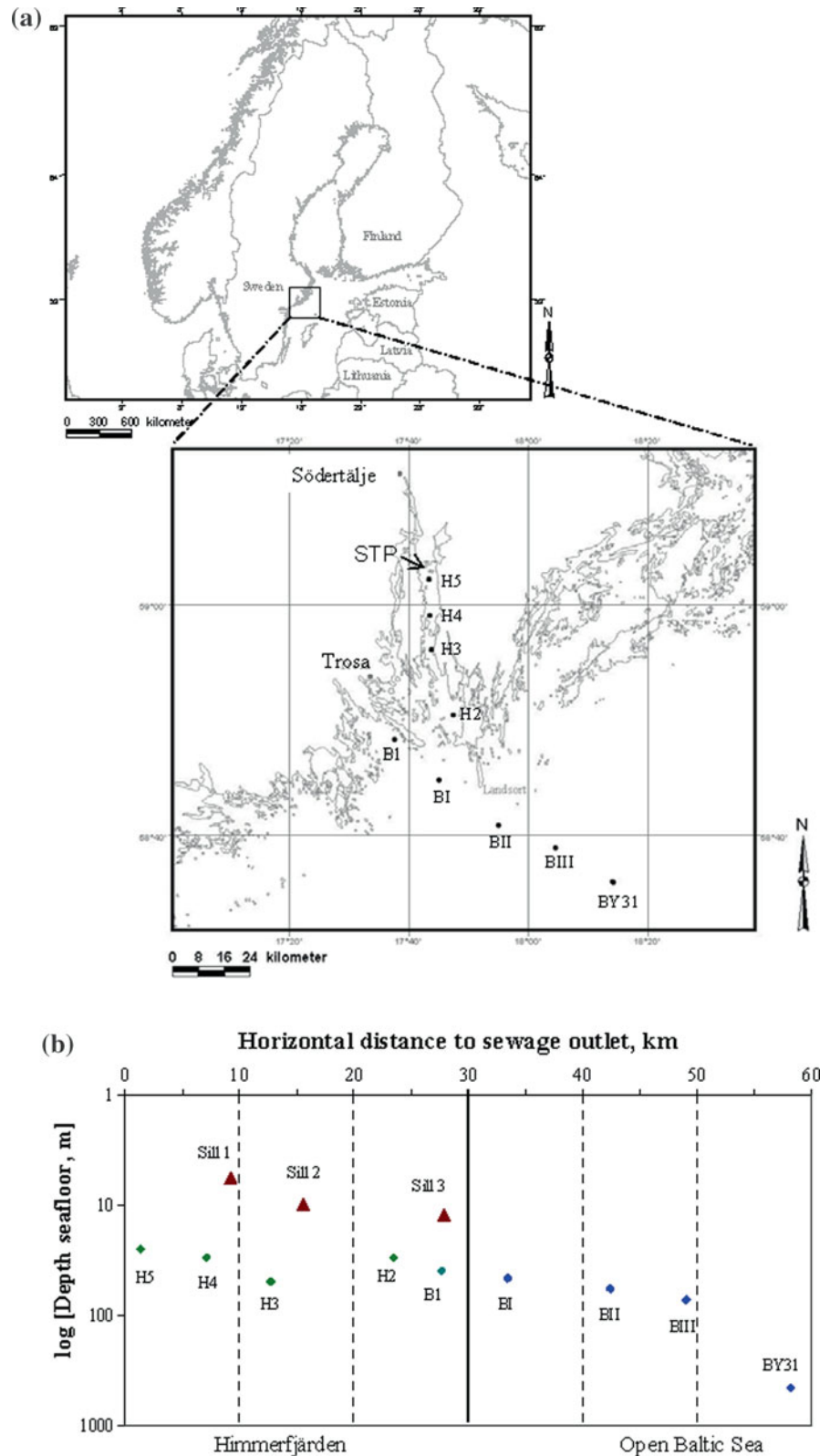
The coastal zone and bio-optical properties

Many of the pressures on water quality in the coastal zone originate on land or at the land–sea boundary, and this boundary can be treated as the source for a number of optical constituents in sea water. These constituents are the following:

- I. Yellow substance or colored dissolved organic matter (CDOM), which is often a marker for terrestrial freshwater or decay processes in the littoral zone;
- II. Phytoplankton pigments, especially chlorophyll *a*, but also carotenoids, provide proxies for phytoplankton biomass, including that which may be stimulated by fluvial or coastal inputs of anthropogenic nutrients; and
- III. Suspended particulate matter (SPM) placed in suspension by tidal or wind-wave stirring of shallow seabeds, which is, therefore, an indicator for physical forcing.

These three optical variables determine the diffuse attenuation and, therefore, the light field, and thus may influence the productivity in the sea. Hence, the spectral diffuse attenuation coefficient, $K_d(490)$, can be used to determine the light conditions for phytoplankton or phytobenthic growth. It has also been shown that $K_d(490)$ can be estimated reliably from

Fig. 1 (a) *Top*: Map of Himmerfjärden and adjacent areas. The transect stations are marked on the map. H2–H5, B1 and BY31 are standard monitoring stations. BY31 is Landsort Deep, the deepest part of the Baltic Sea (459 m). The *arrow* indicates the outlet of the Himmerfjärden Sewage Treatment Plant (STP) close to station H5. (b) *Bottom diagram*: Sea floor depth (logarithmic scale) of the transect stations plotted against horizontal distance to the outlet of the sewage treatment plant. The depths of the sills separating the different basins in Himmerfjärden are marked as *triangles*. The line 30 km off the sewage outlet marks roughly the end of coastal waters and the beginning of open-sea waters as a visual help for analyzing the diagrams



remote sensing observations in the Baltic Sea (Kratzer et al., 2003; Darecki & Stramski, 2004). Furthermore, point estimates of this attenuation coefficient can be obtained from Secchi depth (Kratzer et al., 2003; Pierson et al., 2008). Thus, $K_d(490)$ provides a link between laborious measurements of in-water properties and a variable that can, in principle, be easily mapped over wide areas and thus show the spatial extent of the coastal zone.

It is the thesis of this article that the optical components of sea water, and the optical properties that they control, can be used to map the extent of the coastal zone. This is because they are not only real and important components of marine ecosystems but may also be seen as proxies for some aspects of ecosystem state, and so have the potential to provide a measure of coastal influence that takes account of the “ecosystem approach” instead of the muzzle velocity of a cannon (i.e., instead of the more pragmatic approach based on how far a cannon ball flies). In order to synthesize the effects of these three coastal influences along an axis at right-angles to the shore, we will now look at the contributions of each component to the value of the spectral diffuse attenuation coefficient of downwelling light— $K_d(490)$.

Distribution of optical variables perpendicular to the coast

Two classes of distribution (along an axis perpendicular to the coastline) can be envisaged: continuous and discontinuous. Discontinuous cases include those where hydrodynamic processes create fronts. Shelf-break fronts limit shelf–ocean exchange (Huthnance, 1995) and may thus define the outer boundary of the coastal zone on a narrow shelf. On the broad western Scottish shelf, the haline fronts which form the boundary of the Scottish Coastal Current (Simpson & Hill, 1986) constrain the westward mixing of radio-caesium from the Irish Sea (McKay et al., 1986). Fronts separate estuarine plumes from adjacent seas (Garvine, 1986). Even in the absence of obvious fronts, there are often zones in which mixing along the onshore–offshore axis is diminished. Tett et al. (2003) used a box model that assumes complete mixing within a box representing a fjord or enclosed bay, so that the main physical control on flushing time is the limited exchange across the mouth of

these “regions of restricted exchange.” Such a model was applied to Himmerfjärden bay. Within the bay, advective circulation is weak, and water residence times have been estimated to be upward of 20 days (Engqvist, 1996), increasing with water depth and distance to the mouth of Himmerfjärden. A better model for the distributions of substances along a longitudinal transect originating in the bay would involve spatially continuous eddy-diffusive processes. Box 1 contains solutions for several models of this sort, and suggests that the relationship between a land-derived substance and distance offshore should be that of a polynomial of order 1–3. The first-order polynomial refers to conservative mixing between a nearshore source of a tracer and an offshore sink for the same tracer, and in this case, there will be no obvious break between the nearshore and offshore conditions. The higher-order polynomials result from significant biological or chemical processes along the transect, and in these cases, it may be possible to draw the boundary between coastal and offshore waters at the point where the gradient of the polynomial begins to increase. In any case, even if it is not possible or desirable to make such distinctions, the approach outlined in this mathematical treatment is likely to prove helpful in devising simple models for the coastal region and, in particular, in identifying boundary conditions for them.

Materials and methods

Observations were made along a transect from the head of Himmerfjärden to a station in the Landsort Deep in the deepest part of the Baltic Sea (Fig. 1). The line of stations shown in Fig. 1 extended from H5, near the discharge of a large STP into the Himmerfjärden, to BY31 in the Landsort Deep, with a seabed depth of 459 m. Geophysical and optical variables were measured at these stations during field campaigns in June 2001 and August 2002 (Table 1). GPS positions (GARMIN, GPSMAP, datum: WGS-84) were recorded for each station and used to calculate distances from the wastewater discharge.

The Secchi depth was measured at each transect station (Table 1) using a standard 30-cm white Secchi Disk. A water telescope (collapsible bathyscope,

Table 1 Dates and locations for all optical transect stations in 2001 and 2002 ($n = 34$)

| Date | Transect | Location |
|------------|-------------------------------|-----------------------------|
| 20-06-2001 | Himmerfjärden | H5 – H4 – H3 – H2 – B1 |
| 28-06-2001 | Open Baltic Sea | B1 – BI – BII – BIII – BY31 |
| 02-07-2001 | Himmerfjärden/Open Baltic Sea | H4 – H3 – H2 – B1 – BI |
| 04-07-2001 | Open Baltic Sea | B1 – BII – BY31 |
| 09-08-2002 | Open Baltic Sea | BY31 – BIII – BII – BI |
| 12-08-2002 | Himmerfjärden | H5 – H4 – H3 – H2 |
| 15-08-2002 | Open Baltic Sea | BY31 – BIII – BII – BI |
| 22-08-2002 | Himmerfjärden | H5 – H4 – H3 – H2 |

Nuova Rade) was used to avoid the influence of reflectance at the sea surface having an effect on the viewer's reading. The spectral attenuation coefficient $K_d(490)$ was measured with a radiometric system (TACCS, Satlantic Inc.) that included a chain of four sensors for downwelling irradiance at 490 nm ($E_d(490)$) with a 10-nm bandwidth. The sensors were fixed on a cable at 2, 4, 6, and 8 m depths. The instrument was set to record for 2 min at a rate of 1 sample per second, having first been allowed to float 10–20 m away from the boat to avoid shading. The natural logarithm of the measured downwelling irradiance was plotted against depth and the slope of the line taken as $K_d(490)$. During three transects in 2001, the absorption at 440 nm (a_{440}) was measured in situ with an AC9plus (Wetlabs, USA) at 1–2 m below the surface.

Salinity was measured using a SAIV/AS STD which was lowered to a depth of about 14 m. Surface water samples were taken by bucket and used to check surface temperature by measurement with a hand-held thermometer as well as analysis for optical constituents.

Concentrations of organic and inorganic SPM were measured in triplicate by gravimetric analysis using the method of Strickland & Parsons (1972). Kratzer (2000) showed that the gravimetric method to derive SPM had an error of 10% for 29 Baltic Sea duplicates sampled in different bottles.

For the determination of CDOM, the water was filtered through 0.2- μ m membrane filters and measured spectrophotometrically (300–800 nm) in a 10-cm optical cuvette using a Shimadzu UVPC 2401 spectrophotometer. The optical density (OD), i.e., absorbance, at 440 nm was corrected for the OD at 750 nm, and g_{440} , the absorption coefficient for CDOM at 440 nm, was derived as follows:

$$g_{440} = \ln(10) \times (\text{OD}_{440} - \text{OD}_{750})/L \text{ (m}^{-1}\text{)},$$

Kirk (1994),

where L is the path length of the cuvette in meters (in this case 0.1 m).

For the estimation of photosynthetic pigments, the spectrophotometric method was applied (Jeffrey & Humphrey, 1975; Parsons et al., 1984), using GF/F filters and extraction into 90% acetone. Chlorophyll *a* was calculated according to the trichromatic method (Parsons et al., 1984) that uses the absorption at 664, 647, and 630 nm, corrected for the reading at 750 nm to account for particle scattering. The algorithm used has shown best results when comparing spectrophotometric methods to high-performance liquid chromatography (HPLC) (Jeffrey & Welschmeyer, 1997), and is also included in the MERIS protocols (Doerffer, 2002). Total carotenoids were estimated according to Parsons et al. (1984), using the absorption at 480 and 510 nm, also corrected for the reading at 750 nm. All laboratory methods followed a standard protocol (Kratzer, 2000; Kratzer et al., 2000). Kratzer (2000) showed that the trichromatic method to derive chlorophyll had an error of 7% for 27 Baltic Sea duplicates sampled in different bottles. An international chlorophyll intercalibration exercise was coordinated by the Norwegian Institute of Water Research (NIVA) for the MERIS validation team in 2002; it included both HPLC and spectrophotometric measurements. The results of the intercalibration (Sørensen et al., 2003) showed that our spectrophotometric chlorophyll measurements of natural water samples were within 8.6% of the median value of the international group. The coefficient of variation for the results reported was 5% between the laboratories (one out of eight laboratories used the fluorometric method). There were 11 laboratories using HPLC

measurements, and the overall coefficient of variation between the laboratories was 14%.

Trend analysis—investigating the applicability of the models described in Box 1

Various types of least-squares regression were used to explore the relationships between the values of the optically active constituents and distance from the wastewater outfall, and to check the applicability of the models described in Box 1. Preliminary statistical analysis showed that when comparing the data sets from 2001 and 2002, there were no significant differences in any of the variables measured; therefore, all data were analyzed together. First-, second- and third-order polynomials were fitted using Excel, and the value of the coefficient of determination (r^2) was used to decide which order of polynomial to use.

Results

Water samples

During the field campaign in June 2001, the weather was unstable. Air temperature ranged from 12.1 to 23.1°C, and only seven out of 18 transect stations were observed under cloud-free conditions. Filamentous cyanobacteria were visible by eye, but did not form any surface accumulations during the period of investigation.

In August 2002, the weather was dominated by high pressure. Air temperature ranged from 21 to 27.3°C, and all 17 transect stations were measured under blue skies (sometimes hazy) and with calm seas. Prior to the field campaign (10–17 July 2002), there had been occurrences of cyanobacteria blooms in the open Baltic Sea area, and later during our field campaign, an increase in filamentous cyanobacteria surface accumulations was again observed. The measured salinity (average over 1–2 m depth) ranged from 5.05 to 6.11 ($n = 27$) at the transect stations in 2001 and 2002.

In most of the following diagrams, measured variables are plotted against distance from the outlet of the STP, which is treated as a significant source of terrestrial properties and thus as the nominal starting point of the transect. Figure 2 shows that surface temperature was relatively constant over each

individual transect, but increased progressively over time with the establishment of the seasonal thermocline. As shown in Figs. 3–6, most other variables showed decreasing values with offshore distance, although salinity and Secchi depth increased, and organic SPM (SPOM) showed no simple pattern. There were no obvious abrupt discontinuities in the distributions.

The results of the trend analysis are shown in Table 2. If the difference in r^2 was not significant, then the lower order polynomial was chosen to describe the trend (highlighted equations in Table 2). The results confirm the polynomials derived in Box 1 and show that in most cases, a quadratic equation gave as good a fit as a cubic equation, except for CDOM and salinity, for which a cubic equation provided a significant improvement in explained variance. Amongst significant relationships, the proportion of explained variance was lower for pigments (41–43%) than for CDOM, SPM, inorganic SPM (SPIM), and salinity (69–83%).

Relationships among the variables

Not unexpectedly, there was a considerable correlation among the variables (Table 3), the most marked including an inverse correlation between salinity and CDOM with a correlation coefficient (r) of -0.85 ($n = 26$). There were also very strong correlations

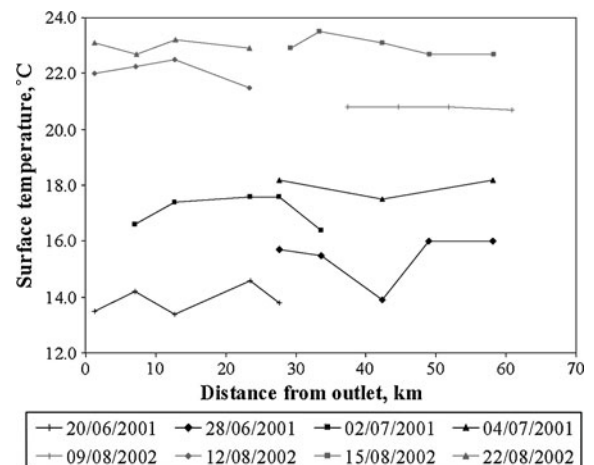


Fig. 2 Horizontal surface temperature profiles measured at the transect stations are shown in Table 1. The temperature transects did not show a difference between coastal and open sea waters. The increase in water temperature was mostly caused by the seasonal trend

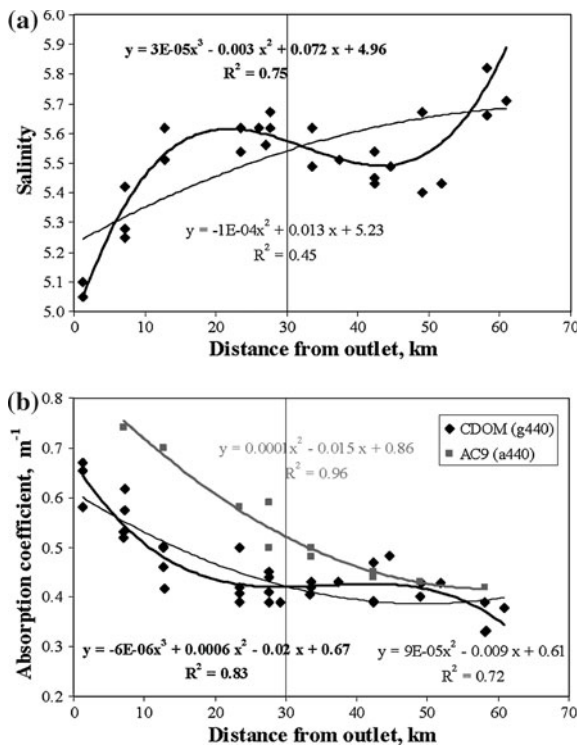


Fig. 3 Transects of **a** salinity and **b** CDOM with polynomial trendlines of second and third order; and including the total absorption at 440 nm, a_{440} , as derived from the AC9. The vertical line at 30 km marks the approximate transition from Himmerfjärden to the open Baltic Sea (between B1/Sill 3 and BI)

between $K_d(490)$ and total SPM ($r = 0.88$, $n = 34$), and between chlorophyll and the total carotenoid concentration ($r = 0.89$, $n = 34$).

In addition, there was a high correlation between reciprocal Secchi depth (z_S) and $K_d(490)$, corresponding to the relationship:

$$K_d(490) = 0.18(\pm 0.03) + 1.57(\pm 0.12) \times z_S^{-1} \quad (1)$$

Multiple regression analysis

Multiple regression analysis was used to estimate the contribution of each optical component to $K_d(490)$. For this, the diffuse attenuation coefficient was first corrected by subtracting 0.022 m^{-1} , the diffuse attenuation of water at 490 nm K_w (Smith & Baker, 1981), from each measured $K_d(490)$ value. The corrected attenuation [$K_d(490) - K_w(490)$] was then regressed against g_{440} and the concentrations of SPM and chlorophyll a (chl- a). A significant intercept was obtained for total SPM as a regression variable, but

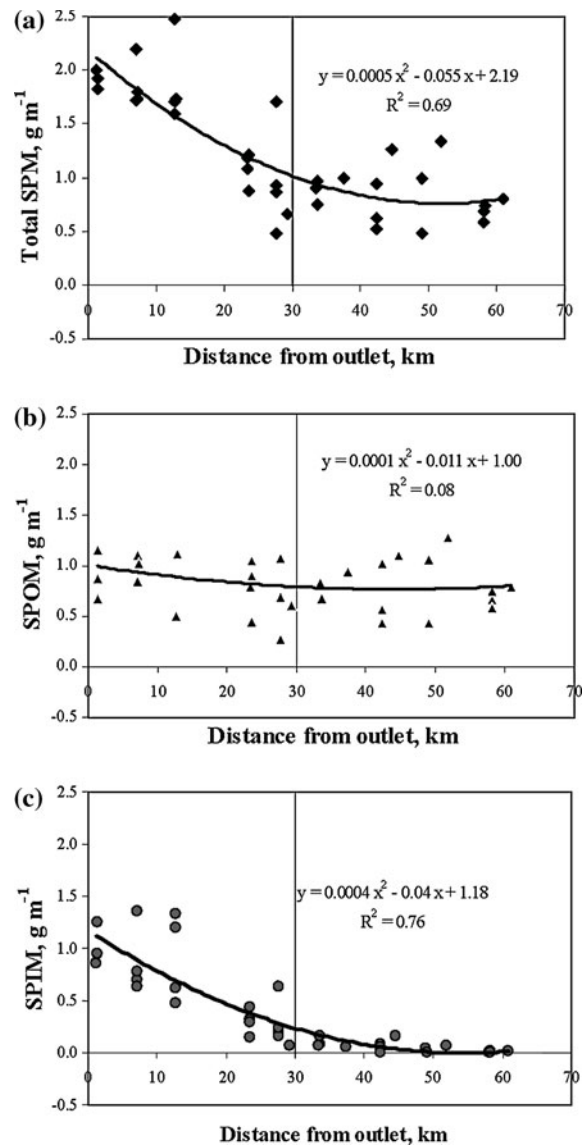


Fig. 4 Transects of **a** SPM load, **b** SPOM load, and **c** SPIM load; all with second order polynomial trendlines

not when using the inorganic fraction, SPIM, instead. The best multiple regression was

$$\begin{aligned} [K_d(490) - K_w(490)] = & -0.0013 + 0.134 \times [\text{SPIM}] \\ & + 0.818 \times g_{440} + 0.0242[\text{chl-}a] \quad (2) \\ r^2 = & 0.87, p = 0.000, n = 34 \end{aligned}$$

(Using a value of 0.016 m^{-1} for $K_w(490)$, as suggested by Mueller (2000), gave a larger negative intercept of -0.0047 , and was rejected.) Analysis of variance for the dataset showed that 49% of the variance in the multiple regression was explained by

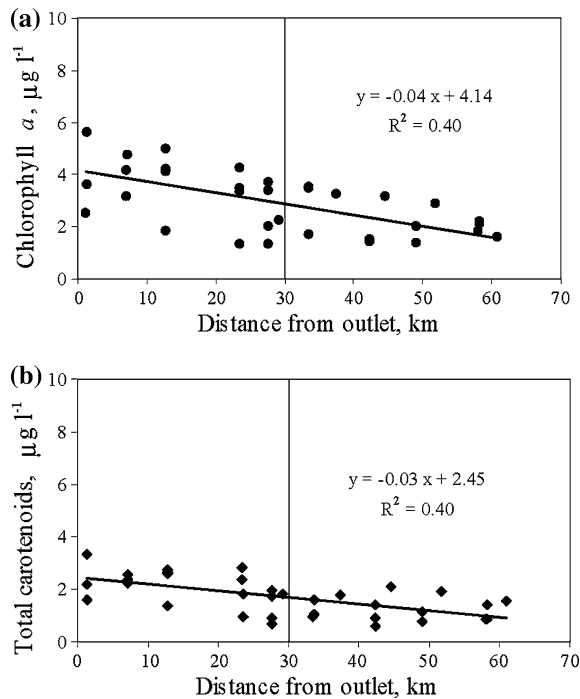


Fig. 5 Transects of **a** chlorophyll *a* concentrations and **b** total carotenoid concentrations, with second order polynomial trendlines

SPIM, 11% by CDOM, and only 2% by chlorophyll. The remainder (38%) was caused by the variability of the optical properties of the three components.

In order to test the influence of chlorophyll on CDOM concentration, we did a multiple regression analysis of g_{440} on salinity and chlorophyll. The regression of g_{440} on salinity explained 69% of the variance, and the regression including chlorophyll explained 77%; so the chlorophyll explained 8% of the variance of g_{440} , and therefore had a significant influence on CDOM.

Synthesis—a coastal attenuation model

Finally, smoothed values of CDOM g_{440} and concentrations of SPIM and chlorophyll at 5 km steps away from the STP were calculated from the polynomial regression equations in Table 2. The coefficients from the multiple regression analysis were then used to weigh the contribution of each optical component to $K_d(490)$. The result of this analysis is shown in the model in Fig. 7d. This coastal model serves as a synthesis and summary of the results of this study.

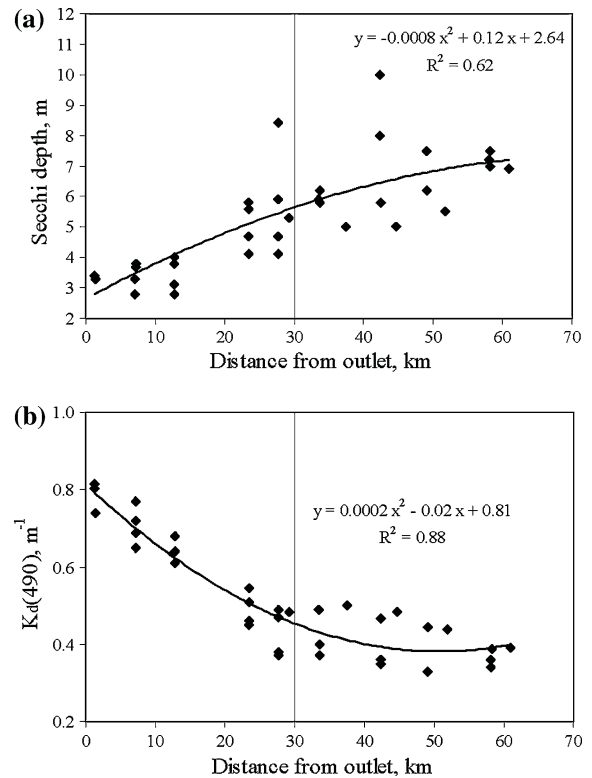


Fig. 6 Transects of **a** Secchi depth and **b** $K_d(490)$ with second order polynomial trendlines

The model shows that CDOM is optically dominant both in the coastal and the open Baltic Sea stations. SPIM is also optically important within Himmerfjärden, and influences the attenuation up to a horizontal distance of about 40–45 km from the sewage outlet, tending below 0.05 g m^{-3} at about 15–20 km off the coast. Chlorophyll makes a significant contribution to the attenuation in both Himmerfjärden and the open Baltic Sea, with slightly decreasing values away from the land.

Discussion

Observations

The study described here was carried out during the summers of two years—2001 and 2002. Figure 8a and b show the development of the seasonal stratification at station B1 in year 2001 and 2002, respectively. Although two quite different weather conditions were encountered in these years, there

Table 2 Trend analysis for geophysical/optical variables with distance to the sewage outlet

| Geophysical parameter | Order polynomial | Polynomial fit ($n = 35$); salinity: $n = 27$ | r^2 | Adjusted r^2 |
|-----------------------|------------------|--|--------------|----------------|
| Temperature | Quadratic | $y = 0.0013 x^2 - 0.0719x + 19.55$ | 0.015 | |
| Salinity | Quadratic | $y = -0.0001 x^2 + 0.013 x + 5.23$ | 0.449 | |
| | Cubic | $y = 0.00003 x^3 - 0.003 x^2 + 0.072 x + 4.96$ | 0.754 | 0.722 |
| CDOM | Linear | $y = -0.0036 x + 0.5583$ | 0.600 | |
| | Quadratic | $y = 0.00009 x^2 - 0.009 x + 0.61$ | 0.718 | |
| | Cubic | $y = -0.000006 x^3 + 0.0006 x^2 - 0.0224 x + 0.6727$ | 0.826 | 0.809 |
| SPM | Linear | $y = -0.0231 x + 1.8769$ | 0.594 | |
| | Quadratic | $y = 0.0005 x^2 - 0.055 x + 2.19$ | 0.685 | 0.666 |
| SPIM | Linear | $y = -0.0194 x + 0.9426$ | 0.678 | |
| | Quadratic | $y = 0.0004 x^2 - 0.0434 x + 1.1762$ | 0.762 | 0.747 |
| SPOM | Quadratic | $y = 0.0001 x^2 - 0.011 x + 1.00$ | 0.084 | 0.027 |
| Chl- <i>a</i> | Linear | $y = -0.0424 x + 4.1443$ | 0.400 | 0.374 |
| | Quadratic | $y = 0.0004 x^2 - 0.067 x + 4.39$ | 0.411 | |
| Carotenoids | Linear | $y = -0.0253 x + 2.4518$ | 0.398 | 0.360 |
| | Quadratic | $y = 0.0004 x^2 - 0.052 x + 2.71$ | 0.432 | |
| Secchi depth | Linear | $y = 0.0755 x + 3.1148$ | 0.602 | 0.577 |
| | Quadratic | $y = -0.0008 x^2 + 0.12 x + 2.64$ | 0.622 | |
| $K_d(490)$ | Linear | $y = -0.0069 x + 0.7154$ | 0.748 | |
| | Quadratic | $y = 0.0002 x^2 - 0.02 x + 0.81$ | 0.877 | 0.869 |

The polynomial regression with the highest coefficient of determination (r^2) was chosen as the best fit (in bold). The lower-order polynomial was chosen as best fit in the case there was no significant difference in r^2 compared to the higher-order polynomial (in bold)

Table 3 Correlation matrix derived from all transects during summer 2001 and 2002 ($n = 34$) showing the correlation between all optical variables

| Mean r | | Salinity | CDOM | SPM | SPIM | SPOM | Chl- <i>a</i> | Carotenoids | Secchi depth |
|------------|-----------------|----------|-------|-------|-------|-------|---------------|-------------|-----------------|
| 0.54 | Salinity | r | | | | | | | |
| | | p | | | | | | | |
| 0.66 | CDOM | r | -0.85 | | | | | | |
| | | p | 0.000 | | | | | | |
| 0.74 | SPM | r | | 0.71 | | | | | |
| | | p | | 0.000 | | | | | |
| 0.64 | SPIM | r | | 0.68 | 0.89 | | | | |
| | | p | | 0.000 | 0.000 | | | | |
| 0.46 | SPOM | r | | | 0.63 | | | | |
| | | p | | | 0.000 | | | | |
| 0.58 | Chl- <i>a</i> | r | | 0.54 | 0.59 | | | | |
| | | p | | 0.000 | 0.000 | | | | |
| 0.63 | Carotenoids | r | | 0.53 | 0.68 | 0.52 | 0.89 | | |
| | | p | | 0.001 | 0.000 | 0.001 | 0.000 | | |
| 0.70 | Secchi depth | r | | -0.65 | -0.88 | -0.76 | -0.59 | -0.61 | -0.72 |
| | | p | | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 | 0.000 |
| 0.76 | $K_d(490)$ | r | -0.70 | 0.86 | 0.88 | 0.82 | 0.51 | 0.64 | 0.72 |
| | | p | 0.000 | 0.000 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 |

Key to significance level (Fowler et al., 1998):

| | |
|---------------------------|----------------------------|
| $p > 0.0011$ | No significant correlation |
| $0.40 \geq r \leq 0.69$ | A modest correlation |
| $0.70 \geq r \leq 0.89$ | A strong correlation |
| $0.90 \geq r \leq 1.00$ | A very strong correlation |

were no systematic differences between years in any of the optical variables measured.

The value of an optical variable in our model (Fig. 7) can be seen as dependent on (1) the distance offshore, (2) the time of year, and (3) all other factors, treated as noise. An advantage in studying only the summer season was to reduce not only the variability caused by seasonal variations but also the noise, since summer conditions are generally more stable due to the seasonal stratification. However, given that we have proposed a general model for coastal influence but only tested it against a limited dataset, it is necessary to consider whether the optical properties and constituents measured in this particular area of the Baltic Sea during the summers 2001 and 2002 can be considered in any way typical of other summers, other seasons, and other areas in the Baltic Sea.

Table 4 shows that our mean chlorophyll values measured in summer compare well with the HELCOM observations in the Baltic Proper over the period 1994–1998 (HELCOM, 2002). Kratzer et al. (1998) measured optical constituents near Ar at the northern coast of Gotland from 23 July to 5 August 1998 (Table 4). The water here may be considered typical of the northwestern Baltic. The SPM and

chlorophyll values observed during 2001/2002 were similar to those at Ar in 1998, whereas the 2001/2002 CDOM values were higher. This can be explained as a result of interannual movements in the north–south trending salinity gradient, and perhaps also by changes in the degree of stratification: in summer 1998, the seasonal thermocline developed only very weakly (Miguel Rodriguez Medina, Baltic Nest Institute, pers. comm., 2007), and few surface accumulations of cyanobacteria were observed (Hansson, 2006).

The seasonal variation of optical components is related to variations in terrestrial runoff. There are two major seasons with high runoff in the Baltic Sea area: in spring, during the thawing period; and in summer, due to the peak of precipitation (Voipio, 1981). An increase in precipitation leads to an increase in terrestrial runoff, which leads to a decrease in salinity (Meier & Kauker, 2003), and an increase in nutrients and humic and fulvic acids. Kowalczyk et al. (2005) investigated CDOM attenuation measured during 39 cruises from 1993 to 2001. They also observed higher values in the coastal areas compared to the open Baltic Sea, and observed higher values in summer than in the autumn/winter period. When comparing the summer to the spring

Fig. 7 Best polynomial fit for the concentration of **a** SPIM; **b** CDOM (yellow substance); **c** chlorophyll; and **d** result of the coastal/diffusion model with stacked contributions of each optical component to $K_d(490)$ assuming a polynomial decline of optical constituents in relation to the source (land). The *vertical dotted line* marks the mouth of the Himmerjärden. The *dashed lines* show confidence limits for the regressions in (a)–(c). They include at least 50% of possible regressions or up to 68%, if the errors in the y-axis variables are normally distributed with constant variance (*graphs* produced in Matlab)

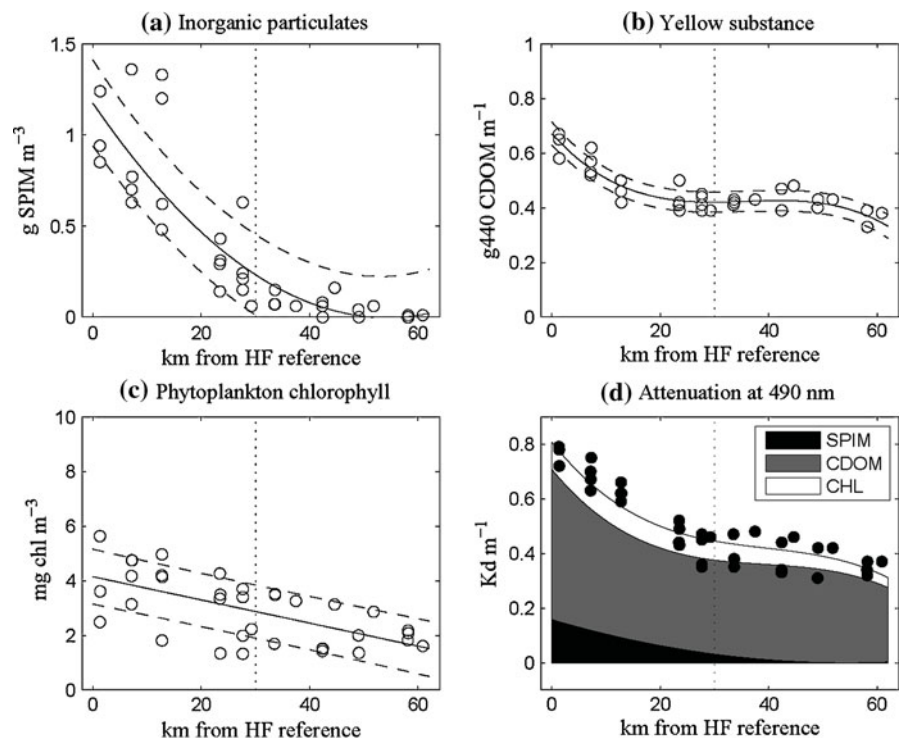


Table 4 Comparison of optical variables in the Baltic Sea by season and geographic area

| Area | Season | Mean | SD | Source | Time measured |
|---|---------------|------|------|-------------------------|-----------------------|
| Chl-<i>a</i> ($\mu\text{g l}^{-1}$) | | | | | |
| Baltic Proper | Spring | 3.28 | 4.35 | HELCOM (2002) | 1994–1998 |
| | Summer | 2.51 | 1.25 | HELCOM (2002) | 1994–1998 |
| | Autumn | 2.07 | 1.28 | HELCOM (2002) | 1994–1998 |
| | Winter | 0.53 | 0.47 | HELCOM (2002) | 1994–1998 |
| Coastal, all stns. | Summer | 3.74 | 2.23 | Fig. 5a | Summers 2001/2002 |
| Open Baltic, all stns. | Summer | 2.25 | 0.77 | Fig. 5a | Summers 2001/2002 |
| Landsort Deep | Summer | 2.15 | 0.29 | Fig. 5a | Summers 2001/2002 |
| Ar, northern Gotland | Summer | 2.90 | 0.59 | Kratzer et al. (1998) | 23 July–5 August 1998 |
| SPM (g m^{-3}) | | | | | |
| Coastal, all stns. | Summer | 1.43 | 0.59 | Fig. 4a | Summers 2001/2002 |
| Open Baltic, all stns. | Summer | 0.95 | 0.22 | Fig. 4a | Summers 2001/2002 |
| Landsort Deep | Summer | 0.73 | 0.13 | Fig. 4a | Summers 2001/2002 |
| Ar, northern Gotland | Summer | 0.85 | 0.25 | Kratzer et al. (1998) | 23 July–5 August 1998 |
| CDOM, g_{440} | | | | | |
| Coastal, all stns. | Summer | 0.48 | 0.10 | Fig. 3b | Summers 2001/2002 |
| Open Baltic, all stns. | Summer | 0.40 | 0.04 | Fig. 3b | Summers 2001/2002 |
| Landsort Deep | Summer | 0.35 | 0.03 | Fig. 3b | Summers 2001/2002 |
| Ar, northern Gotland | Summer | 0.27 | 0.03 | Kratzer et al. (1998) | 23 July–5 August 1998 |
| Open sea | Spring | 0.34 | | Kowalczuk et al. (2005) | 1993–2004 |
| | Summer | 0.32 | | Kowalczuk et al. (2005) | 1993–2004 |
| | Autumn/Winter | 0.27 | | Kowalczuk et al. (2005) | 1993–2004 |
| Pomeranian Bay | Spring | 0.38 | | Kowalczuk et al. (2005) | 1993–2004 |
| | Summer | 0.44 | | Kowalczuk et al. (2005) | 1993–2004 |

data, it depended on the region whether the summer or the spring values were somewhat higher (Table 4). Unpublished data (Kratzer) suggest that all optical components have higher values in the Landsort Deep in spring than in summer.

Correlations between optical components

It is not surprising that there were significant correlations between some of the optical in-water constituents (Table 3). All would be expected to be correlated insofar as they relate to the common factor of distance and hence obey the diffusion model. However, there are particular links between some of the optical parameters that may explain the correlations: CDOM tends to be high in freshwater and this is likely to explain its good (inverse) correlation with salinity; photosynthetic pigments (chlorophyll and carotenoids) occur together in phytoplankters and are therefore correlated, and chlorophyll might co-occur

with suspended particulate organic matter (SPOM) in phytoplankton.

The existence of correlations between variables causes some difficulties in interpreting the results from the multiple regression analysis, and may have biased estimates of the coefficients in Eq. 2. As a check, the coefficients were converted to standardized beam absorption coefficients (a_{490}) by multiplying them by the value of the mean cosine of downward irradiance (μ_0). If one assumes scatter to be negligibly small compared to absorption, then the following relationship may be assumed: $a_{490} = \mu_0 \times K_d(490)$, (Kirk 1994). This is a good assumption at least for open Baltic Sea waters where we have little scatter due to inorganic particles, and where the attenuation is dominated by CDOM absorption. This is exemplified in Fig. 3b, which shows the total absorption of optical constituents (corrected for the absorption by water) at 440 nm measured by the AC9 compared to the CDOM absorption at 440 nm (*Note*: the CDOM measurements

were taken just below the surface, whereas the AC9 integrated the absorption between 1 and 2 m).

Assuming a mean cosine μ_0 of about 0.86 in these waters, the resulting value for chlorophyll (derived from Eq. 2) is $0.021 \text{ m}^2 (\text{mg chl})^{-1}$, which is an estimate of $a^*_{\text{PH}}(490)$, the chlorophyll-specific absorption coefficient at 490 nm. Although realistic in general terms, it is lower than the range of $0.028\text{--}0.043 \text{ m}^2 (\text{mg chl})^{-1}$ (mean = $0.035 \text{ m}^2 (\text{mg chl})^{-1}$; $n = 12$) reported by Kratzer (2000) for phytoplankton in open Baltic waters in summer 1998, calculated using the filter pad method (Kishino et al., 1985; Cleveland & Weidemann, 1993).

In contrast, the mean cosine-corrected CDOM coefficient in Eq. 2 is 0.70. This coefficient is a dimensionless ratio, $g_{490}/g_{440} = a(\text{CDOM}, 490)/a(\text{CDOM}, 440)$ as the CDOM attenuation was measured at 440 nm, whereas the diffuse attenuation was measured at 490 nm. This ratio is higher than the ratio derived from our measured CDOM spectra, which we estimated to be typically about 0.48. This means that the multiple regression analysis leads to an overestimation of CDOM and an underestimation of chlorophyll attenuation. This problem could be solved mathematically by putting certain constraints on the range of values for each of the variables when performing the multiple regression analysis.

Model fit

To the extent that they are fitted by simple polynomials, the results for most optical constituents and the synthetic variable $K_d(490)$ (Figs. 3–7) are compatible with the diffusion model(s) of Box 1. In these models, the terrestrial end of the transect is a notional source of material and the marine end is a notional sink; the marine sink may be literal in the case of SPIM, which falls close to zero in the water over the Landsort Deep (at BY31). As mentioned in the Results section, SPM and CDOM clearly followed the polynomial decline predicted by the diffusive model, but chlorophyll a had a less obvious trend. This may be caused by the dominance of filamentous cyanobacteria during summer: as these are able to fix nitrogen, they are limited by phosphorus (Larsson et al., 2001), and will therefore follow the pattern of phosphate distribution. In the case of, for example, coastal upwelling, and the subsequent replenishment of phosphorus in the surface mixed layer, the biomass of nitrogen-fixing

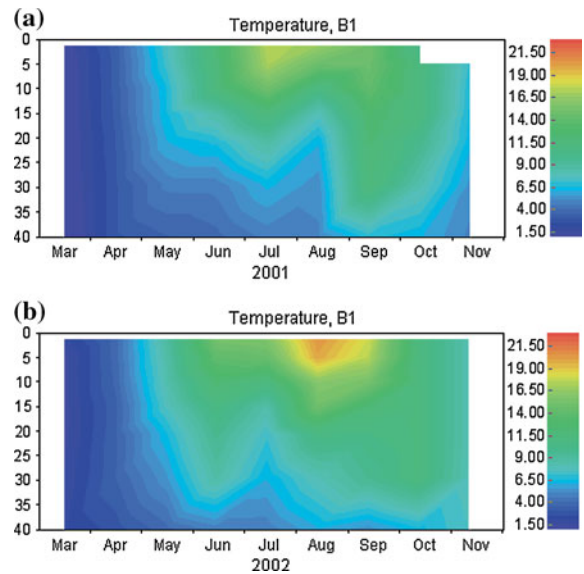


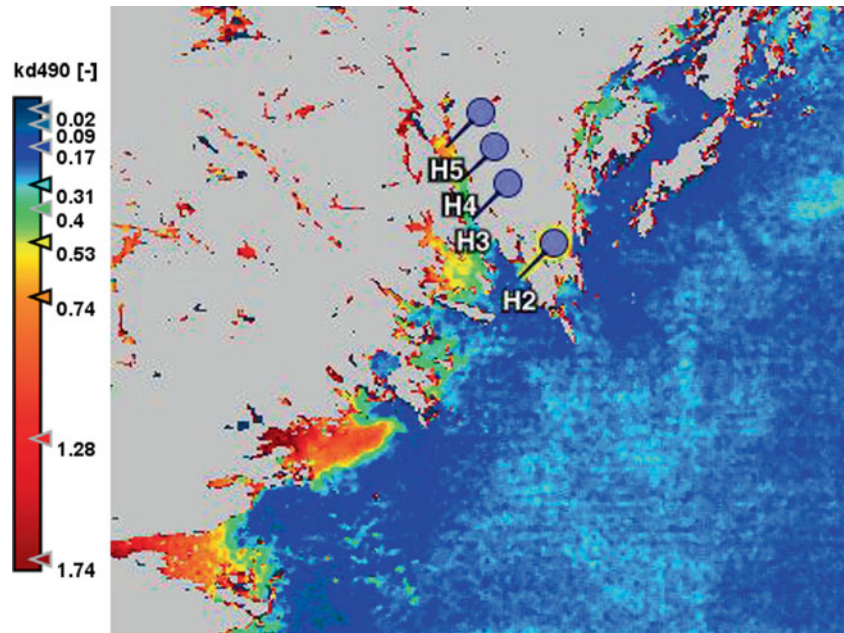
Fig. 8 Development of the seasonal stratification in the top 40 m at Station B1 during the years **a** 2001 and **b** 2002. Images produced by Miguel Rodriguez Medina (using NEST) based on data delivered by the Swedish Meteorological and Hydrological Institute (SMHI) and the Swedish Marine Research Centre (SMF) within the framework of the Swedish Environmental Monitoring Programme (1–2 week sampling)

cyanobacteria may start to increase locally offshore, and therefore the smooth gradient—which is described by a polynomial—may be disturbed.

Upwelling might also lead to an abrupt change in water mass and therefore in the underwater optical characteristics. The presence of upwelling is shown by colder water filaments in some of the sea-surface temperature images in Fig. 9. Gidhagen (1987), Lehmann et al. (2002), and Myrberg & Andrejev (2003) found that the width-scale of upwellings (perpendicular to the coast) in the Baltic Sea is typically in the range of 5–20 km. The occurrence of upwelling in our research area is shown by colder water filaments in some of the sea-surface temperature images shown in Fig. 9, but as shown in the trend lines of Figs. 3–7, there were no abrupt changes in optical constituents in the upwelling areas.

Nevertheless, there is the difficulty of recognizing a discontinuous distribution, or distinguishing between it and a continuous distribution, in the presence of variation. By “continuous” we mean a distribution that is truly described by the simple polynomials dealt with in this article and with shore-normal transports dominated by eddy diffusion at a rate that is either constant

Fig. 10 MERIS $K_d(490)$ full resolution image (300 m resolution) from 22 August 2002 processed using BEAM and the $K_d(490)$ algorithm derived from sea-truthing data (Kratzer et al., 2008)



Box 1 Diffusional model equations

In this one-dimensional, steady-state model, x is the horizontal distance in meters along a transect with origin offshore and terminus at $x = l$. Y is a generalized variable (amount m^{-3}), with boundary conditions of Y_0 at $x = 0$ and Y_l at $x = l$. The defining equation is

$$\frac{\partial Y}{\partial t} = 0 = -K_h \cdot \frac{\partial^2 Y}{\partial x^2} + \beta_Y,$$

where K_h is a horizontal mixing coefficient ($m^2 d^{-1}$) and β_Y is the total of biogeochemical and ecological sources and sinks (amount $m^{-3} d^{-1}$) at a point on the transect. Solutions depend on assumptions about K_h and β_Y , and it is convenient to write

$$\frac{\partial^2 Y}{\partial x^2} = 2 \cdot k_0 \cdot (1 + a_1 \cdot x + a_2 \cdot x^2 \dots), \text{ where } k_0 = \frac{\beta_Y(x=0)}{2K_h(x=0)}$$

and the power series allows for spatial variability in rates. A general solution is

$$Y = k_0 \cdot x^2 \cdot \left(1 + \frac{a_1 \cdot x}{3} + \frac{a_2 \cdot x^2}{12} \dots\right) + b \cdot \frac{x}{l} + Y_0 \text{ where } b = (Y_l - Y_0) - k_0 \cdot l^2 \cdot \left(1 + \frac{a_1 \cdot l}{3} + \frac{a_2 \cdot l^2}{12} + \dots\right).$$

The simplest solutions, which neglect all but the zeroth and first-order (a_1) terms of the series, are

Case 1: $a_1 = 0$ and the substance described by Y is conservative, so that $\beta_Y = 0$:

$$Y = (Y_l - Y_0) \cdot \frac{x}{l} + Y_0.$$

Case 2: $a_1 = 0$, and the ratio of β_Y and K_h is positive and constant:

$$Y = \frac{\beta_Y}{2K_h} \cdot x^2 + b_2 \cdot \frac{x}{l} + Y_0, \text{ where } b_2 = (Y_l - Y_0) - \frac{\beta_Y}{2K_h} \cdot l^2.$$

Case 3: a_1 is positive. This would be true where eddy mixing is scale dependent (Okubo, 1974) and so increases offshore (i.e., with $(l - x)$), whereas the non-conservative term β_Y increases as nutrients or stirring increase toward the shore:

$$Y = k_0 \cdot \left(\frac{a_1}{3} \cdot x^3 + x^2\right) + b_3 \cdot \frac{x}{l} + Y_0, \text{ where } b_3 = (Y_l - Y_0) - k_0 \cdot l^2 \cdot \left(1 + \frac{a_1}{3} \cdot l\right).$$

one shore-normal transect can be considered representative of a considerable shore-parallel distance (Box 1).

Conclusion

The aim of this article has been to assess the width of the coastal zone on the basis of selected ecosystem properties that are influenced by coastal or terrestrial processes. We have shown that a diffusional model fits observations from a Swedish coastal water area in summer. An essential feature of the first- and second-order polynomials that fitted chlorophyll and SPM is that they are smoothly monotonic—that is, they contain no obvious discontinuities that can be used to separate nearshore and offshore waters. The CDOM decay was third order and hence provides some evidence of a break.

One obvious place to look for a limit to the coastal influence is at the mouth of the Himmerfjärden, approximately 30 km from the STP. As shown in Fig. 7, this point corresponds to a weak inflexion in the CDOM curve and to the minimum offshore distance at which the 50% confidence limits includes zero. Alternatively, we may look further offshore, at 40–50 km from the STP and 10–20 km from the open coast. It is at this distance that SPIM and chlorophyll concentrations are statistically indistinguishable from those at the “sink” end of the transect, and here that the CDOM curve begins again to descend toward the open-sea endmember value. This is certainly further than the 1 nautical mile limit of the Water Framework Directive.

Considering the low fluvial SPM input in the northwestern Baltic Sea compared to the southern Baltic Sea (Voipio, 1981), with SPM values of up to 15 g m^{-3} (Siegel et al., 2005), one would expect coastal waters to extend even further in the southern Baltic Sea.

In the case of CDOM and salinity, the inflection point at 45–50 km away from the STP and 15–20 km off the coast may be influenced by the upwelling events shown in Fig. 9, and also to a minor extent by the Swedish coastal current that transports water masses from the Bothnian Sea rich in CDOM and lower in temperature down along the Swedish coast. Considering the trendline for CDOM, it is also possible that some of the CDOM pool may be of

biological origin and may be produced by phytoplankton (Kowalczyk et al., 2006), or in this case by cyanobacteria. However, as reported in the Results section, chlorophyll explained only a small part of the variance in CDOM (about 8%); therefore, so this explanation seems less likely.

Figure 7d shows how the attenuation coefficient is composed of three optical components: CDOM, SPM, and chlorophyll, and how their concentrations change with distance to the shore. As mentioned in the Introduction, each of the components is an indicator for an important ecosystem property, and thus $K_d(490)$ can serve as an indicator that synthesizes coastal influence as well as being a biologically important variable. Thus, it would seem to be a useful variable with which to map the extent of the coastal zone, and one that embodies the “ecosystem approach.”

A robust relationship has been established between water-leaving radiance and $K_d(490)$ (Mueller, 2000) and, to date, $K_d(490)$ has been shown to be the most reliable parameter that can be measured by remote sensing methods (Darecki & Stramski, 2004). A good relationship between $K_d(490)$ and Secchi depth in the Baltic has already been confirmed (Kratzer et al., 2003), and Kratzer et al. (2008) have developed a new $K_d(490)$ algorithm to monitor coastal areas in the northwestern Baltic Sea using MERIS data. Thus, there are old and new tools at hand to apply this approach to estimating the width of the coastal zone.

Acknowledgements This work was funded by the Swedish National Space Board, the EU FP5 project Oceanographic Applications to Eutrophication in Regions of Restricted Exchange (OAERRE), the EU FP6 project Science and Policy Integration for Coastal System Assessment (SPICOSA), and by the Swedish MISTRA program under RESE 5 (Remote sensing for the environment—Methods for detection of changes in aquatic ecosystems and monitoring of algal blooms). Thanks to the Swedish Wallenberg Foundation for an expensive equipment grant. Thanks to Stefanie Hirsch, Roberta Mistretta, Antonia Sandman, and Charlotte Sahlin for their help in the field and the laboratory, to Henrik Lindh (SMHI) and the staff of the Askö Laboratory for support during fieldwork. Many thanks to Miho Ishii and Roberta Mistretta for processing the AVHRR data, and to Bertil Håkansson for his support. Thanks to Paul Sweeny from the U.S. Environmental Protection Agency for information on coastal zone definitions in the USA. Special thanks to Roland Doerffer, Anders Engqvist, Ragnar Elmgren, and Ulf Larsson for discussions and useful comments to the manuscript, and to Miguel Rodriguez Medina for preparing the images of temperature profiles for 2001 and 2002.

References

- Borja, A., 2005. The European water framework directive: a challenge for nearshore, coastal and continental shelf research. *Continental Shelf Research* 25: 1768–1783.
- Cifuentes, L. A., L. E. Schemel & J. H. Sharp, 1990. Qualitative and numerical analyses of the effects of river inflow variations on mixing diagrams in estuaries. *Estuarine, Coastal and Shelf Science* 30: 411–427.
- Cleveland, J. S. & A. D. Weidemann, 1993. Quantifying absorption by aquatic particles: a multiple scattering correction for glass-fibre filters. *Limnology and Oceanography* 38: 1321–1327.
- Darecki, M. & D. Stramski, 2004. An evaluation of MODIS and SeaWiFS bio-optical algorithms in the Baltic Sea. *Remote Sensing Environment* 89: 326–350.
- Doerffer, R., 2002. Protocols for the validation of MERIS water products. European Space Agency, Doc. No. PO-TN-MEL-GS-0043.
- Elmgren, R. & U. Larsson, 2001. Eutrophication in the Baltic Sea area: integrated coastal management issues. In von Bodungen, B. & R. K. Turner (eds), *Science and Integrated Coastal Management*. Dahlem University Press, Berlin: 15–35.
- Engqvist, A., 1996. Long-term nutrient balances in eutrophication of the Himmerfjärden estuary. *Estuarine, Coastal and Shelf Science* 42: 483–507.
- European Communities, 2000. Water Framework Directive. Directive 2000/60/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 of the European Communities L* 327: 1–73.
- Fowler, J., L. Cohen & P. Jarvis, 1998. *Practical Statistics for Field Biology*. John Wiley & Sons, Chichester.
- Garvine, R. W., 1986. The role of brackish plumes in open shelf waters. In Skreslet, S. (ed.), *The Role of Freshwater Outflow in Coastal Marine Ecosystems*. Springer-Verlag, Berlin: 47–65.
- Gidhagen, L., 1987. Coastal upwelling in the Baltic Sea – Satellite and in situ measurements of sea-surface temperatures indicating coastal upwelling. *Estuarine Coastal and Shelf Science* 24: 449–462.
- Hajdu, S., U. Larsson & K. Skärland, 1997. Chapter 9: Växtplankton (Phytoplankton). In Elmgren, R. & U. Larsson (eds), *Himmerfjärden*. Naturvårdsverket Förlag, Stockholm, Rapport 4565.
- Hansson, M., 2006. Cyanobakterieblomningar i Östersjön, resultat från satellitövervakningen 1997–2005. SMHI Rapport 82.
- HELCOM, 2002. Environment of the Baltic Sea Area, Baltic Sea Environment Proceedings No. 82B. Helsinki Commission – Baltic Marine Environment Protection Commission.
- Huthnance, J. M., 1995. Circulation, exchange and water masses at the ocean margin: the role of physical processes at the shelf edge. *Progress in Oceanography* 35: 353–431.
- Jeffrey, S. W. & G. F. Humphrey, 1975. New spectrophotometric equation for determining chlorophyll a, b, c1 and c2. *Biochemie und Physiologie der Pflanzen* 167: 194–204.
- Jeffrey, S. W. & N. A. Welschmeyer, 1997. Appendix F: spectrophotometric and fluorometric equations in common use in oceanography. In Jeffrey, S. W., R. F. C. Mantoura & S. W. Wright (eds), *Phytoplankton Pigments in Oceanography*. Monographs on Oceanographic Methodology. UNESCO Publishing, Berlin: 597–615.
- Kahru, M., 1997. Using satellites to monitor large-scale environmental change: a case study of cyanobacteria blooms in the Baltic Sea. In Kahru, M. & C. W. Brown (eds), *Monitoring Algal Blooms – New Techniques for Detecting Large-Scale Environmental Change*. Springer-Verlag, Berlin: 43–61.
- Kahru, M., B. Håkansson & O. Rud, 1995. Distribution of sea-surface temperature fronts in the Baltic Sea as derived from satellite imagery. *Continental Shelf Research* 15: 663–679.
- Kirk, J. T. O., 1994. *Light and Photosynthesis in Aquatic Ecosystems*, 2nd ed. Cambridge University Press, Cambridge.
- Kishino, M., M. Takahashi, N. Okami & S. Ichimura, 1985. Estimation of the spectral absorption coefficients of phytoplankton in the sea. *Bulletin of Marine Science* 37: 634–642.
- Kowalczyk, P., J. Olszewski, M. Darecki & S. Kaczmarek, 2005. Empirical relationships between coloured dissolved organic matter (CDOM) absorption and apparent optical properties in Baltic Sea waters. *International Journal of Remote Sensing* 26: 345–370.
- Kowalczyk, P., C. A. Stedmon & S. Markager, 2006. Modeling absorption by CDOM in the Baltic Sea from season, salinity and chlorophyll. *Marine Chemistry* 101: 1–11.
- Kratzer, S., 2000. Bio-optical studies of coastal waters. PhD thesis. School of Ocean Sciences, University of Wales, Bangor.
- Kratzer, S., P. Land & N. Strömbeck, 1998. An optical in-water model for the Baltic Sea. *Ocean Optics XIV Conference Papers*, Vol. 2. New Insights from Ocean Color.
- Kratzer, S., D. G. Bowers & P. Tett, 2000. Seasonal changes in colour ratios and optically active constituents in the optical Case-2 waters of the Menai Strait, North Wales. *International Journal of Remote Sensing* 21: 2225–2246.
- Kratzer, S., B. Håkansson & C. Sahlin, 2003. Assessing Secchi and photic zone depth in the Baltic Sea from satellite data. *Ambio* 32: 577–585.
- Kratzer, S., C. Brockmann & G. Moore, 2008. Using MERIS full resolution data to monitor coastal waters – A case study from Himmerfjärden, a fjord-like bay in the north-western Baltic Sea. *Remote Sensing Environment* 112(5): 2284–2300.
- Kreżel, A., M. Ostrowski & M. Szymelfenig, 2005. Sea surface temperature distribution during upwelling along the Polish Baltic coast. *Oceanologia* 47: 415–432.
- Kullenberg, G., 1981. Physical oceanography. In Voipio, A. (ed.), *The Baltic Sea*. Elsevier Oceanography Series 30, Amsterdam.
- Larsson, U., S. Hajdu, J. Walve & R. Elmgren, 2001. Estimating Baltic nitrogen fixation from the summer increase in upper mixed layer total nitrogen. *Limnology and Oceanography* 46: 811–820.

- Lehmann, A., W. Krauss & H.-H. Hinrichsen, 2002. Effects of remote and local atmospheric forcing on circulation and upwelling in the Baltic Sea. *Tellus* 54A: 299–316.
- McKay, W. A., M. S. Baxter, D. J. Ellett & D. T. Meldrum, 1986. Radiocaesium and circulation patterns west of Scotland. *Journal of Environmental Radioactivity* 4: 205–232.
- Meier, H. E. M. & F. Kauker, 2003. Sensitivity of the Baltic Sea salinity to the freshwater supply. *Climate Research* 24: 231–242.
- Milliman, J. D., 2001. Delivery and fate of fluvial water and sediment to the sea: a marine geologist's view of European rivers. *Scientia Marina* 65: 121–132.
- Morel, A. & L. Prieur, 1977. Analysis of variations in ocean colour. *Limnology and Oceanography* 22: 709–722.
- Mueller, L. J., 2000. SeaWiFS algorithm for the diffuse attenuation coefficient, $K(490)$, using water-leaving radiances at 490 and 555 nm. In Hooker S. B. & E. R. Firestone (eds), *SeaWiFS Postlaunch Calibration and Validation Analyses, Part 3*, 11. NASA Goddard Space Flight Center, Greenbelt, MD: 24–27.
- Myrberg, K. & O. Andrejev, 2003. Main upwelling regions in the Baltic Sea – A statistical analysis based on three-dimensional modeling. *Boreal Environment Research* 8: 97–112.
- Okubo, A., 1974. Some speculations on oceanic diffusion diagrams. *Rapport et Procès-Verbaux des Réunions du Conseil International pour l'Exploration de la Mer* 167: 77–85.
- Parsons, T. R., Y. Maita & C. M. Lalli, 1984. *A Manual of Chemical and Biological Methods for Seawater Analysis*. Pergamon Press.
- Pierson, D., S. Kratzer, N. Strömbeck & B. Håkansson, 2008. Relationship between the attenuation of downwelling irradiance at 490 nm with the attenuation of PAR (400 nm–700 nm) in the Baltic Sea. *Remote Sensing of Environment* 112(3): 668–680.
- Savage, C., R. Elmgren & U. Larsson, 2002. Effects of sewage-derived nutrients on an estuarine macrobenthic community. *Marine Ecology Progress Series* 243: 67–82.
- Siegel, H., M. Gerth, G. Tschersich, T. Ohde & T. Heene, 2005. Ocean colour remote sensing relevant water constituents and optical properties of the Baltic Sea. *International Journal of Remote Sensing* 26: 315–330.
- Simpson, J. H. & A. E. Hill, 1986. The Scottish coastal current. In Skreslet, S. (ed.), *The Role of Freshwater Outflow in Coastal Marine Ecosystems*, Vol. G7. Springer-Verlag, Berlin: 195–204.
- Smith, R. C. & K. S. Baker, 1981. Optical properties of the clearest natural waters (200–800 nm). *Applied Optics* 20: 177–184.
- Sørensen, K., M. Grung & R. Röttgers, 2003. An intercomparison of in vitro chlorophyll-a determinations. *Proceedings of the MERIS Cal/Val Meeting at ESRIN, Frascati, Italy, 10–11 December*.
- Stigebrandt, A., 2001. Chapter 2: physical oceanography of the Baltic Sea. In Wulff, F., L. Rahm & P. Larsson (eds), *A System Analysis of the Baltic Sea*. Springer Verlag, Berlin: 19–74.
- Strickland, J. H. D. & T. R. Parsons, 1972. A practical handbook of sea-water analysis. *Bulletin Journal of the Fisheries Research Board of Canada* 167: 185–203.
- Subramaniam, A., S. Kratzer, E. J. Carpenter & E. Soderback, 2000. Remote sensing and optical in-water measurements of a cyanobacteria bloom in the Baltic Sea. *Proceedings of the Sixth International Conference on Remote Sensing for Marine and Coastal Environments* 1: 57–64.
- Tett, P., L. Gilpin, H. Svendsen, C. P. Erlandsson, U. Larsson, S. Kratzer, E. Foulland, C. Janzen, J.-Y. Lee, C. Grenz, A. Newton, J. G. Ferreira, T. Fernandes & S. Scory, 2003. Eutrophication and some European waters of restricted exchange. *Continental Shelf Research* 23: 1635–1671.
- The H. John Heinz III Center for Science, Economics, and the Environment, 2002. *The State of the Nation's Ecosystems: Measuring the Lands, Waters, and Living Resources of the United States*. Cambridge University Press, Cambridge.
- UNCLOS, 1982. *United Nations Convention on the Law of the Sea*, 10 December 1982.
- Victorov, S. V., 1996. *Regional Satellite Oceanography*. Taylor & Francis, London.
- Voipio, A., 1981. *The Baltic Sea*. Elsevier Science Publishers, Amsterdam.

Identification of the coastal zone of the central and eastern Gulf of Finland by numerical modeling, measurements, and remote sensing of chlorophyll *a*

Gennadi Lessin · Viktoria Ossipova ·
Inga Lips · Urmas Raudsepp

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 187–198.
DOI: 10.1007/s10750-009-9770-4 © Springer Science+Business Media B.V. 2009

Abstract A combination of numerical modeling results with measurement and satellite imagery data was used during the biologically active period for the determination of the coastal zone extent in the central and eastern Gulf of Finland. Adopting the approach that the coastal zone can be identified by the spatial distribution of biotic parameters, spatial variations and gradients of chlorophyll *a* (chl-*a*) concentrations were analyzed. The results showed that chl-*a* concentrations vary in a wide range over the biologically active period. During heavy blooms, the coastal zone may appear occasionally and depend on the spatial distribution of the bloom. On average, clear limits of the coastal zone could be defined for the central and eastern Gulf of Finland. In the central Gulf of Finland, water and material exchange are rather intensive, and the coastal zone is narrower than in the eastern Gulf. In the easternmost part of the Gulf of Finland, chl-*a* concentrations were permanently high in an area of about 100 km width due to the discharge of the Neva

River. The study has shown that gradients of chl-*a* spatial distribution can be applied for determining limits of the coastal zone extent. The standardized gradient of zero is shown to be a threshold separating the coastal zone (standardized gradients > 0) from the open sea (standardized gradients < 0).

Keywords Coastal zone · Numerical modeling · Remote sensing · Chlorophyll *a* · Eutrophication · Gulf of Finland

Introduction

The coastal sea is a primary recipient of nutrients and suspended matter from rivers and other land-based sources. It is also influenced by a number of hydrodynamic processes, for example, currents and upwelling. The complexity and diversity of physical and biological processes may result in sharp gradients of biological and chemical parameters between the coastal zone and the open sea. This makes the coastal sea very important for marine management. Eutrophication, i.e., the increase of nutrient loading, of the Baltic Sea is more obvious in the coastal zone, especially in semi-enclosed bays, lagoons, archipelagoes, and estuaries (Telesh, 2004). Successful measures for eutrophication control require identification of the coastal zone extent.

A variety of criteria have been used for the definition of the coastal zone. These definitions are

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from
the Second International Symposium on Research and
Management of Eutrophication in Coastal Ecosystems, 20–23
June 2006, Nyborg, Denmark

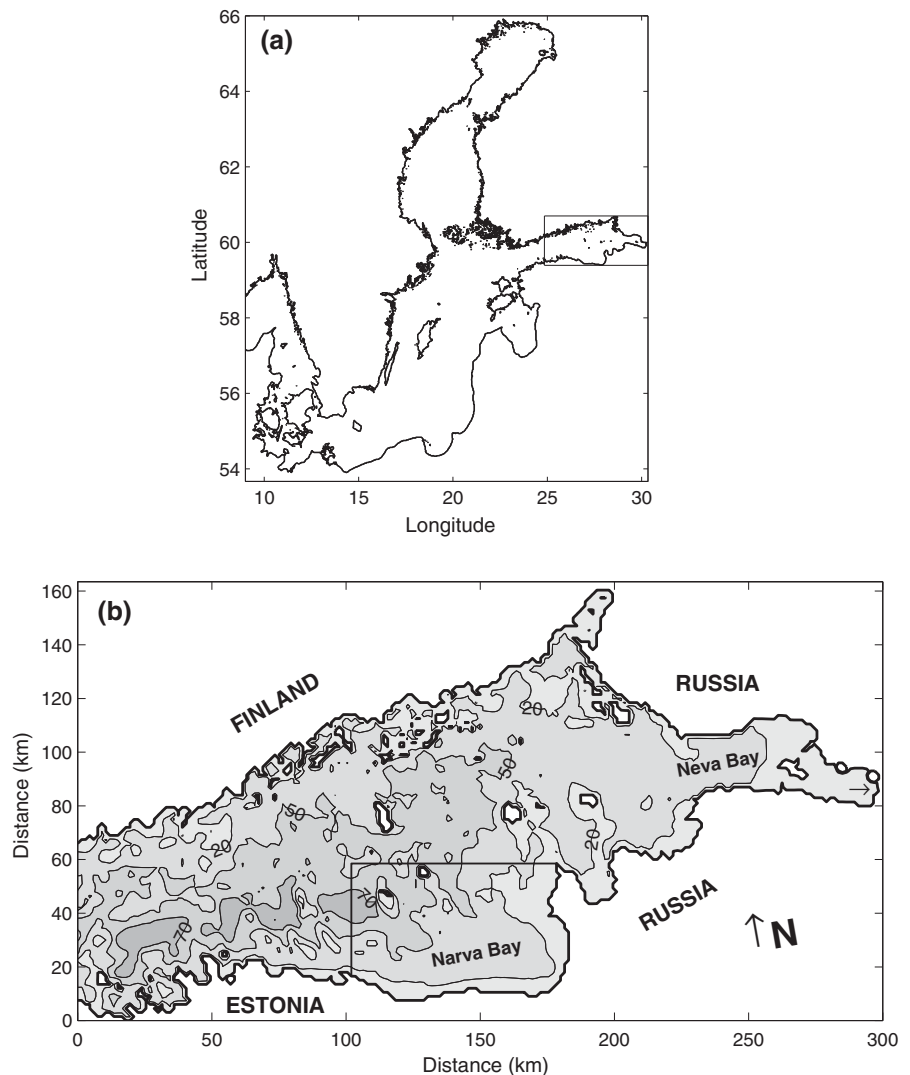
G. Lessin (✉) · V. Ossipova · I. Lips · U. Raudsepp
Marine Systems Institute, Tallinn University of
Technology, Akadeemia tee 21b, 12618 Tallinn, Estonia
e-mail: gennadi@phys.sea.ee

based on geographical extent (European Communities, 2000), geomorphology of the coastal margin (Gazeau et al., 2004), or the extent of river influence (Artioli et al., 2005). However, from the marine management point of view, it is more appropriate to define the coastal zone in terms of biotic distributions and processes, because these are not always aligned with the abiotic structure of nearshore areas (Gibbs et al., 2006).

This study aims at identifying the coastal zone and determining its extent in the central and eastern parts of the Gulf of Finland during the biologically active period by combining the results of numerical modeling with measurements and satellite imagery data. Spatial

variations and gradients of chl-*a* concentration were analyzed following the approach that the coastal zone can be identified from the spatial distribution and magnitude of gradients of biotic parameters (Lessin & Raudsepp, 2007). It is hypothesized that chl-*a* concentrations are higher in the coastal sea compared with the open sea and that this pattern persists over a prolonged period. The Gulf of Finland is an elongated estuary (Fig. 1a) of the Baltic Sea with a mean depth of 37 m and a maximum depth of 123 m. The western part of the Gulf is directly connected to the Baltic Proper. The eastern part receives the largest single freshwater inflow to the Baltic Sea—the Neva River (Alenius et al., 1998).

Fig. 1 The Baltic Sea area (a) and the model domain covering the central and eastern parts of the Gulf of Finland, with the *x*- and *y*-axes showing the distance (km) from the modeling domain origin in the southern Gulf of Finland (b). The small arrow points at the Neva River mouth. Limits of Narva Bay defined according to Piirsoo et al. (1992) are shown



Materials and methods

Numerical model

The ecohydrodynamic three-dimensional model MIKE 3 (DHI Water and Environment, 2001) was applied for the central and eastern parts of the Gulf of Finland (Fig. 1b). The basic equations of the hydrodynamic model consist of a mass conservation equation for compressible fluid, non-linear momentum equations in the three main directions, a conservation equation for salinity and temperature, and the equation of state of the seawater (Rasmussen, 1993). The simulations were performed with a hydrostatic model version. The Smagorinsky formulation was used for horizontal eddy viscosity and the $k - \epsilon$ formulation was used for the vertical turbulent closure model (Burchard et al., 1998; Rasmussen et al., 1999). The main forcing factors included wind stress and heat exchange at the surface, prescribed sea level at the open boundary, and freshwater inflow by rivers. The ecological model includes 11 interdependent state variables: phytoplankton carbon (PC), nitrogen and phosphorus; chl-*a*; zooplankton; detritus carbon, nitrogen and phosphorus; inorganic nitrogen (IN); inorganic phosphorus (IP); and dissolved oxygen. Concentrations of total nitrogen (TN) and total phosphorus (TP) are also calculated. The model implements two phytoplankton groups: diatoms and green algae, which appear in consecutive order in the model. The ecological model describes the relation between available inorganic nutrients and the subsequent phytoplankton growth. The nutrient supply depends on the land-based load and the transport into the area through model open boundaries. The simulation time covered the biologically active period from April 1 to September 30, 2001. The model spatial resolution was 1,500 m horizontally and 2 m vertically, except for the upper layer which was 3 m thick. Model results were stored at 6 h intervals.

The initial temperature (T) and salinity (S) fields were prepared based on a limited number of TS-casts. The initial temperature distribution was assumed to be horizontally uniform while keeping vertical stratification. The salinity values were horizontally interpolated to a model grid using a bilinear interpolation method. The initial concentrations of PC, chl-*a*, and zooplankton were derived from a very limited number of measurements, which only allowed the use of

homogeneous concentrations for those variables. The initial concentrations of 0.2 mg l^{-1} were taken for PC and 0.01 mg l^{-1} for chl-*a*. The initial distributions of IN, IP, detritus carbon, detritus nitrogen, detritus phosphorus, and dissolved oxygen were prepared based on a limited amount of measurement data. The data were interpolated onto the model grid using objective analysis (Lessin & Raudsepp, 2006).

The boundary conditions for ecological variables were prepared based on data from three or four locations, where samples were collected at standard depth. The measurements were performed once per month in April, May, and September and twice per month in June, July, and August. The measured values were interpolated onto a model grid on the open boundary. An accurate description of the open boundary conditions is of great importance for the correct performance of the model. Figure 2 shows a validation of model results (PC, chl-*a*, TN, TP) at station F3 situated in the vicinity of the open boundary. Although measurement data were very scarce, the comparison gives an indication of a satisfactory description of the open boundary conditions.

A thorough model description and setup as well as model validation results for Narva Bay, in the south-eastern Gulf of Finland, are given by Lessin & Raudsepp (2006) and Lessin et al. (2007).

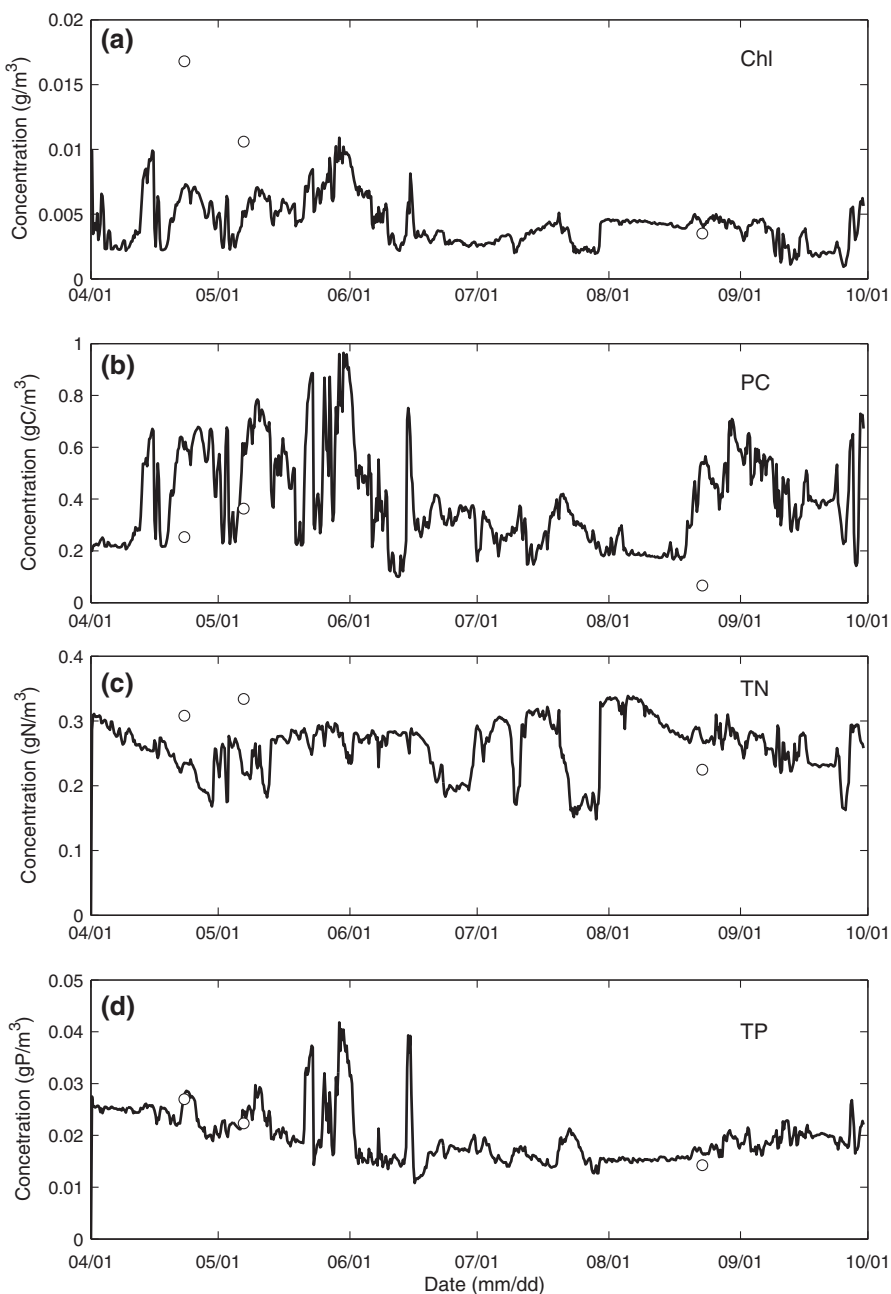
Measurement data

Measurement data originated from the automatic ferry-box system on the ferry operating between Tallinn and Helsinki (Kanoshina et al., 2003). The water intake and sampling were located at 4–5 m depth and data were collected once per week from April to November 2001. Fluorescence was recorded every 10 s (corresponding to a horizontal resolution of approximately 200 m) during the crossing between Tallinn and Helsinki. During each crossing, water samples were collected from nine sampling locations for laboratory analyses of chl-*a*. The fluorescence values were converted to chl-*a* using a linear regression relationship for each crossing separately.

Satellite imagery

Moderate resolution (1 km/pixel) MODIS/Aqua Level 2 images of chl-*a* for the years 2002 and 2003 were obtained from the OceanColor website

Fig. 2 Modeled (*line*) and observed (*dots*) values for chl-*a* (a) and phytoplankton carbon (b) integrated over the upper 1–10 m layer and surface total nitrogen (c) and total phosphorus (d) at station F3 in the vicinity of model open boundary



(<http://oceancolor.gsfc.nasa.gov/>). Due to frequent cloud cover, good quality satellite images for the Gulf of Finland are rare. In total, 40 images that cover the biologically active period were chosen for the analysis. The bio-optical MODIS algorithms with their present standard parameterization are inappropriate for applications in the Baltic Sea because the standard algorithm (MODIS OC3) usually tends to overestimate

chlorophyll concentration, especially during heavy cyanobacterial blooms (Reinart & Kutser, 2006). Therefore, the regional Baltic version of the MODIS chlor_a_2 algorithm (Darecki & Stramski, 2004) was applied for retrieving the chl-*a* concentrations. During image processing, the shallow water and land flags were used for reducing their influence on the chl-*a* concentration in the coastal zone.

Results

The model results on chl-*a* distribution in the surface layer were averaged over the period from April 1 to September 30, which spans the biologically active period in the Gulf of Finland (Fig. 3). Chl-*a* concentrations were relatively high, both close to the coast and offshore in the western part of the model domain (corresponding to the central Gulf of Finland). Along the Estonian coast, the area of high chl-*a* concentration becomes narrower toward the east. In the central part of the model domain (the eastern Gulf of Finland), a distinctive chlorophyll-rich water belt was formed along the southern coast of the Gulf. Chl-*a* concentrations in the central and eastern Gulf

decreased rapidly toward offshore, where a vast area of lower chl-*a* was found. An area of increased chl-*a* concentration was also seen along the Finnish coast. Chl-*a* values started to increase in the eastern part of the model domain and were the highest in the Neva Bay due to the discharge of the Neva River.

Because chl-*a* concentrations are considerably higher during the spring bloom period than during the rest of the year, model results were averaged over the period from June 15 to September 30 to exclude the effect of the spring bloom. Mean chl-*a* distribution for the summer period showed a similar pattern, although concentrations were lower (not shown).

For four transects across the Gulf of Finland (Fig. 4), modeled chl-*a* concentrations averaged over

Fig. 3 Modeled average surface chl-*a* concentrations (mg l^{-1})

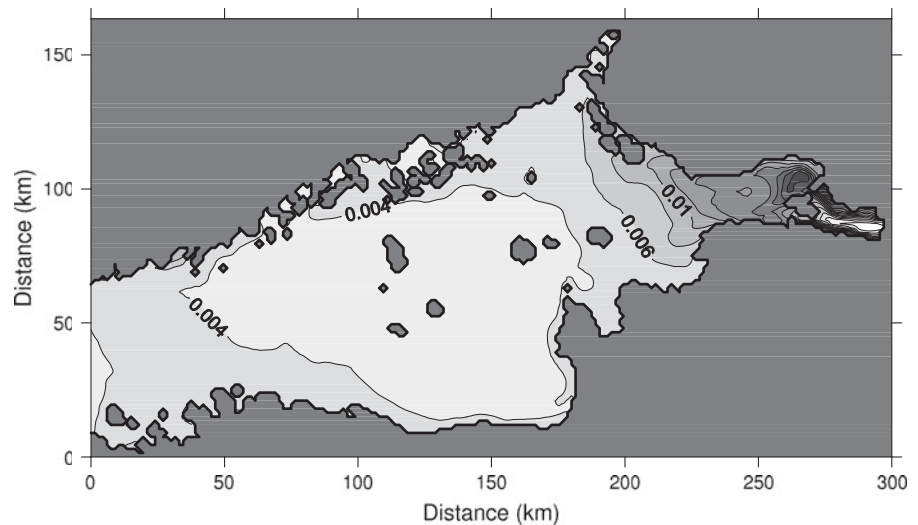
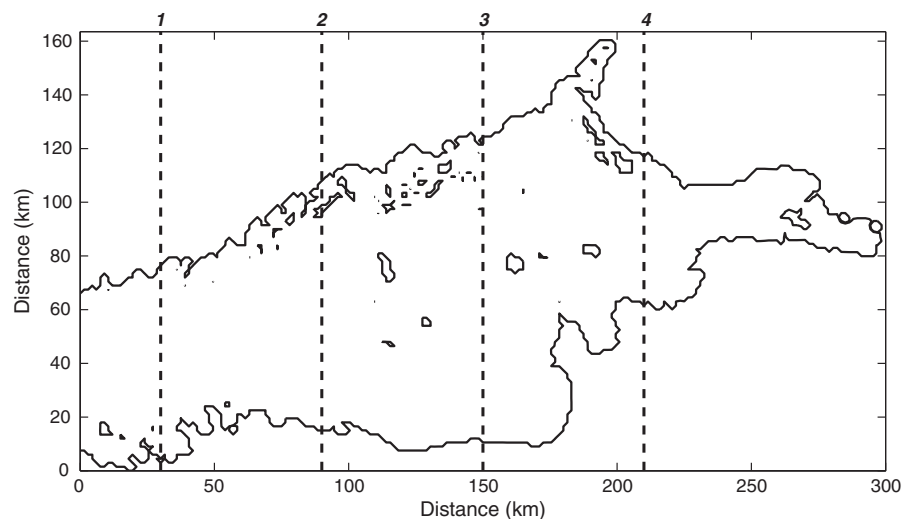


Fig. 4 Location of the transects used for the analysis of modeled extent of the coastal zone



the whole period and the summer period are shown in Fig. 5. High chl-*a* concentrations at the coast and a rather rapid decrease to the open sea values can be seen at the southern coast of the Gulf of Finland on transects 2 and 3 and at the northern coast on transect 1. The latter distribution almost disappears during the summer period. The width of the coastal zone could be estimated at about 5–8 km. The chl-*a* distribution on transect 4 is considerably different. In general, the concentrations were higher in the northern part than in the southern part. The highest concentrations were found offshore, about 12 km from the northern coast with a local minimum in the central part. Model results showed that distribution of chl-*a* at this transect followed the outflow pattern of the Neva River.

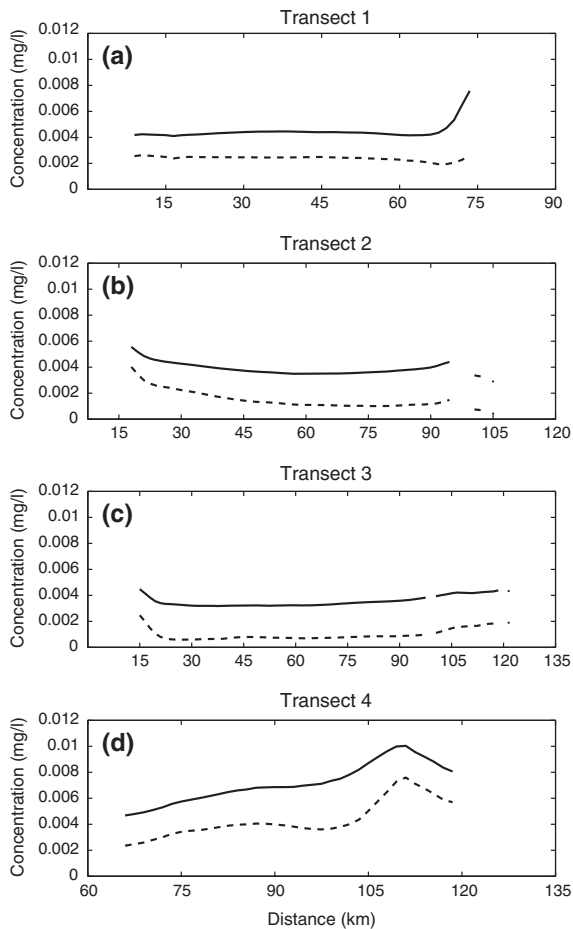


Fig. 5 Mean-modeled concentrations (mg l^{-1}) of chl-*a* along transects 1 (a), 2 (b), 3 (c), and 4 (d). Solid lines show the entire modeled period; dashed lines show the summer period. The x-axis shows the distance (km) from the modeling domain origin in the southern Gulf of Finland

The temporally averaged distribution of measured chl-*a* was calculated for two periods: from April 1 to September 30 and from June 15 to September 30, 2001. The chl-*a* values on each transect were interpolated on a regular grid with respect to geographical latitude to facilitate averaging. The measurements clearly show increased chl-*a* concentrations close to both coasts of the central Gulf of Finland (Fig. 6). (Please recall that the Tallinn–Helsinki ferry crossing is situated at the model open boundary.) The chl-*a* concentration for the whole period is about $0.012\text{--}0.013 \text{ mg l}^{-1}$ close to the coast and drops to 0.01 mg l^{-1} within 2–3 km from the coast (Fig. 6a). The standard deviation increases from 0.005 mg l^{-1} at the southern coast of the Gulf of Finland to 0.01 mg l^{-1} at the northern coast. The high chl-*a* concentrations at the coast and their rapid decrease toward offshore are clearly represented when the effect of the spring bloom is excluded (Fig. 6b). Chl-*a* concentrations were about 0.01 mg l^{-1} near the shore and 0.005 mg l^{-1} in the open sea. Also, the variability of chl-*a* concentrations

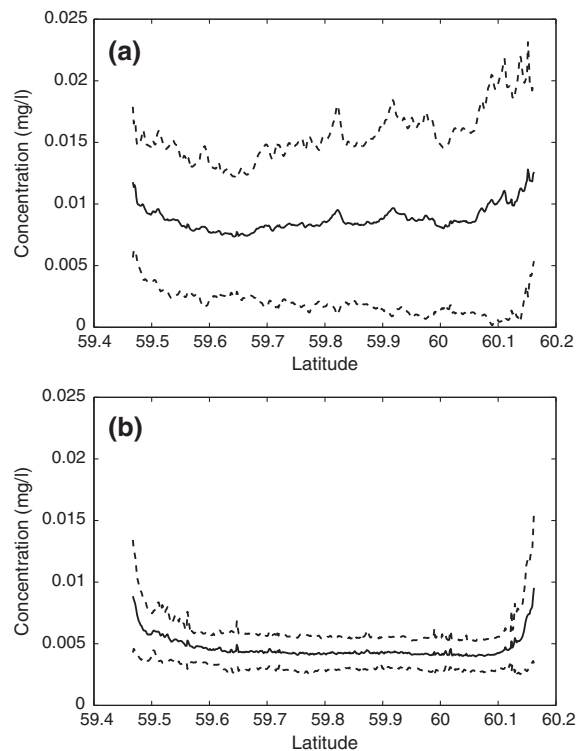


Fig. 6 Measurement data on chl-*a* concentrations (mg l^{-1}) for the entire period (a) and for the summer period (b). Solid lines show average values; dashed lines show upper and lower limits of the standard deviation

was lower offshore ($\sim 0.003 \text{ mg l}^{-1}$) than near the coasts (up to $\sim 0.013 \text{ mg l}^{-1}$).

A comparison of modeled and measured mean chl-*a* concentrations in the central Gulf of Finland indicates that modeled chl-*a* values are underestimated approximately two times. This can be explained by the fact that measurements were performed at 4–5 m depth, where chlorophyll concentrations are usually higher than near the surface (e.g., Kononen et al., 1998).

The temporal variability of the modeled surface layer chl-*a* concentration was analyzed on the four transects across the Gulf of Finland. The chl-*a* values on each transect and time instant were divided by the maximum value on that transect, as the main focus of our study was to analyze the chl-*a* distribution in the coastal area relative to the open sea, i.e., across the Gulf. In general, temporal variations of relative surface chl-*a* are different in different parts of the Gulf of Finland (Fig. 7). In the central Gulf (transect 1), a relatively high chl-*a* concentration was present in a 5–15 km belt near the Finnish coast from the second week of April until July. Later on, the concentrations were often higher in offshore than close to the coast. However, a broad area of elevated chl-*a* concentration formed near the Estonian coast in September. Moving further to the east, i.e., transects 2 and 3, chl-*a* concentrations were rather uniform across the Gulf until May. A broad area (about 20–30 km) of increased chl-*a* concentration formed at the Finnish coast, which existed until the end of August, disappearing occasionally. At the Estonian coast, a narrow zone of high chl-*a* concentration could be identified from the beginning of July. The width of this zone was about 5–10 km, except at transect 2, where it broadened up to ~ 45 km in September. Close to Neva Bay, in the eastern Gulf of Finland (transect 4), concentrations of chl-*a* were high in the northern part of the Gulf, but at some distance from the shore. The chl-*a* concentration was lower in the southern part of the Gulf of Finland compared with the northern part.

All the available satellite data for chl-*a* distribution over the study area (40 images) were averaged monthly. These data were then averaged to produce a composite image (Fig. 8). Although the modeled year is 2001, using images from 2002 and 2003 allows the results of numerical modeling to be generalized. In order to eliminate the effect of the spring bloom,

averaging of satellite data for the summer months only was performed. Although showing lower concentrations, the overall spatial distribution of chl-*a* remained similar (not shown).

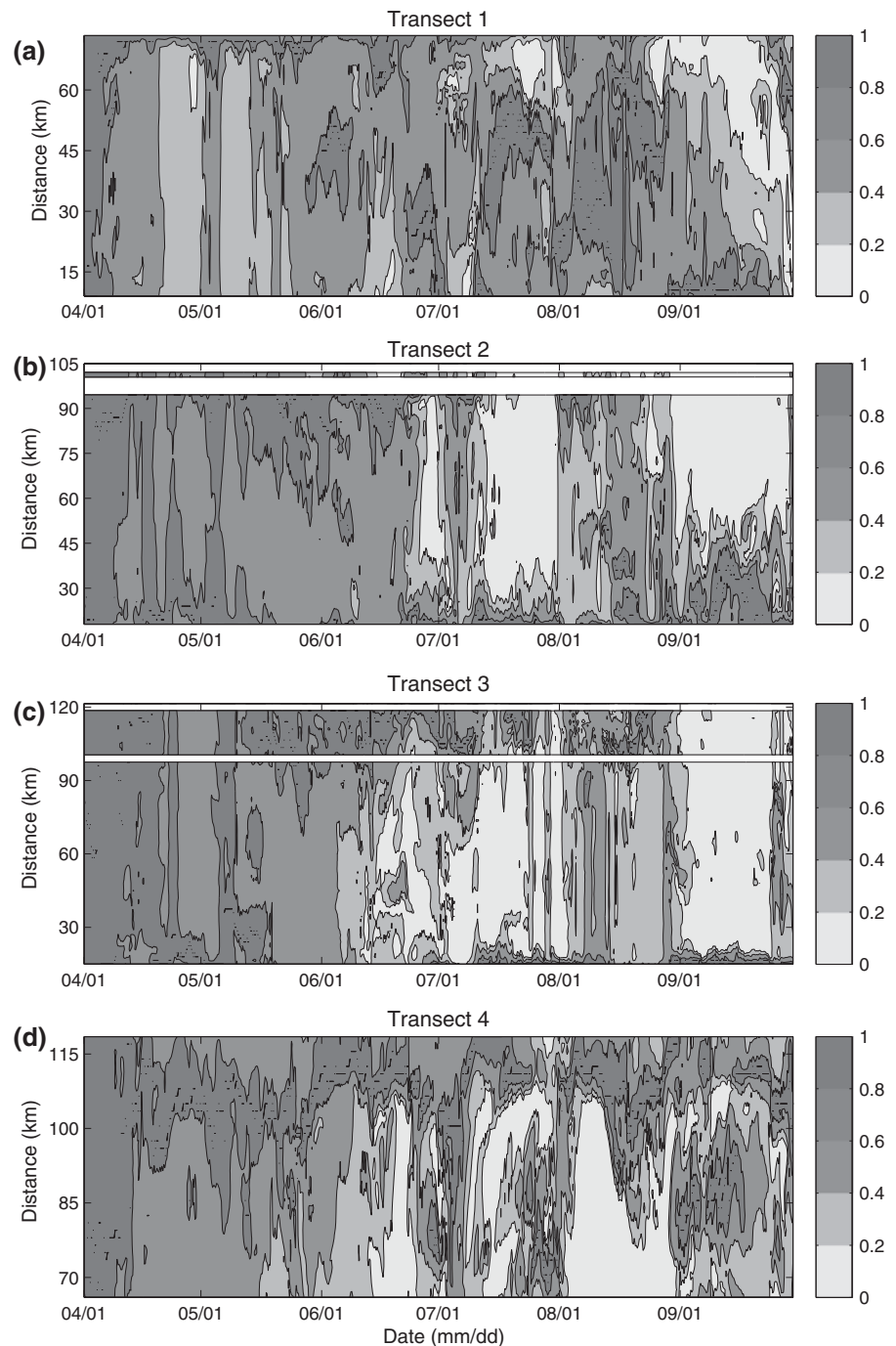
The composite image (Fig. 8) revealed a belt of increased chl-*a* concentrations along the coasts of the Gulf of Finland. This zone was narrow, but clearly discernible along the northern coast of the Gulf. Along the southern coast, a belt of higher concentrations steadily broadened from the westernmost part of the study area toward the south-eastern part (Narva Bay). Concentrations dropped rapidly toward the offshore where a region of low and uniform chl-*a* occurred. The easternmost area of the Gulf (Neva Bay) was characterized by high concentrations of chl-*a* as an effect of the Neva River discharge. The strong patchiness in chl-*a* distribution in the northeastern part of the Gulf can also be attributed to the effect of Neva River action.

The increased chl-*a* concentrations were found along or in the vicinity of the coast both in the model results and in satellite data. Therefore, in order to discern limits of the coastal zone, a calculation of the spatial gradient of chl-*a* concentrations was applied for both mean-modeled and composite satellite data. Because the relative distribution of gradients is of particular interest for the identification of the coastal zone, standardized gradients were calculated as $g_s = (g - \bar{g}) / \sigma_g$, where g is the gradient, \bar{g} is the mean and σ_g is the standard deviation of the gradient calculated over the Gulf of Finland. Both the model (Fig. 9) and satellite (Fig. 10) data showed similar results. The area of positive gradients (presented in black in the figures) generally formed a narrow belt along the coast of the Gulf of Finland. In Neva Bay, the area of positive gradients propagated far offshore westwards. Thus, a standardized gradient of zero can serve as a threshold value delimiting the coastal zone (standardized gradients > 0) from the open Gulf (standardized gradients < 0). Moreover, the coastal zone is characterized by a rapid decline in gradient values toward the offshore where gradients were rather uniform.

Discussion

An application of chl-*a* distribution for the identification of the coastal zone shows that this zone is more a dynamic than a rigid concept. Chl-*a* concentration

Fig. 7 Modeled normalized temporal variability of chl-*a* concentrations (see key on the right side of the figure) along transects 1 (A), 2 (B), 3 (C), and 4 (D). The y-axis shows the distance (km) from the modeling domain origin in the southern Gulf of Finland



varies in a broad range over the biologically active season. In the Gulf of Finland, high chl-*a* values are present during the spring diatom bloom and summer cyanobacterial bloom. The latter shows considerable inter-annual variations (Kahru et al., 1995; Laanemets

et al., 2006). Ignoring seasonal variations to some extent, the possibility to identify the coastal zone as the area of elevated chl-*a* concentration compared to offshore regions was investigated. The results showed higher concentrations and positive standardized

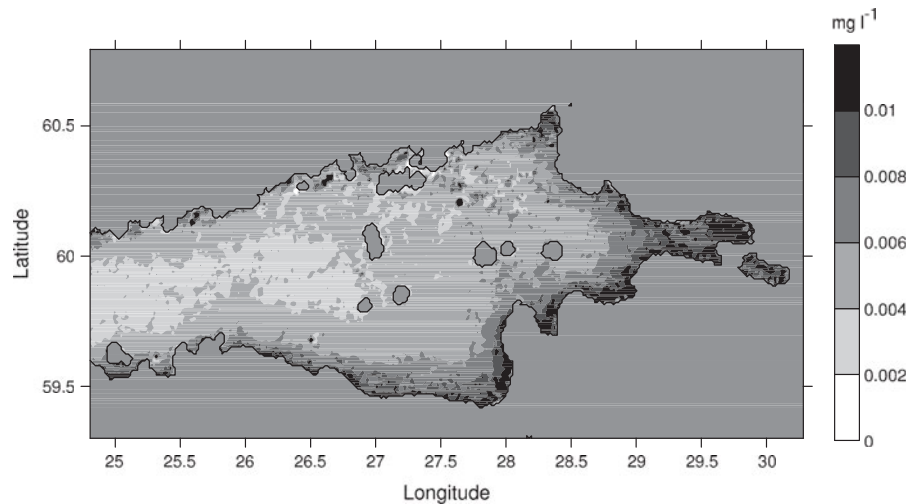


Fig. 8 Composite satellite image of chl-*a* distribution (mg l^{-1}) in the Gulf of Finland

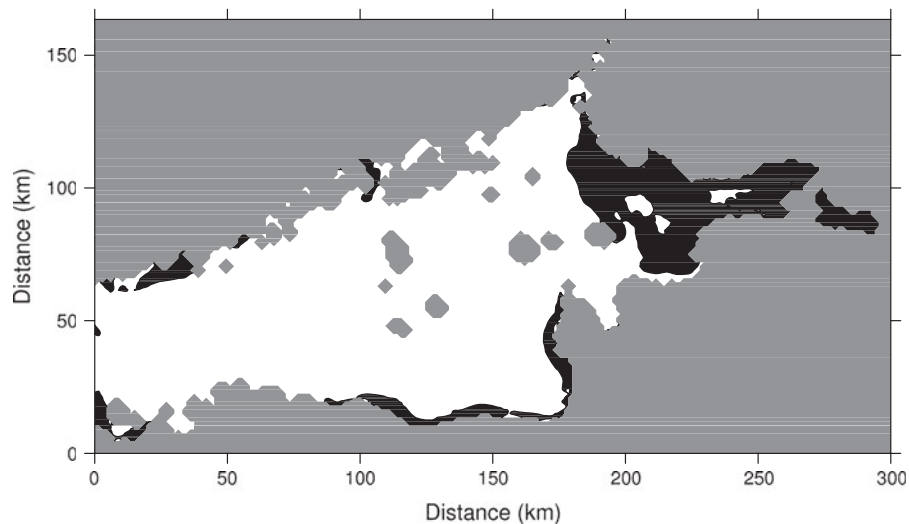


Fig. 9 Standardized gradient of mean modeled surface chl-*a* concentration in the Gulf of Finland. The area of positive gradient values is shown in black

gradients of chl-*a* along the coast of the Gulf of Finland, while offshore concentrations were lower and standardized gradients were negative.

Clear limits of the coastal zone could be identified in the central and eastern Gulf of Finland. Although water and material exchange between the coastal zone and the open sea is rather intensive in the central Gulf of Finland and coastal upwelling filaments may extend far offshore (Vahtera et al., 2005), composite satellite images and measurement data showed relatively low offshore concentrations and the presence of

a narrow coastal zone. According to the measurement data, on average during the biologically active season, a chl-*a* concentration of 0.01 mg l^{-1} delimits the outer border of the coastal zone in the central Gulf of Finland. Taking into account the difference between model and measurements, in the model a chl-*a* concentration of 0.005 mg l^{-1} is the threshold value indicating the limit of the coastal zone. The same concentration has been shown by the satellite data. A narrow belt of positive standardized gradients along the Estonian coast in the central Gulf is reproduced

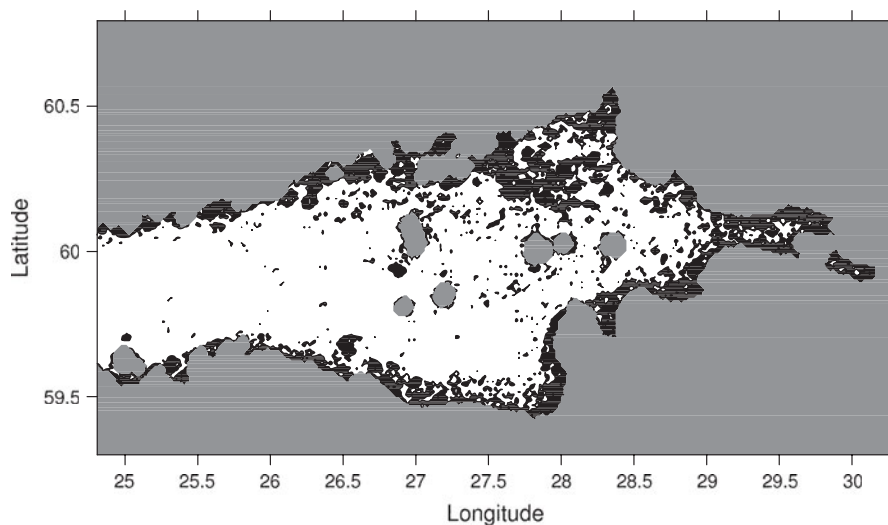


Fig. 10 Standardized gradient of surface chl-*a* concentration in the Gulf of Finland derived from satellite data. The area of positive gradient values is shown in black

better in the satellite data than in the model (this can be partly explained by the higher resolution of satellite images compared with the model grid). Gradients near the northern coast are reproduced similarly by both model and satellite data.

In the eastern Gulf, both satellite and model similarly reproduced the chl-*a* distribution along the southern coast. In the model, a coastal zone of 5–20 km width with a chl-*a* concentration above 0.004 mg l^{-1} is discernible from longitude 26°E toward the east. Satellite data indicate a higher chl-*a* concentration in this zone (0.008 mg l^{-1}). Nevertheless, the limits of the area of positive standardized gradients correspond well in both methods. The northern coast of the eastern Gulf can be characterized by chl-*a* concentrations of $0.004\text{--}0.006 \text{ mg l}^{-1}$ by both modeling and satellite data. However, the satellite image showed high patchiness in that region, and it also produced a vast area of positive gradients. This distribution pattern can be explained by the mean surface layer circulation to some extent. Mean currents between 4 and 10 cm s^{-1} occur to the east along the southern shore, while they are mainly $<4 \text{ cm s}^{-1}$ and offshore near the northern coast (Andrejev et al., 2004).

The chl-*a* concentrations were permanently high in the easternmost region of the Gulf of Finland ($>0.008 \text{ mg l}^{-1}$). Positive gradients of chl-*a* in the Neva Bay can be seen in both the satellite data and model. In the latter, the area of high chl-*a* and strong positive gradients propagates slightly further. The

coastal zone in the easternmost part of the study area could be defined as the region about 100-km wide from the Neva River mouth, following the definition of Artioli et al. (2005).

Temporal variations of chl-*a* concentration emphasize the dynamic nature of the coastal zone. During heavy blooms, i.e., the spring bloom and late summer cyanobacterial bloom in this case, the coastal zone may appear occasionally depending on the spatial distribution of the bloom. Due to the elongated shape of the Gulf of Finland, the cross-Gulf relative chl-*a* distribution was rather uniform during the spring bloom. The late summer cyanobacterial bloom is not directly taken into account in the model. Comparing the cross-Gulf relative chl-*a* concentrations on transects 1–3 shows that the coastal zone can be clearly identified in the summer and autumn on transects 2 and 3. Relatively high modeled chl-*a* concentrations in the offshore area along the western transect during the summer period could be the result of high chl-*a* concentrations prescribed at the model open boundary that are transported eastwards by model currents. Measured data from 2001 on the ferry line between Tallinn and Helsinki give a hint that even in the case of the summer cyanobacterial bloom, a narrow coastal zone of about several kilometers exists. Laanemets et al. (2006) showed that the cyanobacterial bloom was smaller in the central Gulf of Finland in 2001 than in 2002, but still significant.

Conclusion

A combined analysis of numerical modeling results, measurements, and satellite imagery data was used to study the existence of the coastal zone through variations of chl-*a* concentration in the central and eastern parts of the Gulf of Finland. By following the approach that the chl-*a* concentration is elevated near shore compared to the open sea, it was shown that the coastal zone is a dynamic concept that varies in time and space under the influence of ecohydrodynamic processes.

An analysis of the results of the chl-*a* distribution allowed the study area to be divided into two distinct parts: the central and eastern Gulf where high concentrations were present close to the coast; and Neva Bay which is usually high in chl-*a* due to the Neva River influence. It was shown that near the coast, areas of positive standardized gradients of chl-*a* concentration are found. The gradients are highest adjacent to the coast and drop rapidly toward the offshore, where gradients are negative and more or less uniform. Taking that into consideration, a standardized gradient of zero is a suitable criterion for setting the outer boundary of the coastal zone. Model results have shown that in the Neva Bay, elevated chl-*a* values can be found several kilometers offshore. Therefore, it is more appropriate to define the coastal zone in Neva Bay by the area of river influence. Using gradients of the distribution of ecological parameters for defining the extent of the coastal zone has been proposed by Lessin & Raudsepp (2007). Although in the current study different methods showed slightly different chl-*a* concentrations for the same regions, the limits of the coastal zone denoted by the magnitude of standardized gradients were similar.

Acknowledgments This work was partially supported by Estonian Science Foundation research grants numbers 5596 and 7581 and Danish EPA project EISEMM. Measured chl-*a* values in a transect between Tallinn and Helsinki were provided by the Alg@line project.

References

- Alenius, P., K. Myrberg & A. Nekrasov, 1998. The physical oceanography of the Gulf of Finland: a review. *Boreal Environment Research* 3: 97–125.
- Andrejev, O., K. Myrberg, P. Alenius & P. A. Lundberg, 2004. Mean circulation and water exchange in the Gulf of Finland—a study based on three-dimensional modelling. *Boreal Environment Research* 9: 1–16.
- Artioli, Y., G. Bendoricchio & L. Palmeri, 2005. Defining and modelling the coastal zone affected by the Po River (Italy). *Ecological Modelling* 184: 55–68.
- Burchard, H., O. Petersen & T. P. Rippeth, 1998. Comparing the performance of the Mellor–Yamada and the $k-\epsilon$ two-equation turbulence models. *Journal of Geophysical Research* 103: 10,543–10,554.
- Darecki, M. & D. Stramski, 2004. An evaluation of MODIS and SeaWiFS bio-optical algorithms in the Baltic Sea. *Remote Sensing of Environment* 89: 326–350.
- DHI Water and Environment, 2001. MIKE 3: environmental hydraulics. DHI Software User Guide, Documentation and Reference Manual.
- European Communities, 2000. Directive 2000/60/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 of the European Communities*, L327, 22 December 2000.
- Gazeau, F., S. V. Smith, B. Gentili, M. Frankignoulle & J.-P. Gattuso, 2004. The European coastal zone: characterization and first assessment of ecosystem metabolism. *Estuarine, Coastal and Shelf Science* 60: 673–694.
- Gibbs, M. T., A. J. Hobday, B. Sanderson & C. L. Hewitt, 2006. Defining the seaward extent of New Zealand's coastal zone. *Estuarine, Coastal and Shelf Science* 66: 240–254.
- Kahru, M., B. Hakansson & O. Rud, 1995. Distributions of the sea-surface temperature fronts in the Baltic Sea as derived from satellite imagery. *Continental Shelf Research* 15: 663–679.
- Kanoshina, I., U. Lips & J.-M. Leppanen, 2003. The influence of weather conditions (temperature and wind) on cyanobacterial bloom development in the Gulf of Finland (Baltic Sea). *Harmful Algae* 2: 29–41.
- Kononen, K., S. Hällfors, M. Kokkonen, H. Kuosa, J. Laanemets, J. Pavelson & R. Autio, 1998. Development of a subsurface chlorophyll maximum at the entrance to the Gulf of Finland, Baltic Sea. *Limnology and Oceanography* 43: 1089–1106.
- Laanemets, J., M.-J. Lilover, U. Raudsepp, R. Autio, E. Vahtera, I. Lips & U. Lips, 2006. A fuzzy logic model to describe the cyanobacteria *Nodularia spumigena* blooms in the Gulf of Finland, Baltic Sea. *Hydrobiologia* 554: 31–45.
- Lessin, G. & U. Raudsepp, 2006. Water quality assessment using integrated modeling and monitoring in Narva Bay, Gulf of Finland. *Environmental Modeling and Assessment* 11: 315–332.
- Lessin, G. & U. Raudsepp, 2007. Modelling the spatial distribution of phytoplankton and inorganic nitrogen in Narva Bay, southeastern Gulf of Finland, in the biologically active period. *Ecological Modelling* 201: 348–358.
- Lessin, G., I. Lips & U. Raudsepp, 2007. Modelling nitrogen and phosphorus limitation on phytoplankton growth in Narva Bay, south-eastern Gulf of Finland. *Oceanologia* 49: 259–276.

- Piirsoo, K., V. Porgasaar & M. Viik, 1992. Environmental conditions, phytoplankton and chlorophyll *a* in the Narva Bay (the southern part of the Gulf of Finland). Proceedings of the Estonian Academy of Sciences. *Biology* 41: 149–161.
- Rasmussen, E. B., 1993. Three-dimensional hydrodynamic models. Sect. 3.1 hydrodynamic models. In Abbot, M. B. & N. A. Price (eds), *Coastal, Estuarine and Harbour Engineer's Reference Book*. Chapman and Hall, London: 109–116.
- Rasmussen, E. B., J. Pietrzak & R. Brandt, 1999. A coupled ice-ocean model for the Greenland, Iceland and Norwegian Seas. *Deep-Sea Research Part II* 46: 1169–1198.
- Reinart, A. & T. Kutser, 2006. Comparison of different satellite sensors in detecting cyanobacterial bloom events in the Baltic Sea. *Remote Sensing of Environment* 102: 74–85.
- Telesh, V. I., 2004. Plankton of the Baltic estuarine ecosystems with emphasis on Neva Estuary: a review of present knowledge and research perspectives. *Marine Pollution Bulletin* 49: 206–219.
- Vahtera, E., J. Laanemets, J. Pavelson, M. Huttunen & K. Kononen, 2005. Effect of upwelling on the pelagic environment and bloom-forming cyanobacteria in the western Gulf of Finland, Baltic Sea. *Journal of Marine Systems* 58: 67–82.

Interactions of light and organic matter under contrasting resource simulated environments: the importance of clonal traits in the seagrass *Zostera noltii*

Irene Olivé · M. Paz García-Sánchez ·
Fernando G. Brun · Juan J. Vergara ·
J. Lucas Pérez-Lloréns

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 199–208.
DOI: 10.1007/s10750-009-9773-1 © The Author(s) 2009. This article is published with open access at Springerlink.com

Abstract Light reduction in the water column and enhanced organic matter (OM) load into the sediments are two main consequences of eutrophication in marine coastal areas. This study addresses the combined effects of light, OM, and clonal traits in the seagrass *Zostera noltii*. Large *Z. noltii* plants were grown in sand with or without the addition of OM and under two light levels (high light and low light). Whereas some complete plant replicates were grown under homogeneous light and/or OM conditions, other replicates were grown under contrasting light and/or OM levels between the apical and the distal parts of the same plant. The three-way factorial design (light, OM load, and apex position) allowed us to determine

the harmful effect of light reduction and OM enrichment on the growth, photosynthetic performance, and biochemical composition of *Z. noltii*. The addition of OM to the sediment promoted a decrease, or even an inhibition, in net plant growth regardless of the light level when the whole plants were grown under homogeneous light conditions. However, the results differed when plants were grown under contrasting light and/or OM conditions between apical and distal parts. In this case, the harmful effect of OM load was alleviated when apical parts were grown under high light conditions. OM loads also negatively affected the photosynthetic performance, evaluated as leaf fluorescence. The results indicate the importance of clonal traits in the response of *Z. noltii* growth to light conditions and OM enrichment.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

I. Olivé · M. P. García-Sánchez · F. G. Brun ·
J. J. Vergara · J. L. Pérez-Lloréns
Departamento de Biología, Área de Ecología, Facultad de Ciencias del Mar y Ambientales, Universidad de Cádiz, 11510 Puerto Real, Cádiz, Spain
e-mail: irene.olive@uca.es

F. G. Brun (✉)
Netherlands Institute of Ecology (NIOO-KNAW),
Centre for Estuarine and Marine Ecology, Korrिंगaweg 7,
4401 NT Yerseke, The Netherlands
e-mail: fernando.brun@uca.es

Keywords Apical dominance · Clonal integration · Growth · Light · Organic matter · *Zostera noltii*

Introduction

Seagrass beds, with their high productivity and biodiversity, have significant ecological and economic value (Costanza et al., 1997). In the past decades, the eutrophication observed in many coastal waters associated with an increase in nutrient loadings from wastewater, agricultural runoff, and other sources has been considered as a probable cause of the decline in the distribution of seagrasses in

temperate coastal environments (Hemminga & Duarte, 2000).

Two important processes have been linked to eutrophication-driven seagrass declines. First, nutrient enrichment stimulates the massive growth of phytoplankton, epiphytes (micro- and/or macro-), and free-floating opportunistic macroalgae (Valiela et al., 1997), which results in reduced light levels to the seagrasses beneath. Because the compensation irradiance for seagrass growth is higher than that for most ephemeral macroalgae (Enríquez et al., 1996), such light reduction may result in a negative seagrass growth (Longstaff et al., 1999; Brun et al., 2003b).

Second, there is an increased loading of sediments with organic matter (OM) as blooming algae die off (Borum & Sand-Jensen, 1996), increasing the respiration of the benthic microbial community (Jørgensen & Richardson, 1997; Holmer, 1999). Since the diffusion of oxygen from the overlying water into the sediment pore water is low (Kemp et al., 1992), and the basipetal translocation (from leaves to below-ground parts) of sucrose and oxygen and further release to the rhizosphere are reduced (as a consequence of lower light availability) (Smith et al., 1984; Zimmerman et al., 1996), this may generate a reduction in the redox potential of the sediments leading to anoxia. Such conditions are harmful to seagrasses not only because of the decrease of oxygen and sucrose transfer to the below-ground parts, but also because the occurrence of sulfide compounds in the surroundings of the rhizosphere may result in toxicity (Hemminga, 1998; Holmer & Bondgaard, 2001).

Besides the effects of eutrophication on seagrass meadows described above, seagrass growth promotes the emergence of biotic and abiotic gradients between seagrass beds and adjacent bare areas (Vichkovitten & Holmer, 2005) and even within the seagrass meadow itself (Brun et al., 2003a). For example, the 'edge effect', where runner shoots (apical shoots) escape from the meadow, implies a reduction in shoot density and a concomitant increase in light levels at the edge (Duarte & Sand-Jensen, 1990; Brun et al., 2003a; Enríquez & Pantoja-Reyes, 2005; Sintes et al., 2005). Nutrient concentrations in the pore water are lower at the meadow edges because plant biomass has been positively correlated with OM pools and sedimentation rates (Holmer & Nielsen, 1997; Barrón et al., 2004). Therefore, light levels are higher at the meadow periphery whereas nutrient concentrations

(mostly in sediment pore water) and OM are higher in the central parts of the meadow (Pedersen et al., 1997) resulting in a divergent resource allocation within the same clone.

Elucidation of the relative importance of light and OM loading on seagrass viability requires an experimental approach (Hemminga & Duarte, 2000). Most studies have been carried out looking at the effect of light reduction on seagrass physiology, growth, or meadow attributes (see review table in Brun et al., 2006). In comparison, few studies have been devoted to investigating the effect of OM loads (Terrados et al., 1999; Cancemi et al., 2003), while the combined effects of light and OM on seagrass growth, survival and photosynthetic activity have rarely been reported (Short et al., 1995; Holmer & Laursen, 2002; Holmer et al., 2005). In addition, the role of clonality in seagrass responses to light and/or OM loads has not yet been fully studied.

Clonality implies that the rhizome network, in addition to the anchorage and storage roles, is a sort of 'pipe' that physically and physiologically interconnects shoots (throughout 'clonal integration') within a single plant (Cline, 1997) and is governed by apical shoots [throughout 'apical dominance' (Terrados et al., 1997)]. Plants could benefit from the emergence of abiotic gradients within the meadow if clonal traits work at long distances, as was suggested by Marbà et al. (2002). Therefore, this would result in enhanced growth and meadow spreading because nutrients and photosynthates would be re-allocated within the plant mainly toward the apical shoots (Tomasko & Dawes, 1989; Marbà et al., 2002). Accordingly, clonality arises as an important issue for a better understanding of the effects of external forcing in seagrass ecology.

The aim of this study was to evaluate the interaction between light and OM enrichment on growth, photosynthesis (fluorescence), and elemental composition of the seagrass *Zostera noltii* taking into account its clonal nature.

Materials and methods

Sampling site

Specimens of *Zostera noltii* were collected from an intertidal sandy meadow at Santibañez (Cadiz Bay

Natural Park, 36°32'N, 6°15'W) in spring 2004 during the onset of its maximum growth phase (Brun et al., 2003a; Peralta et al., 2005). Plants were gathered carefully to keep below-ground parts intact and transported to the laboratory in an ice-chest. Before any experimental manipulation, plants were gently rinsed in seawater and visible epiphytes were removed by scraping carefully with paper. A total of 72 plants, with 5–6 internodes and one shoot per node, were selected and acclimated to the laboratory conditions in a clear aquarium during 24–48 h. All specimens were measured (number of internodes, internode lengths, total rhizome length, number of shoots, and leaf length), weighed (fresh weight, FW), and the rhizome axis was tagged at both ends (i.e., apical and distal positions).

Experimental design

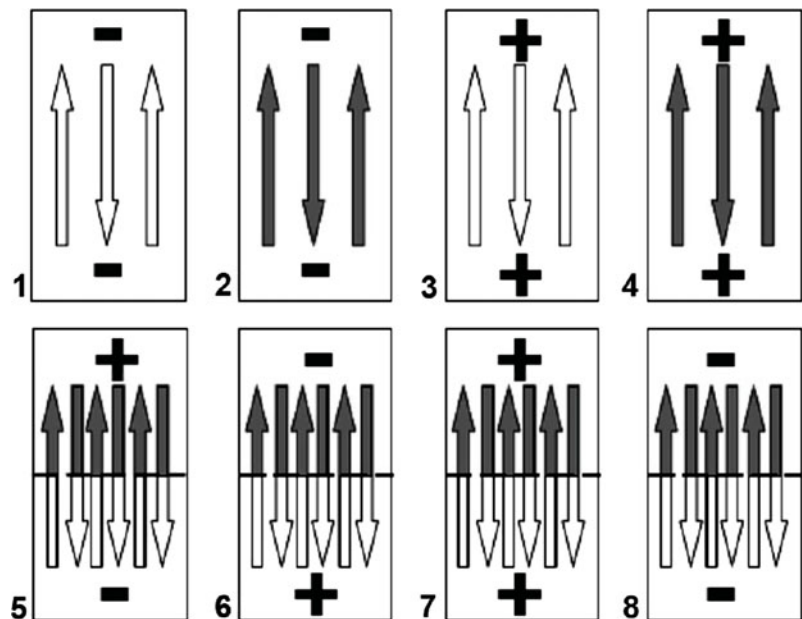
The effects of light, OM load, and ‘plant portion’ (i.e., apical versus distal plant parts) on growth, photosynthesis, and tissue elemental composition of *Z. noltii* were assessed using a three-way factorial design [2 light levels (high versus low) \times 2 OM treatments (non-enriched versus enriched) \times 2 ‘plant portions’ (apical part versus distal part)] in a mesocosm experiment.

Plants were transplanted into the aquaria according to the designed treatment of light [high light (HL) or

low light (LL)], OM [without OM (–) or with OM (+)] and position of the apical shoots. One set of plants was grown under ‘homogeneous conditions’; that is, the entire plant was subjected to the same light and OM treatments (aquaria 1–4, Fig. 1). The remaining set of plants was grown under ‘heterogeneous conditions’; that is, the apical half and distal half parts of the plant were subjected to different combinations of light and/or OM loads (aquaria 5–8, Fig. 1). To achieve this objective, aquaria were split into two halves (before adding the sand) using a holed transparent-perspex septum to allow a tight passage of the rhizome axis in a way that apical and distal plant parts could undergo contrasting experimental conditions. Three plants per aquarium ($n = 3$) were set in ‘homogeneous’ aquaria (1–4, Fig. 1) and six plants per aquarium ($n = 3$ apical orientation and $n = 3$ distal orientation) were set in ‘heterogeneous’ aquaria (5–8, Fig. 1). All the treatments were run in duplicated aquaria.

Plants were grown for 21 days in an incubation chamber (D-1400-3BL, ASL) at 18°C. This time span is long enough to detect any treatment-driven changes in the growth dynamic parameters of *Zostera noltii* (Brun et al., 2002; Peralta et al., 2002). Whole plants (in ‘homogeneous conditions’) or plant halves (in ‘heterogeneous conditions’) were exposed to either saturating light (HL, 120 $\mu\text{mol quanta m}^{-2} \text{s}^{-1}$) or limiting light (LL, 35 $\mu\text{mol quanta m}^{-2} \text{s}^{-1}$)

Fig. 1 Experimental design of the treatments in the aquaria. *White arrows* represent high light (HL), *dark arrows* represent low light (LL). (–) represents sediment without OM and (+) represents sediment with OM. The *head of the arrows* indicates the apical part of the plant. Each *arrow* indicates one plant. All aquaria were duplicated



depending on the experimental design. A 13:11 h light:dark photoperiod was established. Photon flux was measured with a LI 193 SA (LiCor) quantum meter mounted with a 4π PAR (400–700 nm) sensor. Light was supplied with fluorescent tubes (Philips TLD 18w/865). The limiting light level was achieved using neutral density filters to wrap the aquaria. Seawater in the aquaria was renewed once a week. To minimize any variation in light levels among the aquaria undergoing the same light conditions, they were randomly relocated on the shelves of the chamber after each water change. Water in the aquaria was homogenized with an air-pumping system. Redox potential was measured weekly (in each renovation of water) with redox potential electrodes (Redox combination electrode, SebTix ORP, WTW).

Sandy sediment was gently acid-washed and rinsed with tap water to reduce the amount of OM. Seawater was also filtered with cellulose paper. For the enriched OM treatments [(+) OM], acid-washed sediment was supplemented with a mixture of ground *Ulva* sp. (4.7 g DW l⁻¹ dry sediment) and sucrose (18 g l⁻¹ dry sediment).

Plant growth

After 21 days of culture, plants were collected, measured, split into apical and distal portions (and further into above- and below-ground biomass), and weighed to determine the following dynamic growth parameters according to Peralta et al. (2002): plant net growth rate (GR, mg FW plant⁻¹ d⁻¹), internode elongation rate (IER, cm plant⁻¹ d⁻¹), and internode appearance rate (IAR, no. of internodes plant⁻¹ d⁻¹). The ratio of above-ground:below-ground biomass (AG:BG) was also calculated on a fresh-weight basis.

Photosynthetic performance

Before weighing, photosynthetic characteristics of the leaves were assessed by determining the maximum photochemical efficiency of photosystem II (F_v/F_m) using chlorophyll-*a* fluorescence (for details on this technique, see Krause & Weis, 1991). The measurements were made with a PAM-2000 (Walz Effeltrich, Germany). To reduce any within-shoot variability in F_v/F_m , fluorescence was measured in the middle part of the second innermost leaf of randomly selected shoots along the plant. In plants

under heterogeneous conditions (aquaria 5–8), measurements were conducted in both halves of the aquaria after selecting shoots randomly in each half. A leaf clip (DLC, Walz) connected with fiber optics to the fluorometer was mounted in the middle part of the leaf. A 5-s weak far-red pulse was applied to oxidize the electron transport chain (Hanelt, 1998), after which the shutter of the clips was closed. After 5 min of dark acclimation, the ground fluorescence (F_0) was estimated, followed by a saturating pulse to measure maximal fluorescence (F_m), allowing calculation of the variable fluorescence ($F_v = F_m - F_0$) and the maximum photochemical efficiency (F_v/F_m).

Elemental composition

Above-ground parts (shoots) and below-ground parts (rhizomes + roots) from apical and distal portions of the plants were oven-dried (60°C) until constant weight to determine the dry weight of each fraction. Sub-samples of all fractions were powdered and stored for nutrient content analysis. Tissue carbon (C) and nitrogen (N) were determined using a Perkin-Elmer 240 CNH elemental analyzer.

Statistical analysis

The effects of light availability, OM, and plant portion on the growth, photosynthesis and biochemical composition of *Z. noltii* were determined using a hierarchical (nested) analysis of variance (ANOVA). The levels of variability were the aquaria on each treatment ($n = 2$) and the plants sampled within each aquarium ($n = 3$ aquaria 1–4; $n = 6$ (3 apical + 3 distal) aquaria 5–8). Normality of the data and homogeneity of variances were tested (Zar, 1984). Multiple post-hoc comparisons between means were assessed by the Unequal N HSD procedure. For non-parametric data, the Kruskal-Wallis test was used. Results are expressed as mean \pm SE.

Results

Abiotic variables

Initial light levels in the aquaria were 120 and 35 $\mu\text{mol quanta m}^{-2} \text{s}^{-1}$ (HL and LL, respectively). In the HL treatments, there was a slight decrease of

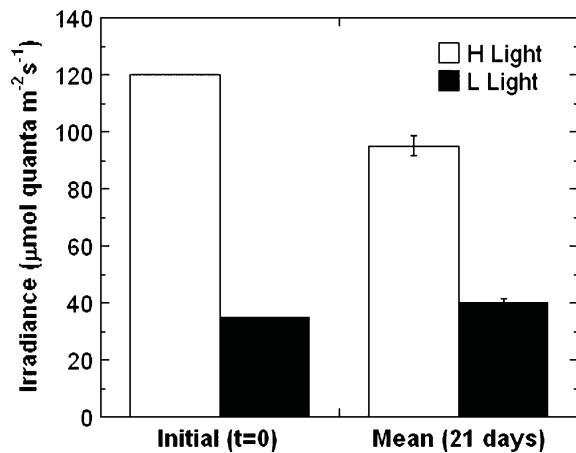


Fig. 2 Irradiance levels at the bottom of the aquaria at the beginning of the experiment [initial ($t = 0$)] and the mean value during the 21 days of culture [mean (21 days)] for aquaria at high light (white column) and low light (dark column) conditions

irradiance during the experiment owing to an increase in turbidity, probably caused by phytoplankton and bacterial growth, even though the seawater was renewed weekly (Fig. 2). Thus, the mean irradiance for all the aquaria during the experiments was $95 \pm 3.6 \mu\text{mol quanta m}^{-2} \text{s}^{-1}$ (HL) and $40 \pm 1.2 \mu\text{mol quanta m}^{-2} \text{s}^{-1}$ (LL) (Fig. 2) and differences in light dose remained significant during the experiment ($P < 0.001$).

The OM content of the sediments at the beginning of the experiments was $2.20 \pm 0.13\%$ DW for OM-enriched sediments [(+) OM] and $0.55 \pm 0.04\%$ DW for non-OM-enriched sediments [(-) OM] (Fig. 3). At the end of the incubations, the percentage of OM in the sediment decreased significantly to half of initial content in the (+) OM sediments ($0.95 \pm 0.03\%$ DW), whereas the percentage increased slightly in the (-) OM sediments ($0.80 \pm 0.03\%$ DW) (Fig. 3). However, significant differences in the sediment OM content between (+) OM and (-) OM treatments persisted up to the end of the experiment ($P < 0.05$). Redox potential in the sediment was unaffected by the light regime but depended largely on the OM content (data not shown). The lowest redox potential values were recorded at the end of the experiment in (+) OM sediments ($-312 \pm 8 \text{ mV}$, $n = 12$), whereas 4-fold higher values were recorded in (-) OM sediments ($-75 \pm 38 \text{ mV}$, $n = 12$).

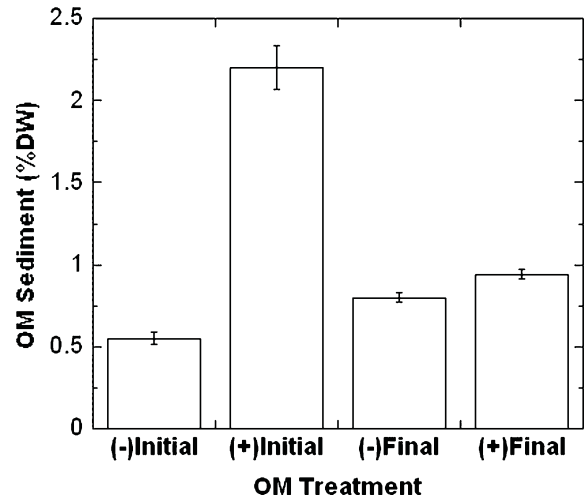


Fig. 3 Organic matter content (% DW) in the sediment without (-) and with (+) OM at the beginning (initial) and at the end (final) of the experiment

Plant dynamic parameters

Both light and OM significantly affected all plant dynamic parameters measured at the end of the experiment (Table 1). When the whole plants were grown under 'homogeneous conditions' (i.e., the entire plant was subjected to the same light and OM treatments; aquaria 1–4, Fig. 4), a significant decrease in net growth rates (GR), net internode elongation rates (IER), and net internode appearance rates (IAR) was observed under (+) OM regardless of the light regime (Fig. 4A–C; aquaria 3 and 4 compared to 1 and 2). However, the results differed when plants were grown under contrasting light between apical and distal parts (Fig. 4, aquaria 7 and 8). Positive IERs were recorded in plants from (-) OM treatments (aquarium 8) regardless of the light regime, while IERs in plants from (+) OM treatments (aquarium 7) were lower and influenced by the light regime (Fig. 4B; aquaria 7 and 8).

Plants maintained under heterogeneous light and/or OM conditions showed positive IARs and IERs (but negative GRs) when apical parts were under HL (Fig. 4A–C aquaria 5 and 6), regardless of the OM conditions. However, lower or negative rates were recorded when the apices were under LL (Fig. 4A–C aquaria 5 and 6).

It should be noted that net GR represents a net biomass balance, while IER and IAR denote the

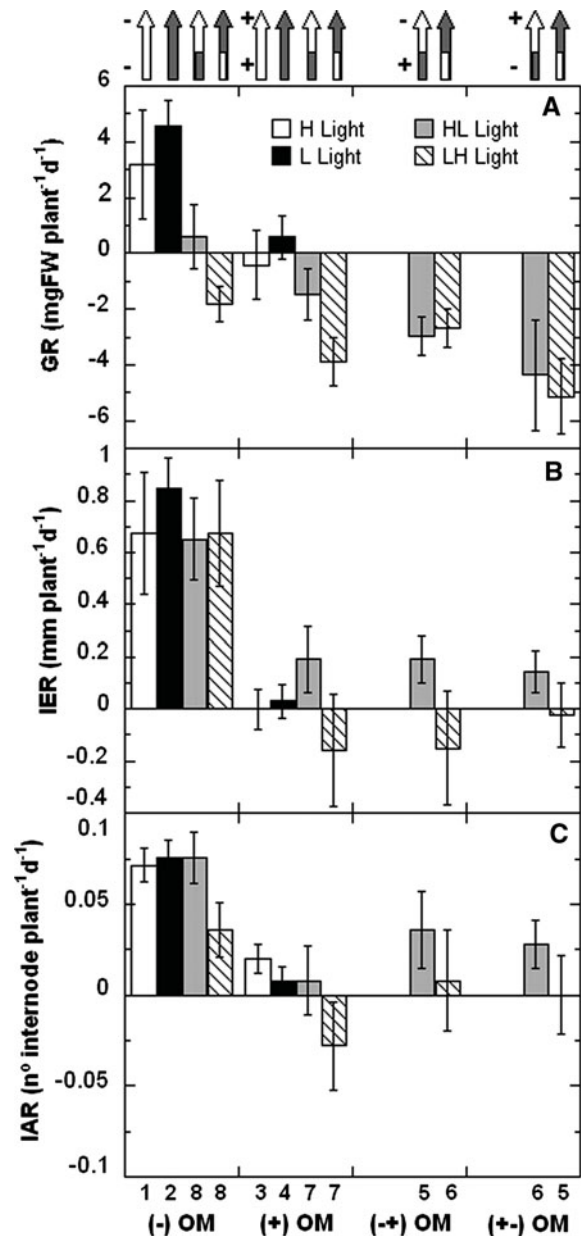
Fig. 4 **A** Growth rates on a fresh-weight basis, GR; **B** internode elongation rates, IER; **C** internode appearance rates, IAR. *White columns* indicate plants grown at high light (HL) and homogeneous conditions, *dark columns* indicate plants grown at low light (LL) and homogeneous conditions, *soft gray columns* indicate plants where apical parts were in HL and distal parts under LL conditions, *hatched columns* indicate plants where apical parts were in LL and distal parts in HL. (–) means plants without OM, (+) means plants with OM, (+ –) means plants where apical parts were grown with OM and distal parts were without OM, (– +) indicates plants where apical parts were grown without OM and distal parts were with OM. The numbers on the *x*-axis indicate the number of the aquarium in which the plants were potted. The head of the arrows on the top of the graph indicates the apical part of the plants

actual, integrated growth of the below-ground system of the plants during the experiment. Therefore, net positive IER and IAR can be accompanied by net biomass losses. In this sense, plants grown under homogeneous conditions without OM, regardless of light level (aquaria 1 and 2), presented significant higher GRs than those from the rest of the treatments (homogeneous and heterogeneous) (Fig. 4A, Table 1). This result could indicate that plants from aquaria 1 and 2 were still actively growing at the end of the experiment. Thus, while OM plays an important role in *Zostera noltii* grown under homogeneous conditions, light becomes the main driving force under heterogeneous conditions. Positive IERs and IARs were mainly recorded in plants where apical portions grew under HL, regardless of the OM treatments.

Above-ground:below-ground fresh biomass ratios (AG:BG) at the end of the experiment showed interesting trends among treatments. On one hand, the AG:BG values for complete plants (0.3 ± 0.02) (aquaria 1–4) were higher than those for distal portions (0.1 ± 0.01) but lower than those obtained for apical portions (0.4 ± 0.03) of split plants. Significant differences were found between complete plants and distal portions ($P < 0.005$) and between apical and distal portions of the split plants ($P < 0.0001$), but no significant differences were detected between complete plants and apical portions. On the other hand, the lowest AG:BG ratios were recorded in plants under LL and (+) OM conditions (data not shown).

Photosynthetic performance (fluorescence)

Maximum photochemical efficiency (F_v/F_m) was significantly affected by the addition of OM (Fig. 5, Table 1). Thus, shoots grown under (–) OM



presented higher F_v/F_m values (ca. 0.7) than shoots grown under (+) OM (ca. 0.4) at the end of the experiment, indicating a sub-optimal status of the photosynthetic apparatus in (+) OM plants. Interestingly, the addition of OM had a greater effect (decrease in F_v/F_m) in shoots grown in heterogeneous conditions [(±) OM treatments] than in shoots from plants grown entirely in OM [(+) OM treatment] (Fig. 5). F_v/F_m values were not significantly affected by light growing conditions (Table 1).

Table 1 *Zostera noltii*

| Variable (factors) | Statistical text | df treatments/df error | <i>F</i> (<i>P</i>) |
|-----------------------------|------------------|------------------------|-----------------------|
| GR | Nested ANOVA | | |
| Light | | 3/60 | 15.17* |
| OM (Light) | | 8/60 | 3.06* |
| IER | Nested ANOVA | | |
| Light | | 3/60 | 2.87* |
| OM (Light) | | 8/60 | 6.40* |
| IAR | Nested ANOVA | | |
| Light | | 3/60 | 3.62* |
| OM (Light) | | 8/60 | 3.03* |
| Fluorescence (F_v/F_m) | ANOVA | | |
| OM | | 3/42 | 6.13* |
| Fluorescence (F_v/F_m) | ANOVA | | |
| Light | | 3/42 | 0.48 (n.s.) |
| %C | ANOVA | | |
| Above- versus below-biomass | | 1/77 | 89.34* |
| %N | ANOVA | | |
| Above- versus below-biomass | | 1/77 | 130.83* |
| %C Shoot | Nested ANOVA | | |
| Light | | 3/19 | 1.2 (n.s.) |
| OM (Light) | | 8/19 | 1.88 (n.s.) |
| Apical–distal (Light-OM) | | 8/19 | 2.23 (n.s.) |
| %C Rhizome | Nested ANOVA | | |
| Light | | 3/19 | 0.05 (n.s.) |
| OM (Light) | | 8/19 | 1.97 (n.s.) |
| Apical–distal (Light-OM) | | 8/19 | 5.58* |
| %N Shoot | Nested ANOVA | | |
| Light | | 3/19 | 0.10 (n.s.) |
| OM (Light) | | 8/19 | 6.78* |
| Apical–distal (Light-OM) | | 8/19 | 5.32* |
| %N Rhizome | Kruskal–Wallis | | |
| Light | | 3 | 0.07 (n.s.) |
| OM | | 3 | 2.98 (n.s.) |
| Apical–distal | | 2 | 21.93** |

Statistical results of the ANOVA and the nested ANOVA analyses examining the effects of the light reduction, OM supply, and apical versus distal part of the plant in growth rates (GR, IER, and IAR), fluorescence measurements (F_v/F_m), and C, N content. When data were non-parametric, a Kruskal–Wallis test was applied
df degrees of freedom,
F test value
P-value where * $P < 0.05$,
** $P < 0.001$, and *n.s.* no significant differences

Tissue elemental composition

Whole plant carbon content was not affected by light or OM whereas OM load affected the N content in shoots (Table 1). Shoot N content decreased (22% reduction; data not shown) under (+) OM conditions regardless of light levels. The mean C and N content in shoots (29% and 2.7%, respectively) were significantly higher than in root-rhizomes (22% and 1.2%, respectively) (statistics in Table 1). Regardless of light or OM conditions, the apices of split plants (aquaria 5–8) showed higher C and N contents (31%

C, 3.0% N in shoots; 24% C, 1.6% N in below-ground tissues) than distal parts (27% C, 2.4% N in shoots; 20% C, 0.9% N in below-ground tissues) (Fig. 6; statistics in Table 1).

Discussion

The three-way factorial design carried out in this study with light, OM enrichment, and relative plant position allowed the authors a better understanding of the interactions among these three factors and their

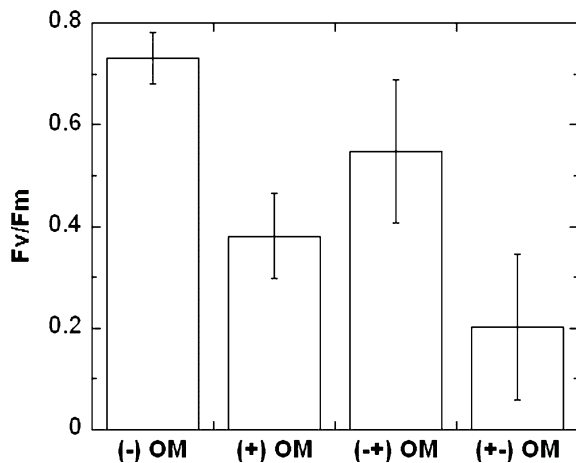


Fig. 5 Maximum photochemical efficiency (F_v/F_m) of shoots grown under different OM conditions in *Z. noltii* plants. (-) and (+) OM indicates plants under homogeneous conditions of OM (without and with OM, respectively); (- +) and (+ -) OM indicates plants under heterogeneous OM conditions [(- +) means without OM in apical-with OM in distal parts and (+ -) means with OM in apical-without OM in distal parts]

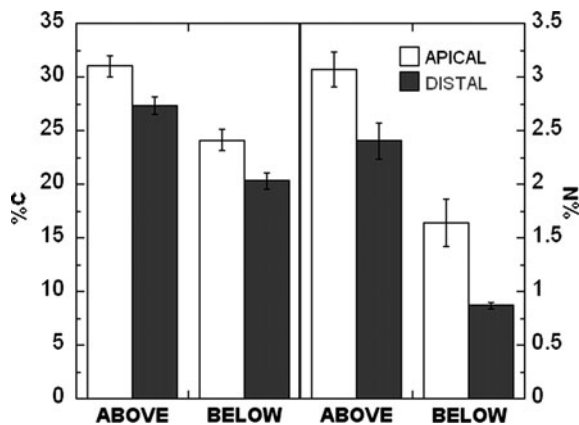


Fig. 6 Carbon and nitrogen content (% DW) in both above-ground and below-ground tissues of *Z. noltii*. White columns indicate apical portions of the plant and gray columns indicate distal portions

relative importance at the whole-plant level response because the experimental conditions resembled the different situations that could be found in the field.

The clonal nature of seagrass involves the concepts of apical dominance and clonal integration. On one hand, apical dominance denotes the governing influence of the plant apex on the growth and orientation of lateral organs (Woolley & Wareing,

1972; reviewed by Phillips, 1975). On the other hand, clonal integration allows plants a free-sharing of resources among integrated, connected seagrass modules (Libés & Boudouresque, 1987; Tomasko & Dawes, 1989; Marbà et al., 2002).

Plant growth estimators and photosynthesis were affected by OM enrichment, as previously described in other seagrass species (Terrados et al., 1999; Holmer & Laursen, 2002; Holmer et al., 2005). Light limitation also reduced GR when apices of *Z. noltii* grew under LL, in agreement with previous findings for this species (Peralta et al., 2002; Brun et al., 2003b). However, the novelty of the present approach is the finding that the whole-plant response to both stressors (assayed either alone or together) is modulated by, and largely dependent on, the relative exposure of the plant apex to different conditions and by the existence of contrasting resource environments affecting the same plant. The capacity of these plants to cope with environmental constraints as well as to enhance their potential for colonization would be increased through clonal integration. Accordingly, stressed shoots could be supported by resources provided by modules growing in more favorable conditions, as reported for *Thalassia testudinum* (Tomasko & Dawes, 1989). Consequently, such inter-shoot allocation and sharing of resources would reduce the total availability of plant resources and subsequently would reduce the whole plant growth under heterogeneous conditions in contrast to those grown under homogeneous conditions, as recorded in the present study. Therefore, the results presented here show the key role that both clonal traits play in the elucidation of the whole-plant response to single or multiple stressors. Since apical dominance and clonal integration have been suggested to be species-specific, and environmental heterogeneity may occur at several spatial and temporal scales, the evaluation of competition and fitness of a given species living in a particular environment should account for clonal traits and abiotic gradients in order to explain and/or predict the final outcome.

Photosynthesis is the main pathway used by primary producers to acquire energy and organic carbon with a parallel production of molecular oxygen. Since most seagrasses usually inhabit hypoxic to anoxic sediments (Terrados et al., 1999), their survival depends largely on oxygen and resource translocation from shoots to below-ground tissues (Hemminga, 1998). Light reduction diminishes photosynthesis and,

thus, carbon assimilation and oxygen production (Hemminga, 1998). Decreases in growth rate (GR) and photosynthetic ability (measured as F_v/F_m) and the presence of necrotic tissues (personal observation) were recorded under OM (+) treatments. This indicates the harmful effect of the OM, probably mediated by high sulfide concentrations in the rhizosphere (Eh ca. -300 mV). It would be expected that the greatest sulfide intrusions occurred in (+) OM treatments under LL, as found by Holmer & Laursen (2002) and Holmer et al. (2005). Those authors recorded a significant reduction in shoot density, leaf GR, and higher sulfide intrusion rates in plants inhabiting OM-enriched sediments under experimental shading. However, our results showed that OM loads caused harmful effects on growth and photosynthesis in complete plants regardless of the light level, both when light and OM were assayed alone or together. The plant damage observed under (+) OM occurred even at HL where it would be expected that light was high enough to drive the allocation of photosynthates and oxygen down to the rhizosphere, thus avoiding anoxia. Nevertheless, the plant response did change when the relative position of *Z. noltii* apices was considered, and contrasting levels of light and/or OM were established along the plant. Synergistic effects of light and OM on plant GR and photosynthesis were recorded (Figs. 4 and 5), revealing the apices as the 'leading' plant parts and more affected by light deprivation than by OM load. In seagrasses, the newly incorporated resources are preferentially allocated into the plant apex (Marbà et al., 2002). Accordingly, the higher AG:BG fresh biomass ratios obtained in apical parts compared to distal parts could suggest that the energetic requirements, including the formation of new photosynthetic tissues, would be higher in apices.

Conclusion

In conclusion, OM load and light levels significantly affected the growth and photosynthesis of *Zostera noltii* plants. The full-factorial design applied accounted for the heterogeneity in abiotic conditions recorded in natural meadows. This experimental setup showed that apical dominance and clonal integration modulated the whole-plant response and indicated the importance of both clonal traits in the plant response to the environmental factors assayed.

Acknowledgments This study was supported by the project CTM2005-00395/MAR of the Spanish National Program in Marine Science and Technology from the Ministerio Español de Educación y Ciencia. I. Olivé and M. P. García-Sánchez hold a grant from the Ministerio Español de Educación y Ciencia. F. G. Brun is supported by an individual Marie Curie fellowship (MEIF-CT-2005-515071) and by a European Reintegration Grant (MERG-CT-2007-205675). This is the NIOO publication number 4380.

Open Access This article is distributed under the terms of the Creative Commons Attribution Noncommercial License which permits any noncommercial use, distribution, and reproduction in any medium, provided the original author(s) and source are credited.

References

- Barrón, C., N. Marbà, J. Terrados, H. Kennedy & C. M. Duarte, 2004. Community metabolism and carbon budget along a gradient of seagrass (*Cymodocea nodosa*) colonization. *Limnology and Oceanography* 49: 1642–1651.
- Borum, J. & K. Sand-Jensen, 1996. Is total primary production in shallow coastal marine waters stimulated by nitrogen loading? *Oikos* 76: 406–410.
- Brun, F. G., I. Hernández, J. J. Vergara, G. Peralta & J. L. Pérez-Lloréns, 2002. Assessing the toxicity of ammonium pulses to the survival and growth of *Zostera noltii*. *Marine Ecology Progress Series* 225: 177–187.
- Brun, F. G., J. L. Pérez-Lloréns, I. Hernández & J. J. Vergara, 2003a. Patch distribution and within-patch dynamics of the seagrass *Zostera noltii* Hornem. in Los Toruños salt-marsh, Cádiz Bay, Natural Park, Spain. *Botanica Marina* 46: 513–524.
- Brun, F. G., J. J. Vergara, G. Navarro, I. Hernández & J. L. Pérez-Lloréns, 2003b. Effect of shading by *Ulva rigida* canopies on growth and carbon balance of the seagrass *Zostera noltii*. *Marine Ecology Progress Series* 265: 85–96.
- Brun, F. G., J. J. Vergara, G. Peralta, M. P. García-Sánchez, I. Hernández & J. L. Pérez-Lloréns, 2006. Clonal building, simple growth rules and phylloclimate as key steps to develop functional-structural seagrass models. *Marine Ecology Progress Series* 323: 133–148.
- Cancemi, G., G. De Falco & G. Pergent, 2003. Effects of organic matter input from a fish farming facility on a *Posidonia oceanica* meadow. *Estuarine, Coastal and Shelf Science* 56: 961–968.
- Cline, M. G., 1997. Concepts and terminology of apical dominance. *American Journal of Botany* 84: 1064–1069.
- Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton & M. van der Belt, 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.
- Duarte, C. M. & K. Sand-Jensen, 1990. Seagrass colonization—biomass development and shoot demography in *Cymodocea nodosa* patches. *Marine Ecology Progress Series* 67: 97–103.

- Enríquez, S. & N. I. Pantoja-Reyes, 2005. Form-function analysis of the effect of canopy morphology on leaf self-shading in the seagrass *Thalassia testudinum*. *Oecologia* 145: 235–243.
- Enríquez, S., C. M. Duarte, K. Sand-Jensen & S. L. Nielsen, 1996. Broad scale comparison of photosynthetic rates across phototrophic organisms. *Oecologia* 108: 197–206.
- Hanelt, D., 1998. Capability of dynamic photoinhibition in Arctic macroalgae is related to their depth distribution. *Marine Biology* 131: 361–369.
- Hemminga, M. A., 1998. The root/rhizome system of seagrass: an asset and a burden. *Journal of Sea Research* 39: 183–196.
- Hemminga, M. A. & C. M. Duarte, 2000. *Seagrass Ecology*. Cambridge University Press, Cambridge.
- Holmer, M., 1999. The effect of oxygen depletion on anaerobic organic matter degradation in marine sediments. *Estuarine, Coastal Shelf Science* 48: 383–390.
- Holmer, M. & E. J. Bondgaard, 2001. Photosynthetic and growth response of eelgrass to low oxygen and high sulfide concentrations during hypoxic events. *Aquatic Botany* 70: 29–38.
- Holmer, M. & L. Laursen, 2002. Effect of shading of *Zostera marina* (eelgrass) on sulfur cycling in sediments with contrasting organic matter and sulfide pools. *Journal of Experimental Marine Biology and Ecology* 270: 25–37.
- Holmer, M. & S. L. Nielsen, 1997. Sediment sulfur dynamics related to biomass-density patterns in *Zostera marina* (eelgrass) beds. *Marine Ecology Progress Series* 146: 163–171.
- Holmer, M., M. S. Frederiksen & H. Møllegaard, 2005. Sulfur accumulation in eelgrass (*Zostera marina*) and effect of sulfur on eelgrass growth. *Aquatic Botany* 81: 367–379.
- Jørgensen, B. B. & K. Richardson, 1997. *Eutrophication in Coastal Marine Ecosystems*. American Geochemical Union Press, USA.
- Kemp, V. M., P. Sampou, J. Garber, J. Tuttle & W. R. Boynton, 1992. Seasonal depletion of oxygen from bottom waters of Chesapeake Bay: roles of benthic and planktonic respiration and physical exchange processes. *Marine Ecology Progress Series* 85: 137–152.
- Krause, G. H. & E. Weis, 1991. Chlorophyll fluorescence and photosynthesis: the basics. *Annual Review of Plant Physiology and Plant Molecular Biology* 42: 313–349.
- Libés, M. & C. F. Boudouresque, 1987. Uptake and long-distance transport of carbon in the marine phanerogam *Posidonia oceanica*. *Marine Ecology Progress Series* 38: 177–186.
- Longstaff, B. J., N. R. Loneragan, M. J. O'Donohue & W. C. Dennison, 1999. Effects of light deprivation on the survival and recovery of the seagrass *Halophila ovalis* (R.Br.) Hook. *Journal of Experimental Marine Biology and Ecology* 234: 1–27.
- Marbà, N., M. A. Hemminga, M. A. Mateo, C. M. Duarte, Y. E. M. Mass, J. Terrados & E. Gacia, 2002. Carbon and nitrogen translocation between seagrass ramets. *Marine Ecology Progress Series* 226: 287–300.
- Pedersen, M. F., C. M. Duarte & J. Cebrian, 1997. Rates of changes in organic matter and nutrient stocks during seagrass *Cymodocea nodosa* colonization and stand development. *Marine Ecology Progress Series* 159: 29–36.
- Peralta, G., J. L. Pérez-Lloréns, I. Hernández & J. J. Vergara, 2002. Effects of light availability on growth, architecture and nutrient content of the seagrass *Zostera noltii* Hornem. *Journal of Experimental Marine Biology and Ecology* 269: 9–26.
- Peralta, G., F. G. Brun, I. Hernández, J. J. Vergara & J. L. Pérez-Lloréns, 2005. Morphometric variations as acclimation mechanisms in *Zostera noltii* beds. *Estuarine, Coastal and Shelf Science* 64: 347–356.
- Phillips, I. D. J., 1975. Apical dominance. *Annual Review of Plant Physiology and Plant Molecular Biology* 26: 342–367.
- Short, F. T., D. M. Burdick & J. E. I. Kaldy, 1995. Mesocosm experiments quantify the effects of eutrophication on eelgrass, *Zostera marina*. *Limnology and Oceanography* 40: 740–749.
- Sintes, T., N. Marba, C. M. Duarte & G. A. Kendrick, 2005. Nonlinear processes in seagrass colonisation explained by simple clonal growth rules. *Oikos* 108: 165–175.
- Smith, R. D., W. C. Dennison & R. S. Alberte, 1984. Role of seagrass photosynthesis in root aerobic processes. *Plant Physiology* 74: 1055–1058.
- Terrados, J., C. M. Duarte & W. J. Kenworthy, 1997. Experimental evidence for apical dominance in the seagrass *Cymodocea nodosa*. *Marine Ecology Progress Series* 148: 263–268.
- Terrados, J., C. M. Duarte, L. Kamp-Nielsen, N. S. R. Agawin, E. Gacia, D. Lacap, M. D. Fortes, J. Borum, M. Lubanski & T. Greve, 1999. Are seagrass growth and survival constrained by the reducing conditions of the sediment? *Aquatic Botany* 65: 175–197.
- Tomasko, D. A. & C. J. Dawes, 1989. Evidence for physiological integration between shaded and unshaded short shoots of *Thalassia testudinum*. *Marine Ecology Progress Series* 54: 299–305.
- Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh & K. Foreman, 1997. Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography* 42: 1105–1118.
- Vichkovitten, T. & M. Holmer, 2005. Dissolved and particulate organic matter in contrasting *Zostera marina* (eelgrass) sediments. *Journal of Experimental Marine Biology and Ecology* 316: 183–201.
- Woolley, D. J. & P. F. Wareing, 1972. The interaction between growth promoters in apical dominance. II. Environmental effects on endogenous cytokinin and gibberellin levels in *Solanum andigena*. *New Phytologist* 71: 1015–1025.
- Zar, J. H., 1984. *Biostatistical Analysis*. Prentice-Hall, New Jersey.
- Zimmerman, R. C., D. G. Kohrs & R. S. Alberte, 1996. Top-down impact through a bottom-up mechanism: the effect of limpet grazing on growth, productivity and carbon allocation of *Zostera marina* L. (eelgrass). *Oecologia* 107: 560–567.

Long-term modeling of large-scale nutrient cycles in the entire Baltic Sea

Oleg P. Savchuk · Fredrik Wulff

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 209–224.
DOI: 10.1007/s10750-009-9775-z © Springer Science+Business Media B.V. 2009

Abstract Management of eutrophication in marine ecosystems requires a good understanding of nutrient cycles at the appropriate spatial and temporal scales. Here, it is shown that the biogeochemical processes controlling large-scale eutrophication of the Baltic Sea can be described with a fairly aggregated model: simple as necessary Baltic long-term large scale (SANBALTS). This model simulates the dynamics of nitrogen, phosphorus, and silica driven by the external inputs, the major physical transports, and the internal biogeochemical fluxes within the seven major sub-basins. In a long-term hindcast (1970–2003), the model outputs reasonably matched observed concentrations and fluxes. The model is also tested in a scenario where nutrient inputs are reduced to levels that existed over 100 years ago. The simulated response of the Baltic Sea trophic state to this very large reduction is verified by a similar simulation made with a much more complex process-oriented model. Both models indicate that after initial, rather rapid changes the system goes into

much slower evolution, and nutrient cycles would not become balanced even after 130 years.

Keywords Eutrophication · Baltic Sea · Biogeochemical model · Nitrogen · Phosphorus · Silica

Introduction

Many studies of the Baltic Sea eutrophication have now resulted in good descriptions of its symptoms and consequences (e.g., Anonymous, 2001; Wulff et al., 2001b; Rönnerberg & Bonsdorff, 2004). The challenge is to use this knowledge for the development of scientifically sound recommendations for cost-effective mitigation measures (Wulff et al., 2001a), for which modeling is an important tool.

Considering eutrophication as an imbalance in the large-scale nutrient cycles, whereby more nutrients come into the system than leave it, the primary goal of such a model is to simulate the interplay between nutrient sources and sinks determined by external loads, inter-basin water exchanges, and internal biogeochemical fluxes. The geographical and ecosystem characteristics of the Baltic Sea, with long residence times of both water and nutrients, require long-term forecasts encompassing the entire Baltic Sea. Studies of processes occurring at local scales could easily demand massive computer resources with highly complex three-dimensional (3D) models.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

O. P. Savchuk (✉) · F. Wulff
Baltic Nest Institute and Department of Systems Ecology,
Stockholm University, 10691 Stockholm, Sweden
e-mail: oleg@mbox.su.se

The ambition of the authors is to develop a model that is fast enough so that it can be used in a decision-support system, linked with other models describing, for example, drainage basin dynamics (Wulff et al., 2007). Fortunately, the large-scale dynamics of nutrients in the Baltic Sea appear to be driven by a few first-order biogeochemical and physical processes that can be described in fairly aggregated models (Sjöberg et al., 1972; Savchuk, 1986; Wulff & Stigebrandt, 1989; Savchuk & Volkova, 1990). A further development of this approach is the simple as necessary Baltic long-term large-Scale (SANBALTS) model presented here in detail.

Materials and methods

The objective of SANBALTS is to simulate the coupled nitrogen (N), phosphorus (P), and silicon (Si) biogeochemical cycles in the major basins of the Baltic Sea (Fig. 1a), treated as homogeneous boxes except the Baltic Proper that is split into the surface (BPs, 0–60 m) and deep (BPD, 60 m to bottom) boxes.

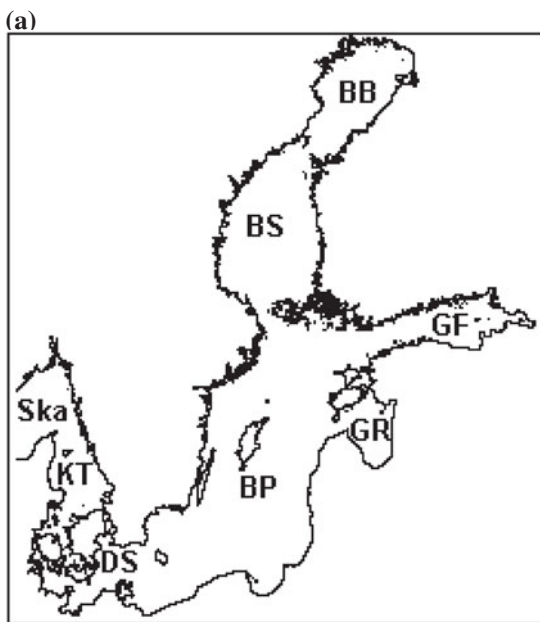
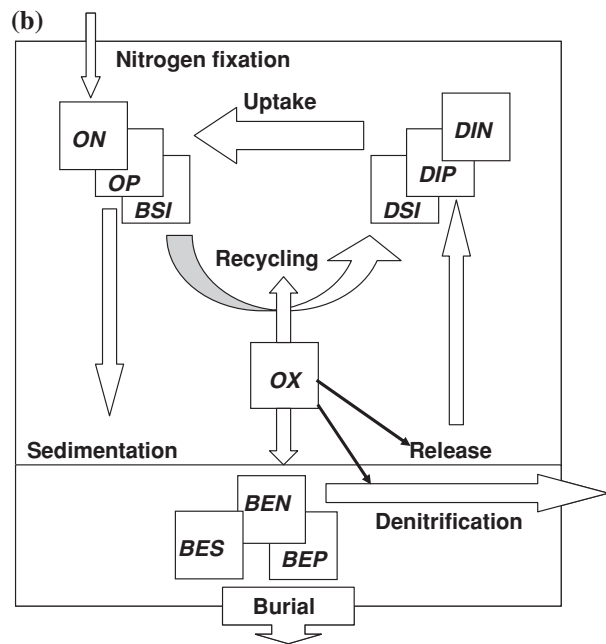


Fig. 1 SANBALTS set-up. **a** Major basins (model boxes): *BB* the Bothnian Bay (1), *BS* the Bothnian Sea (2), *BP* the Baltic Proper (3—surface box, 8—deep box), *GF* the Gulf of Finland

Variables and interactions

Biogeochemical fluxes that together with water transports and external inputs determine nutrient concentrations in the model are primary production, nitrogen fixation, pelagic recycling, sedimentation, output from the sediments, denitrification in the sediments and in the water column in BPD, and burial (Fig. 1b).

The model aims at the express simulation of interactions between annual integrals of external nutrient inputs, inter-basin water transports, and internal biogeochemical processes; hence, the model state variables represent annual averages. In each of the boxes, *DIN* and *DIP* represent dissolved inorganic nitrogen and dissolved inorganic phosphorus, *DSI*—dissolved silicate, *ON* and *OP* include nitrogen and phosphorus content in all the labile organic matter fractions—dead and alive, dissolved and particulate—while *ONS* and *OPS* represent dissolved refractory fractions of organic nitrogen and organic phosphorus. Benthic nitrogen *BEN* and benthic phosphorus *BEP* represent only the bioavailable fractions. *BSI* and *BES* represent biogenic silica in



(4), *GR* the Gulf of Riga (5), *DS* the Danish Straits (6), *KT* the Kattegat (7), *Ska* the Skagerrak boundary. **b** Variables and biogeochemical fluxes

the water column and bottom sediments, respectively. In BPd, the average oxygen (O_2) concentration OX is also modeled, as it is an important regulator of the nutrient cycles.

System of equations

Nutrient dynamics in all eight boxes ($i \neq j = 1, \dots, 8$) driven by continuous interaction of external inputs, water transports, and internal transformations are described by a system of ordinary differential equations:

$$\begin{aligned}
 \frac{dON_i}{dt} &= PP_i + NF_i - RegN_i - SedN_i \\
 &+ \frac{(L_{ON_i} + A_{ON_i} + \sum ON_j Q_{ji} - ON_i \sum Q_{ij})}{Vol_i} \\
 \frac{dONS_i}{dt} &= RegNS_i + \frac{(L_{ONS_i} + \sum ONS_j Q_{ji} - ONS_i \sum Q_{ij})}{Vol_i} \\
 \frac{dDIN_i}{dt} &= RegN_i + RegNS_i - PP_i + OutN_i \frac{Area_i}{Vol_i} \\
 &+ \frac{(L_{DIN_i} + A_{DIN_i} + \sum DIN_j Q_{ji} - DIN_i \sum Q_{ij})}{Vol_i} \\
 \frac{dOP_i}{dt} &= \frac{(PP_i + NF_i)}{a_{NP}} - RegP_i - SedP_i \\
 &+ \frac{(L_{OP_i} + A_{OP_i} + \sum OP_j Q_{ji} - OP_i \sum Q_{ij})}{Vol_i} \\
 \frac{dOPS_i}{dt} &= RegPS_i + \frac{(L_{OPS_i} + \sum OPS_j Q_{ji} - OPS_i \sum Q_{ij})}{Vol_i} \\
 \frac{dDIP_i}{dt} &= RegP_i + RegPS_i - UptP_i + OutP_i \frac{Area_i}{Vol_i} \\
 &+ \frac{(L_{DIP_i} + A_{DIP_i} + \sum DIP_j Q_{ji} - DIP_i \sum Q_{ij})}{Vol_i} \\
 \frac{dBSI_i}{dt} &= UptSi_i - RegSi_i - SedSi_i \\
 &+ \frac{(L_{BSI_i} + \sum BSI_j Q_{ji} - BSI_i \sum Q_{ij})}{Vol_i} \\
 \frac{dDSI_i}{dt} &= RegSi_i - UptSi_i + SedesSi_i \frac{Area_i}{Vol_i} \\
 &+ \frac{(L_{DSI_i} + \sum DSI_j Q_{ji} - DSI_i \sum Q_{ij})}{Vol_i} \\
 \frac{dBEN_i}{dt} &= SedN_i \frac{Vol_i}{Area_i} - SedesN_i - BurN_i \\
 \frac{dBEP_i}{dt} &= SedP_i \frac{Vol_i}{Area_i} - OutP_i - BurP_i \\
 \frac{dBES_i}{dt} &= SedSi_i \frac{Vol_i}{Area_i} - SedesSi_i - BurSi_i. \quad (1)
 \end{aligned}$$

In system (1), Vol_i and $Area_i$ are the water volumes and sediment areas of the corresponding

boxes; evidently there are no external loads to, or horizontal exchanges with, the BPd box ($i = 8$). Biogeochemical fluxes are abbreviated as follows:

| | |
|-----------------|---|
| PP | DIN uptake, equivalent to the net primary production; |
| NF | nitrogen fixation; |
| $UptP$ | DIP uptake that supports both PP and NF according to the Redfield N:P ratio a_{NP} ; |
| $UptSi$ | DSI uptake that supports $diat$ fraction of PP created by diatoms according to the Redfield N:Si ratio a_{NS} ; |
| $RegN (P, Si)$ | pelagic recycling of labile nitrogen, phosphorus, and silica; |
| $RegNS (PS)$ | pelagic recycling of refractory nitrogen and phosphorus; |
| $SedN (P, Si)$ | sedimentation of nutrients, in the case of BP sedimentation from the surface box is split into sedimentation onto the "surface" sediments ($i = 3$) and through the 60-m plane into the pelagic system of the deep box ($i = 8$); these fractions are proportional to the sediment areas in the Baltic Proper: $\frac{Area_3}{Area_3 + Area_8}$; respectively; |
| $SedesN$ | sediment nitrogen mineralization, which is split into denitrification ($SedeN$) and DIN release from the sediments to the water ($OutN$); |
| $OutP$ | DIP release from the sediments to the water as a fraction of sediment phosphorus mineralization ($SedesP$) that is not sequestered ($SeqP$) in the sediments; |
| $SedesSi$ | sediment silica dissolution; |
| $BurN (P, Si)$ | permanent burial from the upper "active" layer of sediments of nitrogen, phosphorus, and silica; |
| L_C and A_C | inputs of variable C from the land after coastal retention and from the atmosphere, respectively; |
| Q_{ij} | water flow from basin i to basin j ($Q_{ij} = 0$ for unconnected basins). |

In addition, concentrations of all pelagic variables in the surface (C_3) and deep (C_8) boxes of the Baltic Proper as well as in the Kattegat (C_7) change due to the following transport processes: deep-water inflow

(Q_{MAIN}) from the Danish Straits (C_6) generating an equal upwelling of deep waters, mixing across the 60-m plane in the Baltic Proper (Q_{MIX}), and the exchange with the Skagerrak occurring due to the water outflow and inflow (Q_{OUT} , Q_{IN}), the latter loaded with concentrations in the Skagerrak (C_{SK}):

$$\begin{aligned}\frac{dC_3}{dt} &= \dots + \frac{C_8 Q_{\text{MAIN}} + Q_{\text{MIX}}(C_8 - C_3)}{\text{Vol}_3} \dots \\ \frac{dC_8}{dt} &= \dots + \frac{C_6 Q_{\text{MAIN}} - C_8 Q_{\text{MAIN}} - Q_{\text{MIX}}(C_8 - C_3)}{\text{Vol}_8} \dots \\ \frac{dC_7}{dt} &= \dots + \frac{(C_{\text{SK}} Q_{\text{IN}} - C_7 Q_{\text{OUT}})}{\text{Vol}_7} \dots\end{aligned}\quad (2)$$

Parameterizations

The biogeochemical processes included are described according to the assumptions below.

DIN uptake depends on nutrient concentrations according to the Liebig minimum law with the limiting nutrient effects *eff* described by Michaelis–Menten kinetics. Assuming that the silicate-consuming diatoms contribute a *diat* fraction of the annual primary production, the corresponding term of the system (1) is given by the following expressions:

$$\begin{aligned}PP_i &= prmax_i [(1 - diat) \times \min(effn_i, effp_i) \\ &\quad + diat \times \min(effn_i, effp_i, effs_i)] \\ effn_i &= \frac{DIN_i}{hs_N + DIN_i}; \quad effp_i = \frac{DIP_i}{hs_p + DIP_i}; \\ effs_i &= \frac{DS_i}{hs_s + DS_i}\end{aligned}\quad (3)$$

that is, a net production annual rate *prmax_i* is basin specific, while half-saturation constants *hs_N*, *hs_p* and *hs_s* are basin invariant.

Nitrogen fixation depends on the N:P ratio and stoichiometric phosphorus surplus:

$$NF_i = rnf \frac{a_{NP} DIP_i - DIN_i}{1 + \left(\frac{DIN_i}{a_{NP} DIP_i}\right)^{10}}, \quad (4)$$

where *rnf* is a basin-invariant nitrogen fixation rate.

Evidently, no primary production and corresponding nutrient uptake are going on in the dark Bpd box (*i* = 8).

Sedimentation is proportional to concentrations and inversely depends on the average basin's depth *hz*:

$$\begin{aligned}SedN_i &= \frac{sink_i DON_i}{hz_i}; \quad SedP_i = \frac{sink_i DOP_i}{hz_i}; \\ SedSi_i &= \frac{2 \times sink_i BSi_i}{hz_i},\end{aligned}\quad (5)$$

where *sink_i* is a basin-specific sinking velocity; the sinking rate of biogenic silica is double that of organic nutrients.

Regeneration of nutrients is simulated as first-order reactions:

$$\begin{aligned}RegN_i &= wdesN_i ON_i; \quad RegP_i = wdesP_i OP_i; \\ RegSi_i &= wdesSi_i BSi_i \\ RegNS_i &= wdesNS_i ONS_i; \quad RegPS_i = wdesPS_i OPS_i \\ SedesN_i &= sdesN_i BEN_i; \quad SedesP_i = sdesP_i BEP_i; \\ SedesSi_i &= sdesSi_i BES_i\end{aligned}\quad (6)$$

with basin-specific mineralization rates *wdes* (in the water column) and *sdes* (by the sediments). Pathways of inorganic nutrients generated by the sediments differ. The basin-specific fraction of nitrogen mineralization flux *sden_i* is denitrified and leaves the system:

$$SedeN_i = sden_i SedesN_i \quad (7)$$

while the remainder is released into the water column:

$$OutN_i = SedesN_i - SedeN_i \quad (8)$$

The basin-specific fraction of phosphorus mineralization flux *seq_i* is sequestered in the sediments:

$$SeqP_i = seq_i SedesP_i \quad (9)$$

resulting in a phosphorus flux into the water column smaller than the mineralization flux:

$$OutP_i = (1 - seq_i) SedesP_i. \quad (10)$$

All the silica dissolved from the benthic opal is released into the water column.

All benthic nutrients are buried according to the basin-specific burial rates *bur_i*:

$$\begin{aligned}BurN_i &= bur_i BEN_i; \quad BurP_i = bur_i BEP_i; \\ BurSi_i &= bur_i BES_i.\end{aligned}\quad (11)$$

Nutrient dynamics in the Baltic Proper deep box are dependent on the modeled oxygen concentration *OX* that reflects the balance between “ventilation” by the transport processes similar to expressions (2) and

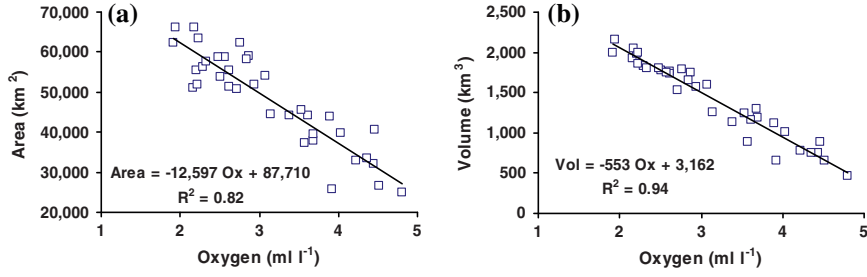


Fig. 2 Empirical relationships between the annual averages of oxygen concentrations in the Baltic Proper below 60 m in 1969–2004 and **a** anoxic sediment areas, and **b** hypoxic water volumes confined by waters containing less than 2 ml l⁻¹ oxygen

oxygen consumption by mineralization of organic matter including “oxygen reimbursement” (e.g., Savchuk & Wulff, 2001) by denitrification:

$$\begin{aligned} \frac{dOX}{dt} = & \frac{Q_{\text{MAIN}OX_{DS}} - Q_{\text{MAIN}OX} - Q_{\text{MIX}}(OX - ox_{BP_s})}{Vol_8} \\ & - alm(RegN_8 + RegNS_8) \\ & + ald \times Wdenitr \\ & - \frac{(alm \times SedesN_8 - ald \times SedeN_8)Area_8}{Vol_8}. \end{aligned} \quad (12)$$

In Eq. 12 ox_{DS} and ox_{BP_s} are the prescribed oxygen concentrations in waters flowing in from the Danish Straits and residing at the upper boundary of the halocline (60-m depth) in the Baltic Proper; alm and ald stand for oxygen demand for mineralization and “oxygen reimbursement” by denitrification.

This simulated oxygen concentration OX is used to compute, from the empirical relationships (Fig. 2), a hypoxic volume $HypVol$ containing waters with the oxygen concentration less than 2 ml l⁻¹ and an area of anoxic sediments covered by such waters $AnArea$, as well as dynamic ratios $denitr$ and $redox$ between these computed values and the total volume and area of the Baltic Proper deep box:

$$\begin{aligned} HypVol = f_{\text{vol}}(OX); \quad denitr = & \frac{HypVol}{Vol_8}; \\ AnArea = f_{\text{area}}(OX); \quad redox = & \frac{AnArea}{Area_8}. \end{aligned} \quad (13)$$

Assuming that the water denitrification $Wdenitr$ is occurring with a rate $wden$ only in the hypoxic volume, the corresponding differential equation from system (1) becomes

$$\frac{dDIN_8}{dt} = \dots - wden \times DIN_8 \times denitr. \dots$$

The nitrogen and phosphorus sediment fluxes (see Eqs. 7, 9) are then modified as follows:

$$SedeN_8 = sden_8 SedesN_8(1 - redox) \quad (14)$$

$$SeqP_8 = seq_8 SedesP_8(1 - redox). \quad (15)$$

For every year, the rates of all the processes described above are modified by the deviation $wtemp$ of the current annual average water temperature in each basin $T_i(t)$ from its long-term mean T_{mi} ; both annual values and means are prescribed from observations:

$$wtemp_i(t) = \frac{T_i(t)}{T_{mi}}. \quad (16)$$

As seen from these formulations, SANBALTS has a theoretical model foundation sensu, for instance, Ahlgren et al. (1988), because it is based on mass balances and first-order processes known on the basis of accumulated knowledge. However, SANBALTS also relies heavily on field data and empirical estimates by a tuning of basin-specific constants in basin-independent parameterizations of biogeochemical processes. The model was tuned in a hindcast for 1970–2003, a period for which large amounts of observational data are available. The majority of similar constants have been tuned in concert (Tables 1, 2) through a visual comparison between simulated and observed concentrations. For example, the rates of primary production were prescribed, taking into account inter-basin differences in the long-term means of the water temperature T_{mi} , the lengths of the phytoplankton growth season, and proportions of the 0–15 m (“euphotic”) layer of a basin’s total volume.

Table 1 Morphological characteristics of the model boxes and calibrated basin-specific constants

| Box (see Fig. 1a): | BB | BS | BPs | GF | GR | DS | KT | BPd |
|---|-------|-------|-------|-------|-------|-------|-------|-------|
| Volume (km ³) | 1,400 | 4,400 | 9,500 | 1,000 | 400 | 300 | 530 | 3,500 |
| Area (10 ³ km ²) | 36.5 | 76 | 115 | 29 | 16 | 20 | 22.5 | 100 |
| Depth (m) | 38 | 58 | 44 | 34 | 25 | 15 | 24 | 35 |
| <i>prmax</i> (mg m ⁻³ year ⁻¹) | 250 | 300 | 800 | 600 | 1,300 | 2,200 | 1,500 | |
| <i>sink</i> (m year ⁻¹) | 73 | 82 | 120 | 92 | 102 | 140 | 142 | 95 |
| <i>wdesN</i> (P) (year ⁻¹) | 2.5 | 4.5 | 8.5 | 7.5 | 10.0 | 16.0 | 15.5 | 4.0 |
| <i>wdesSi</i> (year ⁻¹) | 1.0 | 4.5 | 8.0 | 7.5 | 10.0 | 16.0 | 15.5 | 4.0 |
| <i>sdesN</i> (P) (year ⁻¹) | 0.20 | 0.22 | 0.33 | 0.25 | 0.28 | 0.45 | 0.30 | 0.15 |
| <i>sdesSi</i> (year ⁻¹) | 0.001 | 0.015 | 0.030 | 0.025 | 0.035 | 0.100 | 0.100 | 0.035 |
| <i>sden</i> | 0.20 | 0.35 | 0.20 | 0.60 | 0.55 | 0.35 | 0.45 | 0.8 |
| <i>seq</i> | 0.90 | 0.78 | 0.50 | 0.20 | 0.25 | 0.35 | 0.25 | 0.90 |
| <i>bur</i> (mm year ⁻¹) | 0.86 | 0.54 | 1.25 | 0.98 | 1.00 | 0.80 | 1.33 | 0.40 |

Table 2 Calibrated basin-invariant constants

| | | | |
|--|------|--|-------|
| <i>a_{NP}</i> (mg N mg ⁻¹ P) | 7.0 | <i>mf</i> (year ⁻¹) | 0.5 |
| <i>a_{NS}</i> (mg N mg ⁻¹ Si) | 0.5 | <i>wdesNS</i> (year ⁻¹) | 0.025 |
| <i>hs_N</i> (mg N m ⁻³) | 14.0 | <i>wdesPS</i> (year ⁻¹) | 0.050 |
| <i>hs_P</i> (mg P m ⁻³) | 2.0 | <i>wden</i> (year ⁻¹) | 3.0 |
| <i>hs_S</i> (mg Si m ⁻³) | 28.0 | <i>alm</i> (g O ₂ mg ⁻¹ N) | 0.02 |
| <i>diat</i> | 0.5 | <i>ald</i> (g O ₂ mg ⁻¹ N) | 0.003 |

The sinking and mineralization rates were set proportional to inter-basin differences in the long-term means of the water temperature; the burial rate was related to freshwater runoff, etc.

Driving forces

Long-term variations in the annual nutrient inputs to the basins from land were reconstructed from several sources. The main source of riverine inputs is the Baltic Environment Database (BED) based on an initial compilation of river loads by Stålnacke et al. (1999), and recently updated with data provided by several national monitoring agencies. Nutrient inputs from point sources were interpolated between specific years covered by the HELCOM periodic load compilations (e.g., HELCOM, 2004) and extrapolated backward assuming changes in the urban population and sewage water treatment. For the years 1994–2003, the unpublished HELCOM data on riverine and direct sources were used. Organic fractions of nitrogen and phosphorus, calculated as the difference

Table 3 Modifications of nutrient input from the land (see text for details)

| Box (see Fig. 1a): | BB | BS | BPs | GF | GR | DS | KT |
|------------------------------|------|------|------|------|------|------|------|
| Labile fraction of organic N | 0.20 | 0.25 | 0.30 | 0.25 | 0.35 | 0.40 | 0.40 |
| Labile fraction of organic P | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 | 0.9 |
| Proportion of biogenic Si | 0.05 | 0.10 | 0.15 | 0.10 | 0.10 | 0.15 | 0.15 |
| Coastal retention of N | 0.0 | 0.04 | 0.10 | 0.10 | 0.0 | 0.08 | 0.09 |
| Coastal retention of P | 0.0 | 0.07 | 0.13 | 0.2 | 0.0 | 0.0 | 0.0 |

between total amount and inorganic fractions, were split in the model into labile and refractory fractions, while the riverine input of biogenic silica (Conley, 1997) was assumed to be proportional to the silicate load (Table 3). Finally, the bioavailable (labile organic and inorganic fractions) inputs of nitrogen and phosphorus were subjected to some reduction due to coastal retention, initially estimated from field data (Sjöberg and co-workers, material presented in Savchuk, 2005) and modified during calibration (Table 3).

Annual atmospheric deposition of inorganic nitrogen was reconstructed from estimates made by Granat (2001) as well as from published and unpublished data from the Cooperative Programme for Monitoring and Evaluation of the Long-range Transmissions of Air Pollutants in Europe (e.g., Bartnicki et al., 2005).

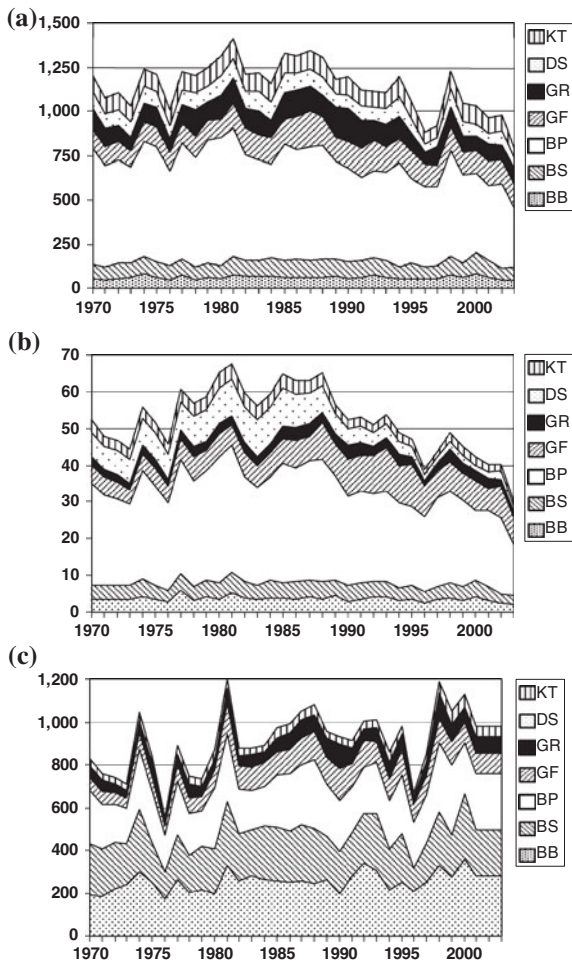


Fig. 3 a Total nitrogen, b phosphorus, and c silica inputs (10^3 t year $^{-1}$) from the land and atmosphere into the major basins of the Baltic Sea

Deposition of organic nitrogen was assumed to be 10% of the deposition of inorganic nitrogen. Annual phosphate input from the atmosphere was assumed constant and equal to $15 \text{ kg P km}^{-2} \text{ year}^{-1}$ for all basins (Savchuk, 2005 and references therein).

The combined nutrient inputs from all terrestrial and atmospheric sources for 1970 until 2003 are presented in Fig. 3. Elevated nitrogen input in the 1980s can be explained by greater atmospheric deposition (HELCOM, 1997) and riverine transports caused by a wetter climatic period (Bergström et al., 2001).

Time series of nutrient imports from the Skagerrak were prescribed as a product of annual water flows and average concentrations. Time series of annual

average concentrations were calculated with the SwingStations and DAS tools from the data in BED (Sokolov et al., 1997; Sokolov & Wulff, 1999) for the area $57^{\circ}46' - 57^{\circ}58' \text{N}$, $10^{\circ}13' - 11^{\circ}33' \text{E}$ down to 80 m; organic fractions are calculated as the difference between total and inorganic fractions and were split in the model into labile and refractory fractions similar to those in the Kattegat (see Table 3).

Time series of the annually and strait-wide integrated water flows between basins as well as vertical transports in the Baltic Proper have been adapted from long-term hydrodynamic simulations made for 1970–2003 by a hydrodynamic model with high temporal and vertical resolution (Gustafsson, 2003). These integral flows are comparable to other estimates obtained by different methods (Savchuk, 2005 and references therein). Implementation of these time series of integrated water flows within the SANBALTS settings results in a realistic simulation of salinity, considered a passive tracer (Fig. 4), and thus justifies a scaling-up approach that would allow a similar treatment and coupling of the climatic scenarios to biogeochemistry for hydrographic changes in the Baltic Sea.

The capability of SANBALTS to serve as a tool in studies of the large-scale Baltic nutrient biogeochemistry was also tested in a scenario simulation in which nutrient inputs were prescribed corresponding to a level that possibly existed a hundred years ago (Schernewski & Neumann, 2005; Savchuk et al., 2008), that is, well beyond the tuning conditions. In this “pre-industrial” scenario, all the annual values in the 1970–2000 forcing time series were reduced by constant factors of 0.3 and 0.25 for bioavailable fractions of N and P in the terrestrial loads, respectively, while atmospheric N and P depositions were reduced by 90%. The integral inputs of total nitrogen and phosphorus to the entire Baltic Sea were thus reduced on average by 62% and 73%, respectively. Starting from concentrations simulated for the year 2000 and keeping water flows and nutrient input from the Skagerrak unchanged, these time series were used recurrently four times to drive the model for 123 years.

Data for comparison

Time series of box-wide and annually averaged salinity, nutrient, and oxygen concentrations corresponding to SANBALTS integrations in space and time have been computed on 3D fields constructed for

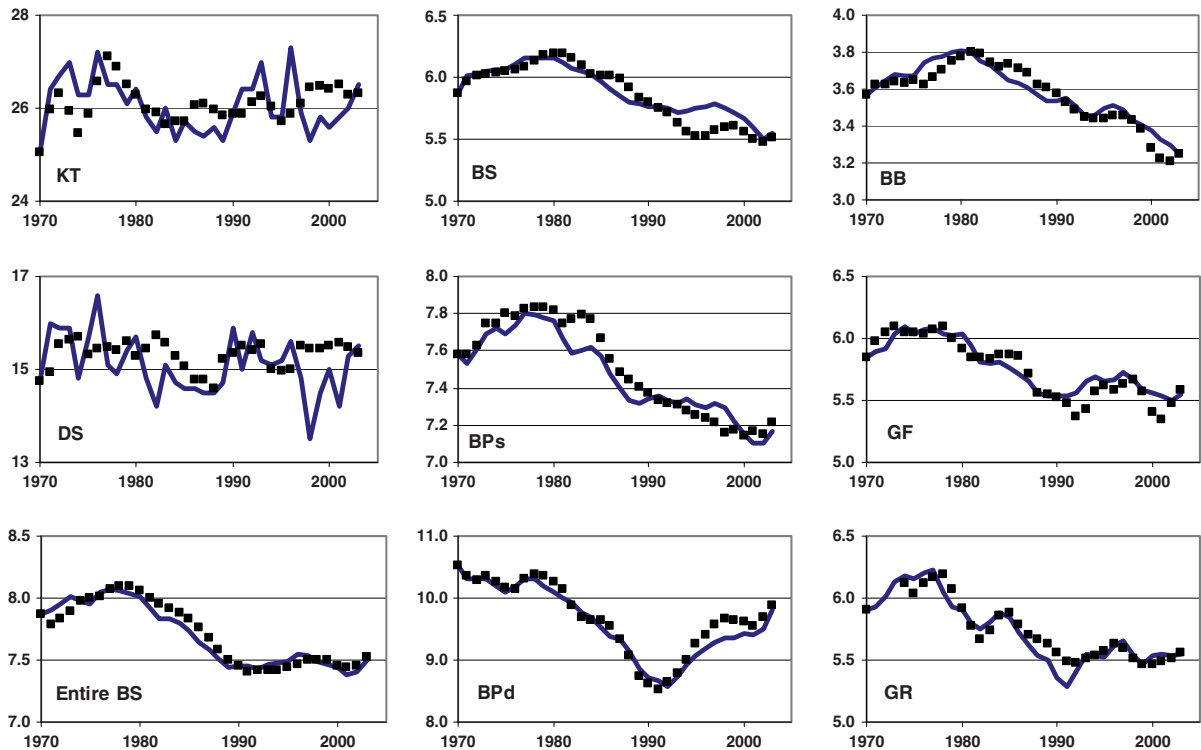


Fig. 4 Average salinity (psu) in the major basins of the Baltic Sea reconstructed from observations (*symbols*) and simulated by SANBALTS as a passive tracer (*curves*). Integrals for the

entire Baltic Sea are in the *left bottom panel*. Note the differences in the scales

the years from 1970 to 2003 over every basin by the DAS tool from data in BED (Sokolov et al., 1997). Although this technique has given the possibility to consistently utilize hundreds of thousands of oceanographic observations, some estimates for particular basins and years may be biased due to uneven spatial and seasonal data coverage. Even with such uncertainty, these time series represent an important quantitative record of the recent history of the Baltic Sea eutrophication (Fig. 5), complementary to earlier trend studies (e.g., Nehring, 1987; Andersson & Andersson, 2006; Papush & Danielsson, 2006) based on measurements at single stations and depths.

Results

Long-term dynamics of inorganic nutrient concentrations simulated by SANBALTS for 1970–2003 are presented in Fig. 5 together with annual means of appropriately aggregated and integrated observations.

A visually good match between model and reconstruction from measurement is confirmed by low values of the Theil's inequality index (Table 4) computed as:

$$TI = \frac{\sqrt{\sum(C_n - C_m)^2 / N}}{\sqrt{\sum C_n^2 / N} + \sqrt{\sum C_m^2 / N}}, \quad (17)$$

where C_m and C_n are simulated and re-constructed concentrations, respectively (Theil, 1961). The TI values for oxygen concentration, hypoxic area, and volume in the Baltic Proper deep layers are 0.058, 0.058, and 0.080, respectively. A significant discrepancy for phosphate in the Bothnia Bay is probably related to uncertainties in measurements of extremely low concentrations. Besides possible poor tuning, a weaker match in the Gulf of Finland and the Gulf of Riga can be caused by both shorter time series of marine observations and questionable values of the nutrient land loads, as has already been assumed by Savchuk & Wulff (2007).

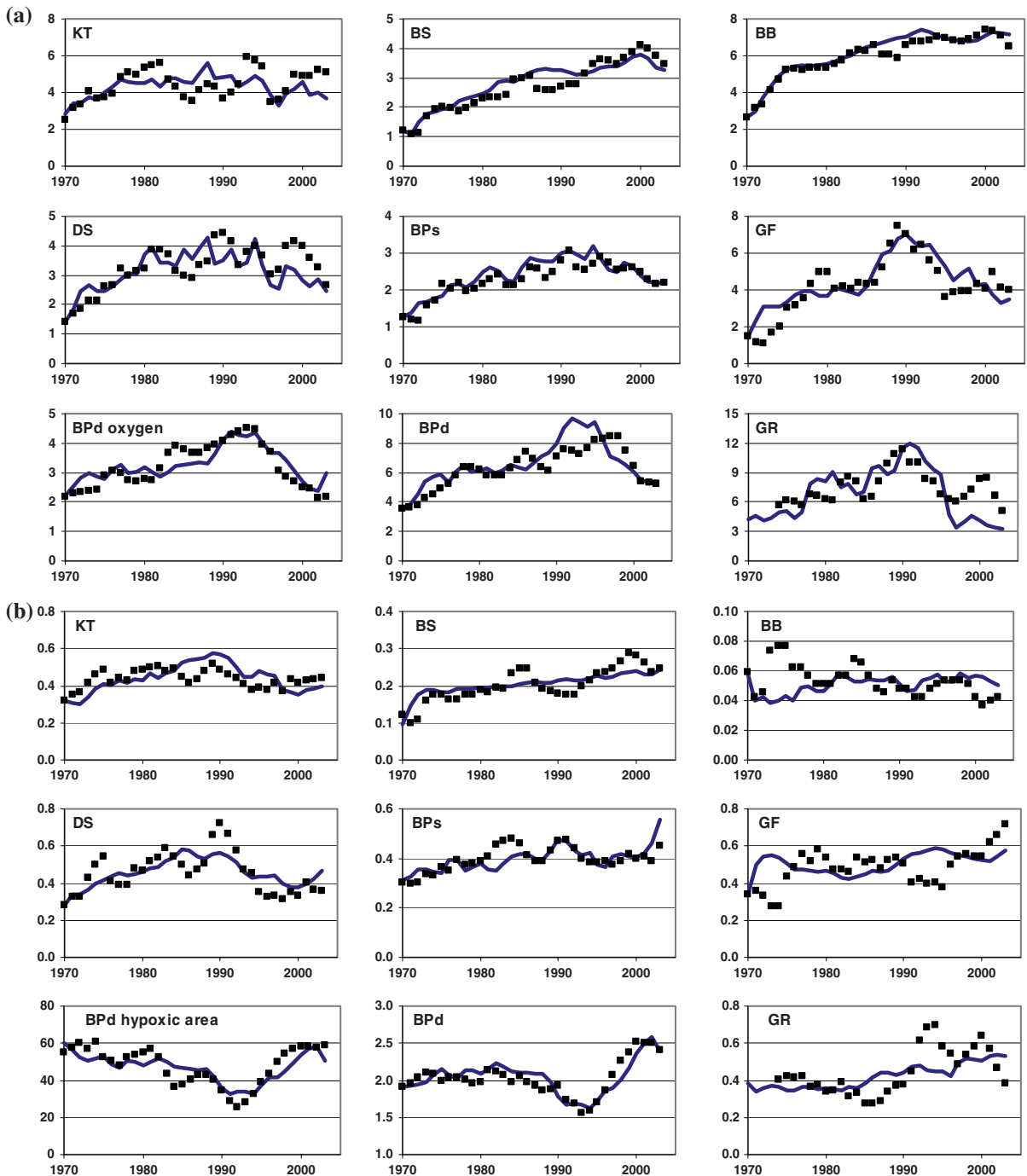


Fig. 5 Average concentration (μM) of **a** DIN , **b** DIP , and **c** DSI in the major basins of the Baltic Sea reconstructed from observations (*symbols*) and simulated by SANBALTS (*curves*). Reconstructed and simulated oxygen dynamics (ml l^{-1}) in the

Baltic Proper deep layers are also shown in (a) as well as variations of correspondent hypoxic areas (10^3 km^2) in (b) and volumes (10^3 km^3) in (c). Note the differences in the scales

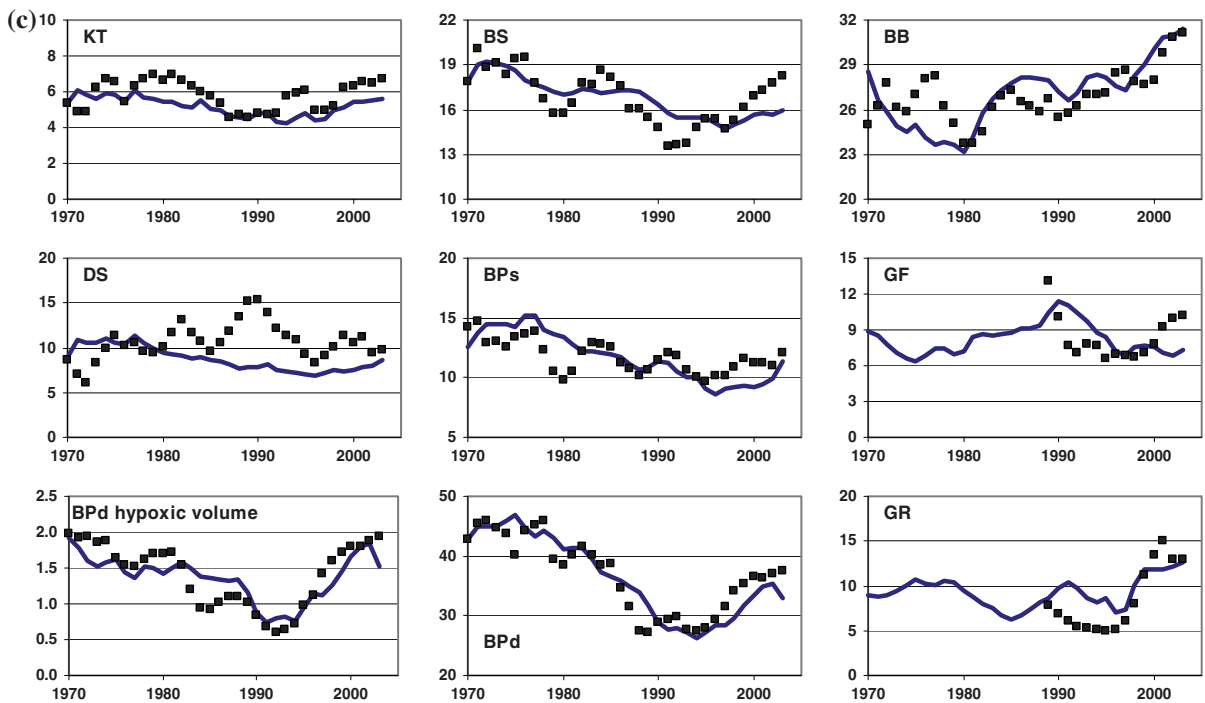


Fig. 5 continued

Table 4 Comparison of simulated to observed concentrations of dissolved inorganic nitrogen, phosphate, and silicate by Theil's index in the Baltic Sea basins

| Box (see Fig. 1a): | BB | BS | BPs | GF | GR | DS | KT | BPd |
|--------------------|-------|-------|-------|-------|-------|-------|-------|-------|
| <i>DIN</i> | 0.029 | 0.058 | 0.048 | 0.094 | 0.137 | 0.085 | 0.084 | 0.072 |
| <i>DIP</i> | 0.127 | 0.069 | 0.051 | 0.117 | 0.114 | 0.081 | 0.071 | 0.029 |
| <i>DSI</i> | 0.032 | 0.034 | 0.062 | 0.118 | 0.138 | 0.171 | 0.082 | 0.037 |

Results of a “pre-industrial” scenario simulation with drastically reduced nutrient inputs from the land and atmosphere are shown in Fig. 6. Generally, the transition into a new, oligotrophic state resembles that obtained in a simplified “flushing out” experiment with invariable forcing, including its temporal characteristics (Savchuk & Wulff, 2007). The expected quasi-cyclic pattern is caused by the recurrent forcing.

In order to determine whether a slow reaction could be an artifact produced by the aggregated SANBALTS model tuned for contemporary conditions, a more complex and process-oriented model was used. This model, BALTSEM, also describes the coupled nitrogen, phosphorus, and silica cycles in

horizontally integrated basins, but with a very high temporal and vertical resolution; ecosystem seasonal dynamics are reproduced through several planktonic variables (Gustafsson, 1999, 2003; Savchuk, 1999). Starting from initial concentrations simulated for the year 2000, BALTSEM was run for four 34-year cycles, that is, for 136 years under recurrent boundary conditions, in which terrestrial and atmospheric inputs of nutrients were reduced in a similar manner to that of the “pre-industrial” scenario implemented for SANBALTS. Examples of oxygen and nutrient dynamics simulated by BALTSEM for the Baltic Proper in a hindcast for 1970–2003 and for years 103–136 of the reduction scenario are presented in Fig. 7.

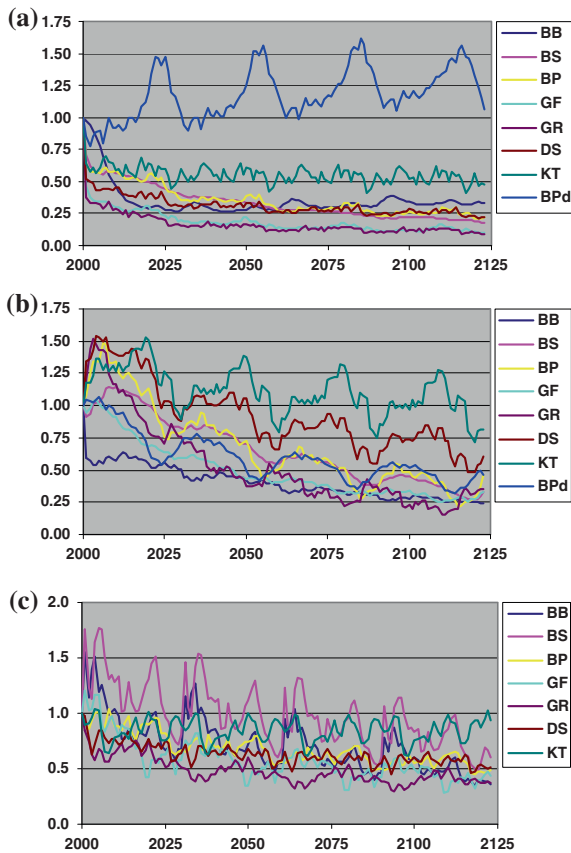


Fig. 6 Simulated development of **a** the *DIN* and **b** the *DIP* concentrations as well as of **c** the nitrogen uptake in the major basins of the Baltic Sea under reduced nutrient inputs shown as deviation from initial values of the year 2000

Discussion

Hindcast of the recent nutrient dynamics in the Baltic Sea

The simulated long-term dynamics of inorganic nutrients (see Fig. 5) demonstrate that the model is reasonably capable of reproducing the large-scale variations of nutrient pools in all the major Baltic Sea basins, including the variations caused by redox alterations in the Baltic Proper (see Table 4). However, these encouraging results may be partly due to the extensive tuning of many parameters in this model. Therefore, considering that external inputs and transport flows are more reliable (see Figs. 3, 4), independent estimates of biogeochemical fluxes are more important for the testing of this model than comparisons against concentrations. Unfortunately,

there are not many such estimates available even for the most studied processes, especially at annual and basin-wide scales. In fact, such scaling up is an important implementation of SANBALTS.

Primary production, converting inorganic nutrients into organic matter, is one of the most important determinants and indicators of the trophic status of aquatic systems (e.g., Nixon, 1995). The sum of simulated *DIN* uptake and nitrogen fixation, expressed in weight units, was multiplied by 6 for conversion to carbon (C) values and was compared to estimates compiled from different sources (Table 5). The simulated values correspond to net primary production, while measurements usually are somewhere between gross and net production rates (e.g., Sandberg et al., 2000). Within this uncertainty, simulated ranges compare well with published estimates.

The trophic status of the Baltic Sea, especially in its southern nitrogen-limited basins, depends on nitrogen availability that, in turn, is to a large extent determined by the balance between nitrogen fixation and denitrification. Nitrogen fixation is also important in other parts of the Baltic Sea because it can compensate to a great degree for the reductions in terrestrial nitrogen loads (Savchuk & Wulff, 1999; Vahtera et al., 2007). Generally, simulated inter-annual variations of nitrogen fixation are within or close to the ranges of existing estimates (Table 6). However, the maximum of $360 \times 10^3 \text{ t N year}^{-1}$ simulated for the Baltic Proper is less than half of the most recent maximum estimate of $792 \times 10^3 \text{ t N year}^{-1}$ (Wasmund et al., 2005).

Expressed in similar units, total denitrification estimated by Schneider et al. (2002) for the domain below 70 m in the Gotland Deep is lower than the denitrification modeled for the larger domain below 60 m in the Baltic Proper, as would be expected. Simulated denitrification in the water column of the Baltic Proper is within the range reported from two expeditions in the Gotland Deep by Brettar & Rheinheimer (1991).

Sediment denitrification has been tuned to reflect the inter-basin differences in productivity and sedimentation of organic matter (see Table 1). As a result, simulated rates of sediment denitrification are close to estimates obtained from scattered measurements (see Table 6), while total simulated denitrification of $1,067 \pm 70 \times 10^3 \text{ t N year}^{-1}$ compares reasonably well to a recent estimate, for a lesser area, of $855 \times 10^3 \text{ t N year}^{-1}$ (Voss et al., 2005).

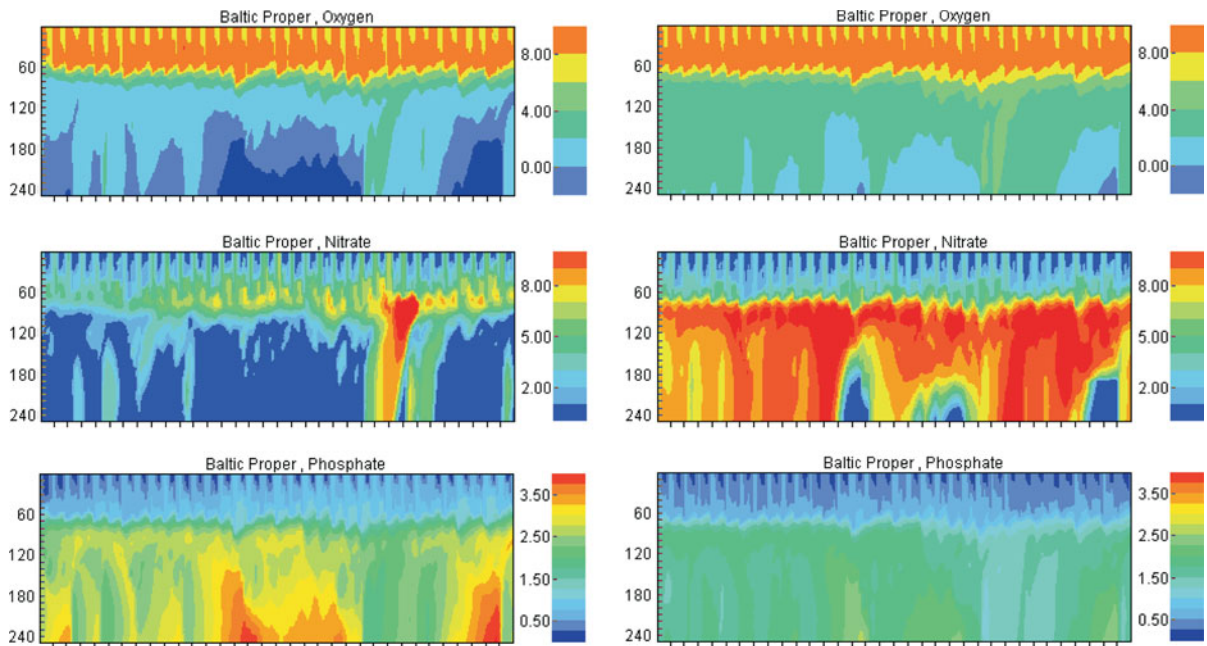


Fig. 7 Time–depth plots of oxygen (*top*, ml/l), nitrate (*middle*, $\mu\text{M N}$), and phosphate (*bottom*, $\mu\text{M P}$) concentrations in the Baltic Proper simulated by the BALTSEM model in a hindcast

for 1970–2003 (*left*) and a scenario for 103–136 years of running under reduced nutrient inputs (*right*). Y-axis in meters, X-axis in years

Table 5 Simulated primary production (mean \pm SD, $\text{g C m}^{-2} \text{year}^{-1}$) in comparison to published estimates

| Basin | BB | BS | BPs | GF | GR | DS | KT |
|-------------------|----------------------------------|-----------------------------------|------------------------------------|-----------------------------------|------------------------|----------------------|------------------------------------|
| PP _{sim} | 24 \pm 5 | 75 \pm 14 | 156 \pm 32 | 104 \pm 20 | 164 \pm 22 | 151 \pm 14 | 173 \pm 23 |
| PP _{est} | 17 ^a –28 ^b | 52 ^a –113 ^b | 165 ^b –200 ^a | 80 ^a –128 ^c | 200–260 ^{a,d} | 200–250 ^e | 150 ^f –200 ^g |

^a Wasmund et al. (2001a), ^b Sandberg et al. (2000), ^c Raateoja et al. (2004), ^d Savchuk (2002) and references therein, ^e Markager & Storm (2003), ^f Pearson & Rosenberg (1992), ^g Richardson & Heilmann (1995)

As these comparisons to both long-term nutrient concentration variations and available basin-wide estimates of the most important fluxes show, SANBALTS has been tuned well enough to reasonably reproduce the large-scale features of nutrient dynamics in the major Baltic Sea basins occurring under the contemporary nutrient inputs.

A scenario simulation

Generally, the responses of the Baltic Sea nutrient cycles to greatly reduced nutrient inputs (see Fig. 6) appear rather plausible and similar to those obtained under invariable driving forces (Savchuk & Wulff, 2007; Savchuk et al., 2008). The *DIN* concentrations were reduced rapidly within a few years in the

nitrogen-limited basins, from 30% to 40% of the present values in the Gulf of Finland and the Gulf of Riga and to about 60% in the Baltic Proper surface layer and in the Kattegat. In the phosphorus-limited Bothnian Bay, the initial transition was more gradual and a 60–70% reduction took about 20 years. In the Baltic Proper deep layers, the *DIN* concentration even started to increase after an initial reduction due to improved oxygen conditions that resulted in a smaller denitrification in the water column.

Correspondingly, due to a decreased demand for phosphorus, *DIP* concentrations initially increased in several basins by 20–40% compared to year 2000. The subsequent reductions of *DIP* were much slower than those of *DIN*. Similarly slow was the reduction of primary production, which in 50 years hardly

Table 6 Comparison of simulated and estimated biogeochemical fluxes in different basins

| Simulated by SANBALTS (mean \pm SD) | Independently estimated from measurements/models | Source |
|--|--|------------------------------|
| Nitrogen fixation (10^3 t N year ⁻¹) | | |
| 247 \pm 38 (BP) | 30–260 (BP) | Rahm et al. (2000) |
| | 180–430 (BP) | Larsson et al. (2001) |
| | 370 (BP) | Wasmund et al. (2001b) |
| | 434 (BP) | Wasmund et al. (2005) |
| Total denitrification (mmol N m ⁻² year ⁻¹) | | |
| 401 \pm 33 (BP below 60 m) | 220–340 (BP below 70 m) | Schneider et al. (2002) |
| Water denitrification (mg N m ⁻³ day ⁻¹) | | |
| 0.76 \pm 0.17 (BP, waters below 60 m) | 0.62–1.54 (Gotland Deep, 1986, 1987) | Brettar & Rheinheimer (1991) |
| Sediment denitrification (mmol N m ⁻² day ⁻¹) | | |
| 0.07 \pm 0.02 (BB) | | |
| 0.19 \pm 0.03 (BS) | 0.25–0.30 (BS) | Tuominen et al. (1998) |
| 0.24 \pm 0.03 (BPs) | 0.02–0.30 (northern BP) | Tuominen et al. (1998) |
| 0.50 \pm 0.09 (GF) | 0.16–0.28 (GF) | Conley et al. (1997) |
| | 0.15–0.65 (central GF) | Tuominen et al. (1998) |
| | 0.01–1.26 (eastern GF) | Gran & Pitkänen (1999) |
| 0.84 \pm 0.10 (GR) | 0.64–1.57 (GR) | Savchuk (2002) |
| 0.61 \pm 0.07 (DS) | 0.10–1.86 (Kiel Bight) | Kähler (1991) |
| 0.65 \pm 0.11 (KT) | | |

decreased by 50% anywhere, except in the Gulf of Riga. Even after 120 years the production stayed at relatively high levels. In addition to differences in nutrient limitation, such responses of the Baltic Sea basins to nutrient reductions are well explained by a larger openness of the nitrogen cycle due to high rates of internal sources and sinks (nitrogen fixation and denitrification) compared to the phosphorus cycle. This distinction between cycles is also reflected in shorter nitrogen residence times compared to those of phosphorus (e.g., Wulff et al., 1990; Savchuk, 2005), especially when the sediment nutrient pools are taken into account (Savchuk & Wulff, 2001; Savchuk, 2002).

Due to the recurrent forcing and similar to SANBALTS scenario simulation (see Fig. 6), the general patterns and successions of inter-annual variations are the same in both the hindcast and scenario BALTSEM simulations (see Fig. 7). The direction and result of the system transformation due to a large reduction of nutrient inputs are also quite similar to those simulated by the simpler SANBALTS. The nutrient pools have decreased in the surface layers, while in the deep layers *DIN* and *DIP*

concentrations increased and decreased, respectively, as a response to reductions in the occurrence, extension, and duration of hypoxic events.

Although the outputs from SANBALTS and BALTSEM are not fully comparable because of the differences in scales, domains, variables, and other settings, the responses of the two models can be compared in relative units. In that sense, the more complex model appears to retain even more nutrients in the system than the simpler one. At the end of the scenario simulations, the pools of bioavailable nitrogen and phosphorus in the water column were, respectively, 74% and 58% of the initial amount in the BALTSEM simulation versus 50% and 42% in the SANBALTS simulation.

In contrast, the nitrogen uptake in SANBALTS was less affected by nutrient reductions than in BALTSEM (Fig. 8). According to SANBALTS formulations, the only variation in forcing that directly determines the uptake of nitrogen is the relative deviation of annually averaged water temperatures and no account is taken of phenomena occurring at a seasonal scale such as the actual variations in duration of the vegetation season, depth and duration

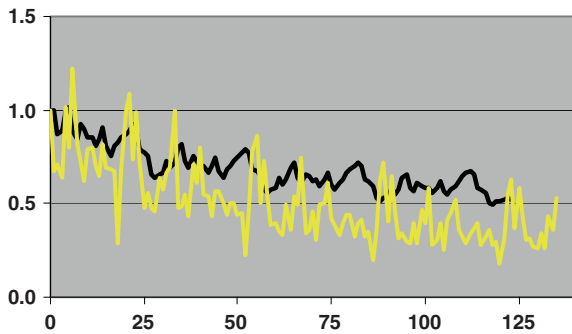


Fig. 8 Nitrogen uptake integrated for the entire simulated domain as produced by the BALTSEM (*thin line*) and SANBALTS (*thick line*) models in scenario simulations and presented as deviations from the first year

of vertical mixing, relative contribution of different phytoplankton variables, etc. Therefore, SANBALTS is predictably less responsive to inter-annual variations in driving forces than BALTSEM, where all these effects are explicitly accounted for.

These quantitative differences in the models' behavior in relation to very large changes in the boundary conditions call for further development of both models. However, the qualitative similarity in responses from the two quite different models, including a centennial scale of transition, indicates that even in its present state the rapid SANBALTS can serve as a useful tool for a quick evaluation of the general directions and magnitudes of large-scale basin-wide transformations.

Conclusion

A highly aggregated model, SANBALTS, is presented that describes coupled nitrogen, phosphorus, and silica cycles at annual and basin-wide scales in the major basins of the Baltic Sea. The model's reliability in terms of describing the contemporary Baltic Sea eutrophication is demonstrated in a comparison of a hindcast simulation (1970–2003) to observed concentrations and empirically estimated biogeochemical fluxes.

The capability of SANBALTS to work beyond the calibration conditions was tested in a scenario simulation with greatly reduced—to less than one-quarter—inputs of bioavailable nutrients. A comparison to a similar simulation conducted with a much more complex and process-oriented model (BALTSEM) has

shown some quantitative differences between the two models. After about 120–130 simulated years with reduced nutrient inputs, the BALTSEM simulation retained about 50% more nutrients in the Baltic compared to the SANBALTS simulation.

Despite these differences, the qualitative behavior of the two models was quite similar, and both models show that at least 130 years would need to elapse for the Baltic Sea to reach a new nutrient balance with reduced inputs.

Acknowledgments The authors appreciate the excellent work by Miguel Rodriguez Medina in maintaining and updating BED with data kindly provided over a number of years by many institutions and agencies, a list and acknowledgments of which are found at: <http://nest.su.se/bed/ACKNOWLEDGE.shtml>. The assistance by Pekka Kotilainen with HELCOM data and by Jerzy Bartnicki with EMEP data is greatly appreciated. Bo Gustafsson was very helpful in the extraction of integrated water flows from his original hydrodynamic model. This study was a part of the MARE project funded by the Swedish Foundation for Strategic Environmental Research (MISTRA).

References

- Ahlgren, I., T. Frisk & L. Kamp-Nielsen, 1988. Empirical and theoretical models of phosphorus loading, retention and concentration vs. lake trophic state. *Hydrobiologia* 170: 285–303.
- Andersson, P. M. & L. S. Andersson, 2006. Long Term Trends in the Seas Surrounding Sweden. Swedish Meteorological and Hydrological Institute, Reports of Oceanography, 32.
- Anonymous, 2001. Special issue 'Man and the Baltic Sea'. *Ambio* 30: 171–326.
- Bartnicki, J., A. Gusev, T. Berg & H. Fagerli, 2005. Atmospheric Supply of Nitrogen, Lead, Cadmium, Mercury and Lindane to the Baltic Sea in 2003. EMEP Centres Joint Report for HELCOM, EMEP/MSC-W Technical Report 3/2005. http://www.helcom.fi/environment2/hazsubs/EMEP/en_GB/emep2003/.
- Bergström, S., H. Alexandersson, B. Carlsson, W. Josefsson, K.-G. Karlsson & G. Westing, 2001. Climate and hydrology of the Baltic Basins. In Wulff, F., L. Rahm & P. Larsson (eds), *A Systems Analysis of the Baltic Sea*. Springer, New York: 75–110.
- Brettar, I. & G. Rheinheimer, 1991. Denitrification in the central Baltic: evidence for H₂S-oxidation as motor of denitrification at the oxic–anoxic interface. *Marine Ecology Progress Series* 77: 157–169.
- Conley, D. J., 1997. Riverine contribution of biogenic silica to the oceanic silica budget. *Limnology and Oceanography* 42: 774–777.
- Conley, D. J., A. Stockenberg, R. Carman, R. W. Johnstone, L. Rahm & F. Wulff, 1997. Sediment-water nutrient fluxes in the Gulf of Finland, Baltic Sea. *Estuarine Coastal and Shelf Science* 45: 591–598.

- Gran, V. & H. Pitkänen, 1999. Denitrification in estuarine sediments in the eastern Gulf of Finland, Baltic Sea. *Hydrobiologia* 393: 107–115.
- Granat, L., 2001. Deposition of nitrate and ammonium from the atmosphere to the Baltic Sea. In Wulff, F., L. Rahm & P. Larsson (eds), *A Systems Analysis of the Baltic Sea*. Springer, New York: 133–148.
- Gustafsson, B., 1999. Simulation of the long-term circulation of the Baltic Sea. Proceedings of the BASYS conference, Warnemünde, Germany: 147–153.
- Gustafsson, B., 2003. A time-dependent coupled-basin model of the Baltic Sea. Report C47. Earth Sciences Centre, Göteborg University.
- HELCOM, 1997. Airborne pollution load to the Baltic Sea, 1991–1995. *Baltic Sea Environment Proceedings* No. 69.
- HELCOM, 2004. The fourth Baltic Sea pollution load compilation (PLC-4). *Baltic Sea Environment Proceedings* No. 93.
- Kähler, P., 1991. Eutrophication and sediment denitrification in coastal marine waters, the example of Kiel Bight. *Kieler Meeresforschungen Sonderheft* 8: 112–116.
- Larsson, U., S. Hajdu, J. Walve & R. Elmgren, 2001. Estimating Baltic nitrogen fixation from the summer increase in upper mixed layer total nitrogen. *Limnology and Oceanography* 46: 811–820.
- Markager, S. & L. Storm, 2003. Miljøeffektvurdering for Havmiljøet Del 1: Empirisk modellering af miljøtilstanden i de åbne indre farvande. Rapport fra Institut for Miljøvurdering.
- Nehring, D., 1987. Temporal variations of phosphate and inorganic nitrogen compounds in central Baltic deep basins. *Limnology and Oceanography* 32: 494–499.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Papush, L. & Å. Danielsson, 2006. Silicon in the marine environment: dissolved silica trends in the Baltic Sea. *Estuarine, Coastal and Shelf Science* 67: 53–66.
- Pearson, T. H. & R. Rosenberg, 1992. Energy-flow through the SE Kattegat – a comparative examination of the eutrophication of a coastal marine ecosystem. *Netherlands Journal of Sea Research* 28: 317–334.
- Raateoja, M., J. Seppälä & H. Kuosa, 2004. Bio-optical modelling of primary production in the SW Finnish coastal zone, Baltic Sea: fast repetition rate fluorometry in Case 2 waters. *Marine Ecology Progress Series* 267: 9–26.
- Rahm, L., A. Jönsson & F. Wulff, 2000. Nitrogen fixation in the Baltic Proper – an empirical study. *Journal of Marine Systems* 25: 239–248.
- Richardson, K. & J. P. Heilmann, 1995. Primary production in the Kattegat – past and present. *Ophelia* 41: 317–328.
- Rönnerberg, C. & E. Bonsdorff, 2004. Baltic Sea eutrophication: area-specific ecological consequences. *Hydrobiologia* 514: 227–241.
- Sandberg, J., R. Elmgren & F. Wulff, 2000. Carbon flows in the Baltic Sea food webs – a re-evaluation using a mass balance approach. *Journal of Marine Systems* 25: 253–280.
- Savchuk, O. P., 1986. The study of the Baltic Sea eutrophication problems with the aid of simulation models. *Baltic Sea Environment Proceedings* 19: 51–61.
- Savchuk, O. P., 1999. Simulation of the Baltic Sea eutrophication. Proceedings of the BASYS Conference, Warnemünde, Germany: 108–122.
- Savchuk, O. P., 2002. Nutrient biogeochemical cycles in the Gulf of Riga: scaling up field studies with a mathematical model. *Journal of Marine Systems* 32: 253–280.
- Savchuk, O. P., 2005. Resolving the Baltic Sea into 7 sub basins: N and P budgets for 1991–1999. *Journal of Marine Systems* 56: 1–15.
- Savchuk, O. P. & V. V. Volkova, 1990. Forecasting the future state of the Baltic Sea eutrophication. Proceedings of the 17th Conference of Baltic Oceanographers, Norrköping, Sweden.
- Savchuk, O. & F. Wulff, 1999. Modelling regional and large-scale responses of Baltic Sea ecosystems to nutrient reductions. *Hydrobiologia* 393: 35–43.
- Savchuk, O. & F. Wulff, 2001. A model of the biogeochemical cycles of nitrogen and phosphorus in the Baltic. In Wulff, F., L. Rahm & P. Larsson (eds), *A Systems Analysis of the Baltic Sea*. Springer, New York: 373–415.
- Savchuk, O. & F. Wulff, 2007. Modeling the Baltic Sea eutrophication in a decision support system. *Ambio* 36: 141–148.
- Savchuk, O. P., F. Wulff, S. Hille, C. Humborg & F. Pollehne, 2008. The Baltic Sea a century ago – a reconstruction from model simulations, verified by observations. *Journal of Marine Systems* 74: 485–494.
- Schernewski, G. & T. Neumann, 2005. The trophic state of the Baltic Sea a century ago: a model simulation study. *Journal of Marine Systems* 53: 109–124.
- Schneider, B., G. Nausch, H. Kubsch & I. Peterson, 2002. Accumulation of total CO₂ during stagnation in the Baltic Sea deep water and its relationship to nutrient and oxygen concentrations. *Marine Chemistry* 77: 277–291.
- Sjöberg, S., F. Wulff & P. Wåhlström, 1972. Computer simulations of hydrochemical and biological processes in the Baltic. *Askö Laboratory Contributions* 1: 1–91.
- Sokolov, A. & F. Wulff, 1999. SwingStations: a web-based client tool for the Baltic environmental database. *Computers & Geosciences* 25: 863–871.
- Sokolov, A., A. A. Andrejev, F. Wulff & M. Rodriguez Medina, 1997. The data assimilation system for data analysis in the Baltic Sea. *Systems Ecology Contributions* 3: 1–66.
- Stålnacke, P., A. Grimvall, K. Sundblad & A. Tonderski, 1999. Estimation of riverine loads of nitrogen and phosphorus to the Baltic Sea, 1970–1993. *Environmental Monitoring and Assessment* 58: 173–200.
- Theil, H., 1961. *Economic Forecasts and Policy*. North-Holland Publishing Company, Amsterdam.
- Tuominen, L., A. Heinänen, J. Kuparinen & L. P. Nielsen, 1998. Spatial and temporal variability of denitrification in the sediments of the northern Baltic Proper. *Marine Ecology Progress Series* 172: 13–24.
- Vahtera, E., D. Conley, B. Gustafson, H. Kuosa, H. Pitkänen, O. Savchuk, T. Tamminen, N. Wasmund, M. Viitasalo, M. Voss & F. Wulff, 2007. Internal ecosystem feedbacks enhance nitrogen-fixing cyanobacteria blooms and complicate management in the Baltic Sea. *Ambio* 36: 186–193.
- Voss, M., K.-C. Emeis, S. Hille, T. Neumann & J. W. Dippner, 2005. Nitrogen cycle of the Baltic Sea from an isotopic

- perspective. *Global Biogeochemical Cycles* 19(GB3001): 1–15.
- Wasmund, N., A. Andrushaitis, E. Lysiak-Pastuszak, B. Müller-Karulis, G. Nausch, T. Neumann, H. Ojaveer, I. Olenina, L. Postel & Z. Witek, 2001a. Trophic status of the south-eastern Baltic Sea: a comparison of coastal and open areas. *Estuarine, Coastal and Shelf Science* 53: 849–864.
- Wasmund, N., M. Voss & K. Lochte, 2001b. Evidence of nitrogen fixation by non-heterocystous cyanobacteria in the Baltic Sea and re-calculation of a budget of nitrogen fixation. *Marine Ecology Progress Series* 214: 1–14.
- Wasmund, N., G. Nausch, B. Schneider, K. Nagel & M. Voss, 2005. Comparison of nitrogen fixation rates determined with different methods: a study in the Baltic Proper. *Marine Ecology Progress Series* 297: 23–31.
- Wulff, F. & A. Stigebrandt, 1989. A time-dependent budget model for nutrients in the Baltic Sea. *Global Biogeochemical Cycles* 3: 53–78.
- Wulff, F., A. Stigebrandt & L. Rahm, 1990. Nutrient dynamics of the Baltic Sea. *Ambio* 19: 126–133.
- Wulff, F., E. Bonsdorff, I.-M. Gren, S. Johansson & A. Stigebrandt, 2001a. Giving advice on cost effective measures for a cleaner Baltic Sea: a challenge for science. *Ambio* 30: 254–259.
- Wulff, F., L. Rahm & P. Larsson (eds), 2001b. *A Systems Analysis of the Baltic Sea*. Ecological Studies, Vol. 148. Springer, Berlin.
- Wulff, F., O. Savchuk, A. Sokolov, C. Humborg & C. M. Mörth, 2007. Management options and effects on a marine ecosystem: assessing the future of the Baltic. *Ambio* 36: 243–249.

Response of eutrophication in the eastern Gulf of Finland to nutrient load reduction scenarios

Oleg P. Savchuk · Tatjana R. Eremina ·
Alexey V. Isaev · Ivan A. Neelov

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 225–237.
DOI: 10.1007/s10750-009-9776-y © Springer Science+Business Media B.V. 2009

Abstract The trophic status of the eastern Gulf of Finland, where the largest Baltic metropolis St. Petersburg sits at the mouth of the largest Baltic river Neva, is elevated but existing recommendations on water protection measures are controversial. In this study, the effects of nutrient load reductions on this ecosystem were estimated with the aid of a three-dimensional coupled hydrodynamic-biogeochemical model. As a reference, the contemporary seasonal dynamics were simulated with nutrient inputs corresponding to the recent estimates of point and riverine sources. In order to eliminate the effects of natural inter-annual variations, the computations were run under recurrent annual forcing for 3 years, until quasi

steady-state seasonal dynamics were reached. Reasonable comparability of simulated concentrations and biogeochemical fluxes to available field estimates provides credibility to scenario simulations. These simulations show that substantial reductions of nutrient point sources in St. Petersburg would affect only the Neva Bay as the immediate receptor of treated sewage waters, where primary production could decrease by up to 20%. Eutrophication in the other parts of the Neva Estuary and in the entire eastern Gulf of Finland would change insignificantly owing to increased nutrient import from the offshore waters. Therefore, more significant changes can occur only via a reduction in nutrient pools in the open Gulf of Finland and the Baltic Proper, which would require a longer time.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

O. P. Savchuk (✉)
Baltic Nest Institute, Stockholm University,
10691 Stockholm, Sweden
e-mail: oleg@mbox.su.se

T. R. Eremina · A. V. Isaev
Russian State Hydrometeorological University,
Malookhtinsky pr., 98, 195196 Saint-Petersburg,
Russian Federation

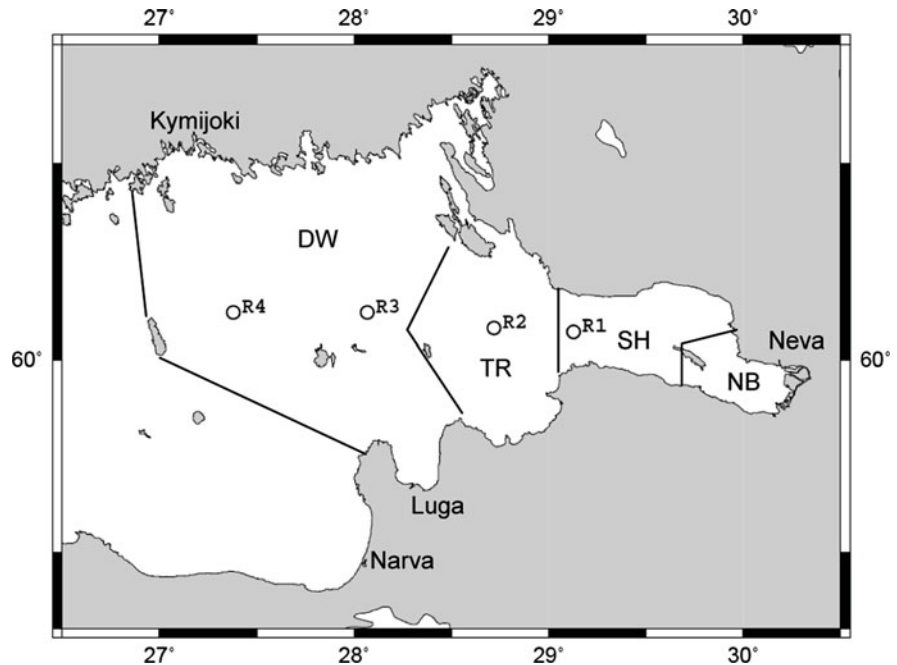
I. A. Neelov
Arctic and Antarctic Research Institute, Bering str. 38,
199397 Saint-Petersburg, Russian Federation

Keywords Eutrophication · Numerical model ·
Nutrient load reduction · Gulf of Finland

Introduction

The eastern Gulf of Finland (Fig. 1) comprises only 3% of the surface area and 1% of the volume of the Baltic Sea, but receives from its watershed over 10% of the total nitrogen and phosphorus inputs from land into the entire Baltic Sea (HELCOM, 2004). Of this load that entered directly into the Neva Bay in 2000, about 12% of nitrogen and 24% of phosphorus were generated by the largest Baltic metropolis, St.

Fig. 1 Map of the eastern Gulf of Finland with locations and delineations of the Russian monitoring stations (Rs), the Neva Bay (NB), the shallow (SH), transit (TR), and deep-water (DW) areas as well as the mouths of the rivers Kymijoki, Neva, Luga, and Narva; the background map is from <http://www.aquarius.geomar.de/omc>. The NB, SH, and TR areas together are considered as the Neva Estuary (Pitkänen, 1991)



Petersburg (approx. 5 million inhabitants). Consequently, the trophic status of this area is elevated (Pitkänen et al., 1993; Davidan & Savchuk, 1997; Leppänen et al., 1997). However, the development of scientifically sound recommendations for nutrient load reductions in this area is complicated by variations in limiting factors and estuarine nutrient-retention processes (Pitkänen, 1991; Pitkänen & Tamminen, 1995; Savchuk, 2000; Pitkänen et al., 2001). Several models have been implemented for this purpose, but their results are controversial.

From simulations with the EIA-SYKE three-dimensional ecosystem model, one modeling team anticipates that the better removal of nitrogen and, especially, phosphorus at the St. Petersburg wastewater treatment plants would result in about a 15–25% reduction in the annual average algal biomass in the Neva Estuary (Kiirikki et al., 2003; Pitkänen et al., 2007). However, results from the box version of our model have indicated that the easternmost Gulf of Finland acts as a purification facility and imports phosphorus from the open Gulf. Consequently, a reduction of the phosphorus load from St. Petersburg would increase this import, hardly improving the trophic status of the area (Davidan & Savchuk, 1997; Savchuk & Wulff, 1999; Savchuk, 2000).

Subsequently, the same biogeochemical module as in our box model was implemented within a fully three-dimensional model (Neelov et al., 2003). A six-year simulation made with a relatively high spatial resolution but under climatic boundary conditions was compared to the available data. The comparison showed that the model adequately reproduced typical seasonal dynamics in the surface layers but failed to simulate chronologically correctly the inter-annual variations in the deep layers caused by exact succession of year-to-year changes in the driving forces. The simulated biogeochemical dynamics are determined by continuous interactions of natural and anthropogenic drivers, effects that are extremely difficult to separate. In order to eliminate the effects of natural variations, the numerical experiments in this study have been run under recurrent boundary conditions until the quasi steady-state seasonal dynamics were reached.

Materials and methods

Details of the hydrodynamic and biogeochemical modules of the St. Petersburg Eutrophication Model (SPBEM) with all the necessary references are given

by Savchuk & Wulff (1996, 2001), Neelov et al. (2003), and Bashmachnikov et al. (2005). Here the authors briefly introduce the variables and processes accounted for in SPBEM.

In the hydrodynamic module, the ocean model is coupled to the sea ice model. The ocean model simulates the dynamics of three-dimensional fields of water velocity, temperature, and salinity as well as two-dimensional sea level variations. The intensity of vertical mixing is calculated from a local balance of the turbulent energy generated by the breaking of wind waves and the shear of the current velocity and damping by water stratification. The ice model describes seasonal evolution of the ice thickness and compactness. In this study, SPBEM is implemented for the entire Gulf of Finland with a horizontal grid step of 2 n.m., and a vertical resolution of 2 m in the layer of upper 30 m and 5 m below.

The biogeochemical module describes nitrogen and phosphorus cycling in the coupled pelagic and sediment sub-systems. The module contains eight pelagic (phytoplankton, zooplankton, detritus nitrogen and phosphorus, ammonium, nitrate, phosphate, and oxygen) and two sediment (bioavailable nitrogen and phosphorus) state variables. The variables interact by the following fluxes: nutrient utilization by phytoplankton including nitrogen fixation, zooplankton feeding and excretion, plankton mortality and sedimentation, mineralization of organic matter in the water column and by the sediments including denitrification and burial, photosynthetic aeration, and oxygen consumption.

Quasi steady-state seasonal dynamics intended to describe the contemporary trophic state of the Gulf of Finland were calculated with the following boundary conditions. Atmospheric forcing corresponding to 1999 was the same as that used in a previous experiment (Neelov et al., 2003). The boundary conditions at the entrance to the Gulf of Finland obtained from the simulation of the entire Baltic Sea were also taken from that experiment. Atmospheric depositions of ammonium ($0.70 \text{ mg N m}^{-2} \text{ d}^{-1}$) and nitrate ($0.76 \text{ mg N m}^{-2} \text{ d}^{-1}$) were the prescribed constants for the entire year except for June, when these fluxes were increased to 1.17 and $1.76 \text{ mg N m}^{-2} \text{ d}^{-1}$, respectively (Bartnicki et al., 2005). Atmospheric deposition of phosphorus was given as $0.045 \text{ mg P m}^{-2} \text{ d}^{-1}$ through the year (Savchuk, 2005 and references therein).

The entire Baltic Sea simulation used to generate boundary conditions was made with terrestrial nutrient loads given as monthly averages for 1986–1990 (Neelov et al., 2003). In the present Gulf of Finland simulation, all the nutrient inputs (Fig. 2) were assumed to be concentrated into four sources with locations corresponding to the mouths of the largest rivers, that is, Neva, Narva, Kymijoki, and Luga, all of which are situated in the eastern Gulf of Finland (see Fig. 1). Monthly mean (1996–2000) riverine loads were taken from the Baltic Environment Database (BED) that has recently been updated using data from the North-West Administration of Roshydromet (NW RHM) responsible for environmental monitoring. Organic fractions were calculated as the difference between total amounts and inorganic fractions. Loads from point sources corresponding to the year 2000 were given according to HELCOM (2004) data on total nitrogen and phosphorus, and further dividing these fluxes equally between organic and inorganic fractions. Finally, only 30% of organic nitrogen from every source and 65% of organic phosphorus from the St. Petersburg area were assumed to be bioavailable (e.g., Seitzinger et al., 2002; Stepanauskas et al., 2002). Thus, the annual terrestrial loads of bioavailable nitrogen and phosphorus inputs into the Gulf of Finland were prescribed as 66834 t and 5603 t, respectively, with 63% and 67% of these coming from the St. Petersburg area into the Neva Bay.

In order to approximate the trophic status typical for the end of the last century, the initial conditions for temperature, salinity, oxygen, and dissolved inorganic nutrients have been given as average winter (January–March 1998–2001) three-dimensional fields reconstructed by the Data Assimilation System (DAS) from observations in the Baltic Environment Database (Sokolov et al., 1997). Initial homogeneous fields for plankton and detritus variables were filled in with typical season-specific values.

Depth-dependent initial distributions of bioavailable nitrogen and phosphorus in the bottom sediments were given similar to the distributions obtained in simulations of the Baltic Proper (Savchuk & Wulff, 1996). Because the resuspension of sediments is not yet accounted for in SPBEM, simulated sediment nutrients are accumulating at the shallow depths, especially close to the river mouths, and are depleting slightly in the deepest grid cells. In order to avoid artificial changes in the bottom-water nutrient fluxes and

Fig. 2 Monthly inputs of bioavailable nitrogen (*left*) and phosphorus (*right*) into the Gulf of Finland from Finland, Russia, and Estonia used in SPBEM simulations

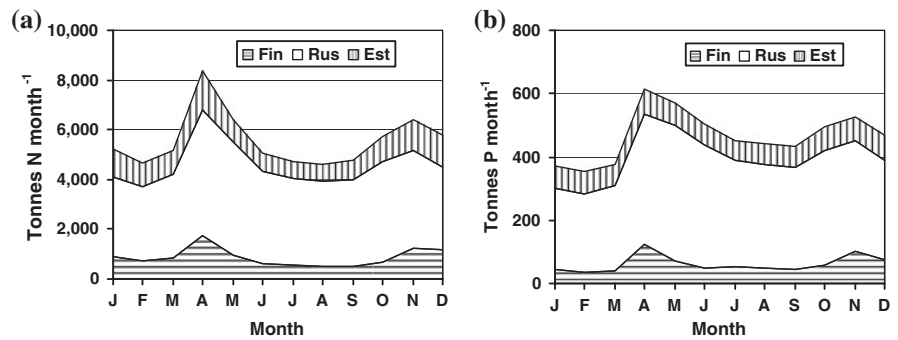
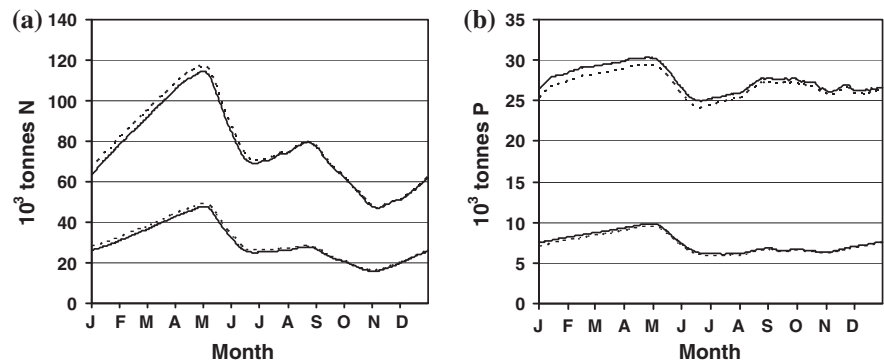


Fig. 3 Simulated seasonal dynamics of the integral water pools of nitrogen (*left*) and phosphorus (*right*) in the entire Gulf of Finland (*upper curves*) and its eastern part (*lower curves*) over the second (*dashed line*) and third (*solid line*) years of computation



maintain the sediment feedback at an approximately similar level in all the numerical experiments, the sediment nutrient distribution in the beginning of each consecutive year of simulation was restored to its initial state.

Results

Seasonal dynamics of nitrogen and phosphorus pools simulated during the third year of computations differ from the dynamics simulated during the second year by only a few percent in both the reference (Fig. 3) and scenario (not shown) experiments. In the reference run, further simulations over the fourth and fifth years make these differences even smaller and were deemed unnecessary considering the long time required for computations. Therefore, the following analysis was made on the results of the third year of both reference and scenario runs.

The reference run and model validation

Validation of the model by data from BED for the central offshore and north-eastern coastal areas

(Neelov et al., 2003) has shown that SPBEM realistically reproduced seasonal dynamics of nutrients and plankton in the surface layers. However, in the deep layers, simulated oxygen and phosphate concentrations were over- and underestimated, respectively. In this study, aimed mainly at the eastern Gulf of Finland, that analysis is supplemented by a comparison with data obtained in the Russian monitoring programme run by the NW RHM and collected in summer cruises of the Baltic Floating University. Unfortunately, all the observations were made only during the ice-free period and even after the spring phytoplankton bloom. Therefore, these data cannot directly serve for an explicit assessment of the nutrient pools as a target of nutrient reductions similar, for example, to the “winter nutrient concentrations” that are normally used to characterize accumulated nutrient reserves. As a proxy for such characteristics, the authors have used the near-bottom concentrations averaged over all the measurements made during summer and autumn (Table 1). A comparison of these averages to similarly processed modeled variables shows that the model realistically reproduces the phosphate levels observed in the deep layers of the eastern Gulf of Finland, while it

Table 1 Mean observed and simulated oxygen (ml l^{-1}), phosphate (mg P m^{-3}), and inorganic nitrogen (mg N m^{-3}) concentrations, and resulting inorganic N:P ratio (based on mass) in the eastern Gulf of Finland

| | R4 (50 m) | | R3 (40 m) | | R2 (30 m) | | R1 (20 m) | | Neva Bay | |
|-----------------|-----------|------|-----------|------|-----------|------|-----------|------|----------|------|
| | Obs. | Sim. | Obs. | Sim. | Obs. | Sim. | Obs. | Sim. | Obs. | Sim. |
| O ₂ | 4.1 | 5.4 | 4.2 | 5.5 | 4.8 | 6.4 | 6.2 | 6.9 | 7.1 | 7.9 |
| PO ₄ | 48 | 43 | 47 | 46 | 36 | 35 | 22 | 26 | 9.9 | 10 |
| DIN | 100 | 110 | 137 | 138 | 108 | 134 | 80 | 132 | 297 | 336 |
| N:P | 2.1 | 2.6 | 2.9 | 3.0 | 3.0 | 3.9 | 3.7 | 5.0 | 30.0 | 33.6 |

Note: Averaging was made for the indicated depths (in the shallow Neva Bay for the entire water body) over June–November of the third year of simulation and over measurements made at monitoring stations in June–November 1996–2002

overestimates the oxygen concentration, apparently because of insufficient vertical resolution. Simulated concentrations of dissolved inorganic nitrogen (DIN) are comparable to the concentrations observed in the deep area but are higher than those observed in the transit area and, especially, the shallow area. However, even these somewhat overestimated DIN reserves in the near-bottom layers cannot prevent the summer expansion of strong phosphorus limitation of the primary production westward, as shown by changes in the simulated spatial distribution of inorganic nutrients and the N:P ratio (Fig. 4). Although such annual persistence of estuarine phosphorus limitation differs from the seasonal changes of limiting nutrient (Conley, 2000), it is not unusual either in the Neva Estuary (Pitkänen & Tamminen, 1995) and other Baltic coastal areas (e.g., Meeuwig et al., 2000) or elsewhere (e.g., Nedwell et al., 1999; Howarth & Marino, 2006).

Spatial and temporal dynamics of concentration and biomass are governed by interactions between the physical water transports and the biogeochemical translocations of nutrients among the ecosystem variables. Because the model is driven by the recurrent water circulation, the ecosystem's response to perturbations in external impacts is determined by the changes in biogeochemical processes. Unfortunately, there are only a few field studies of these processes in the eastern Gulf of Finland, and the available information is even more fragmented than measurements of concentrations.

Simulated dynamics of the most important biogeochemical fluxes in the eastern Gulf of Finland are exemplified in Fig. 5 and summed up in Table 2. The net primary production, equivalent to the phytoplankton nitrogen uptake, increases sharply in the last week

of April and reaches its peak level of 1.5–2.5 $\text{g C m}^{-2} \text{d}^{-1}$ at the beginning of May, a few days before the culmination of the spring phytoplankton bloom. Later, the production drops abruptly due to nutrient limitation but increases again along with the development of the heterotrophic community that intensively recycles nutrients in the water column. At the western edges of the deep area, such nutrient recycling in June–August is supplemented by nitrogen fixation and the production stabilizes at 0.2–1.2 $\text{g C m}^{-2} \text{d}^{-1}$. In September, the production decreases owing to both the cessation of nitrogen fixation and the reduction of nutrient excretion by heterotrophs, but increases again in October to 0.5–0.9 $\text{g C m}^{-2} \text{d}^{-1}$ because of the increased admixing of nutrients from the deep layers. The plankton dynamics also determine the seasonal development of nutrient sedimentation with its spring and autumn maxima divided by a summer minimum. As a result, about half the nutrients—transformed from the dissolved to the particulate fraction by phytoplankton—eventually end up in the sediments (see Table 2).

Both seasonal variations and annual integrals of simulated primary production compare well with existing estimates based on scattered measurements (Davidan & Savchuk, 1997; Telesh et al., 1999; Golubkov et al., 2005), including a doubling of summer rates in the shallow area compared to the deep area (Pitkänen et al., 1993). Simulated nitrogen fixation also appears to be rather realistic both in comparison to measurements in the open Baltic Proper of 11–158 $\text{mg N m}^{-2} \text{d}^{-1}$ (Wasmund et al., 2001) and by its limited eastward extension (Kahru et al., 2000). Ranges of simulated sedimentation rates of 20–40 $\text{mg N m}^{-2} \text{d}^{-1}$ and 3–7 $\text{mg P m}^{-2} \text{d}^{-1}$ are fully comparable to the ranges estimated from

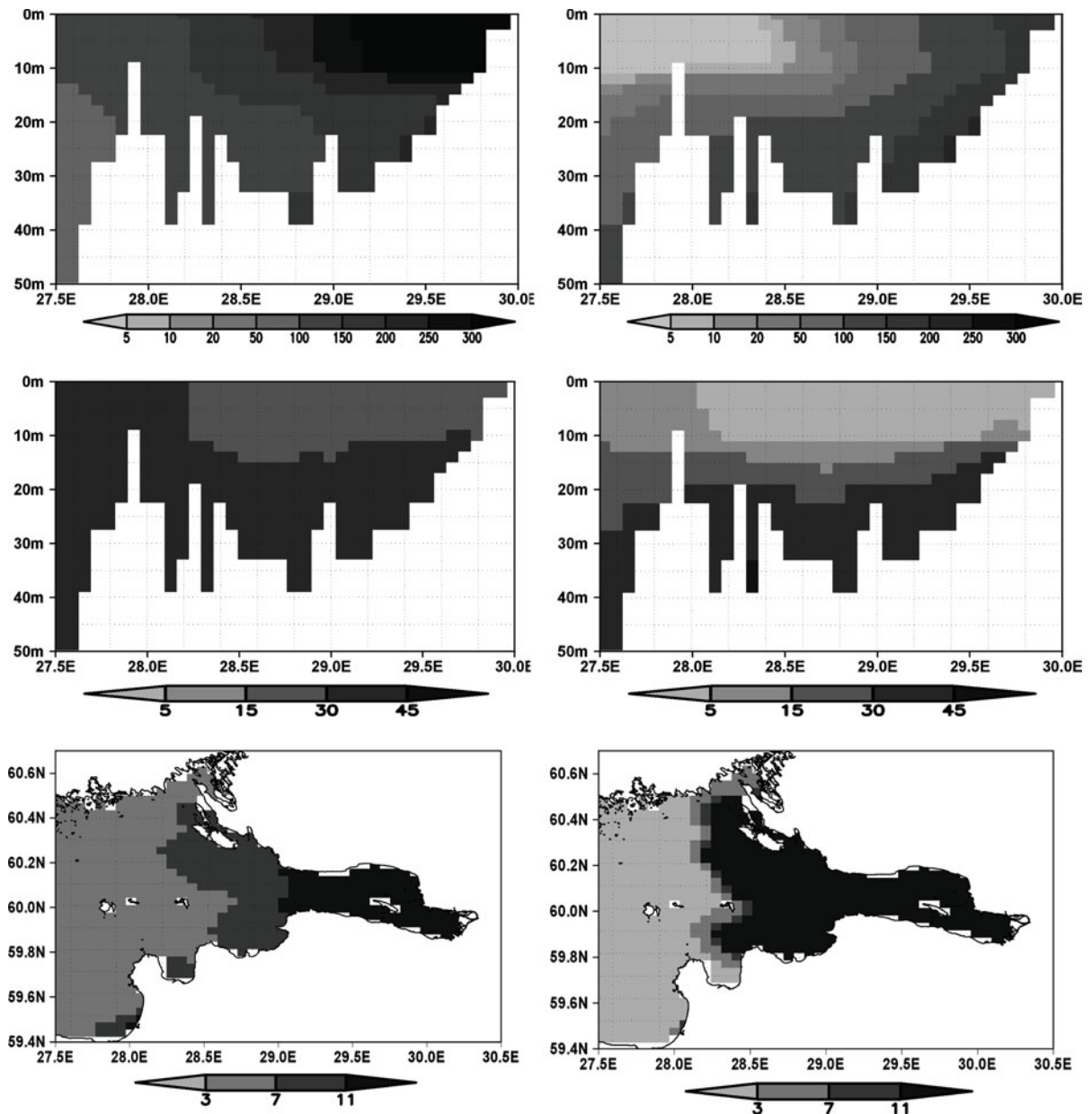


Fig. 4 Simulated distributions of inorganic nitrogen (mg N m^{-3} , *top*) and phosphorus (mg P m^{-3} , *middle*) at the cross section along $60^{\circ}05'N$, and the corresponding N:P ratio

(based on mass, *bottom*) in the surface layer before the spring bloom (on April, 1 *left*) and in the summer (on June, 30 *right*)

sediment traps deployed in the eastern Gulf of Finland in August 1992 and 1995 (Lehtoranta et al., 2004).

Simulated nitrogen and phosphorus release from the bottom sediments into the water column varies within a range of $10\text{--}20 \text{ mg N m}^{-2} \text{ d}^{-1}$ and $2\text{--}5 \text{ mg P m}^{-2} \text{ d}^{-1}$ (see Fig. 5); these ranges are in a good

agreement with the rates measured at the aerobic sediment surface (Lehtoranta, 1998; Pitkänen et al., 2001; Kiirikki et al., 2006). The biochemical oxygen consumption by the model sediments does not exceed $600\text{--}700 \text{ mg O}_2 \text{ m}^{-2} \text{ d}^{-1}$, i.e., assuming a respiratory quotient equal to $3.5 \text{ mg O}_2 \text{ mg}^{-1} \text{ CO}_2$, it is well below $240 \text{ mg C m}^{-2} \text{ d}^{-1}$, which was suggested by

Fig. 5 Simulated seasonal dynamics of biogeochemical fluxes in the deep (St. R4, *left*) and transit (St. R2, *right*) areas of the eastern Gulf of Finland. *Top panels:* PP Primary production, NF nitrogen fixation, and SN sedimentation; *bottom panels:* Den sediment denitrification, and NR release of nitrogen, and PR phosphorus into the sediments into the water column. Note the different scales

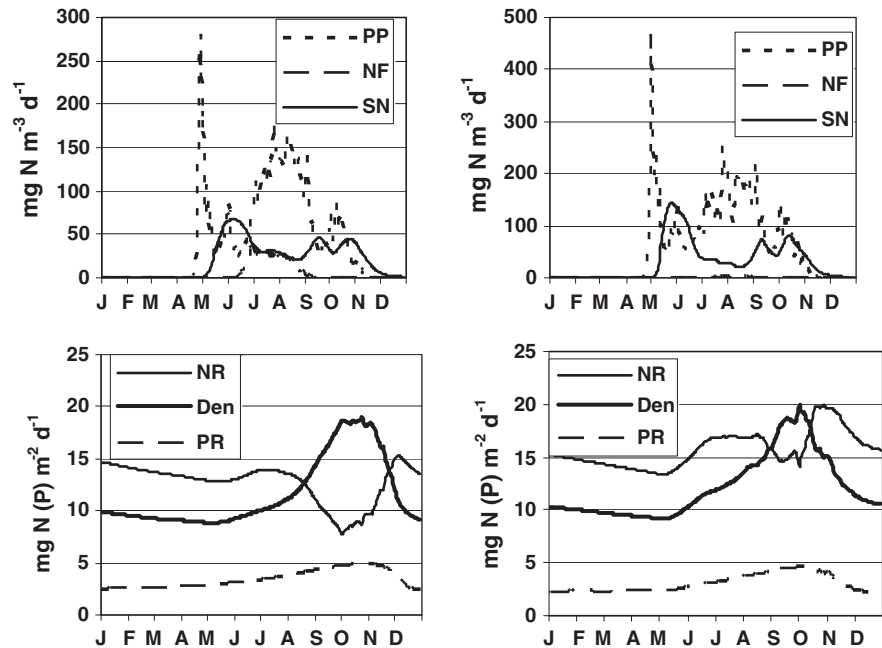


Table 2 Annual integrals of simulated biogeochemical fluxes in the eastern Gulf of Finland

| Fluxes | R4 | R3 | R2 | R1 | Neva Bay |
|---|------|------|------|------|----------|
| Food assimilation (g C m ⁻²) | 55 | 50 | 76 | 82 | 25 |
| Primary production (g C m ⁻²) | 95 | 103 | 136 | 162 | 161 |
| Nitrogen fixation (g N m ⁻²) | 1.75 | 0.59 | 0.11 | 0.12 | 0.04 |
| N recycling (g N m ⁻²) | 9.0 | 8.1 | 12.2 | 13.2 | 4.9 |
| P recycling (g P m ⁻²) | 1.3 | 1.2 | 1.8 | 1.9 | 0.96 |
| N sedimentation (g N m ⁻²) | 7.5 | 8.7 | 11.3 | 13.0 | 57 |
| P sedimentation (g P m ⁻²) | 1.1 | 1.2 | 1.6 | 1.8 | 12 |
| N release from the bottom (g N m ⁻²) | 4.6 | 5.2 | 5.7 | 5.7 | 10 |
| P release from the bottom (g P m ⁻²) | 1.3 | 1.2 | 1.1 | 1.0 | 1.6 |
| Oxygen demand (g O ₂ m ⁻²) | 265 | 331 | 325 | 419 | 438 |
| Sediment denitrification (g N m ⁻²) | 4.3 | 4.2 | 4.4 | 4.4 | 7.1 |

Notes: Locations are shown in Fig. 1; conversion from nitrogen to carbon units was made using a factor of 6; primary production represents the sum of DIN uptake and nitrogen fixation; recycling represents the sum of zooplankton excretion and mineralization of detritus; sedimentation represents the sum of phytoplankton and detritus deposited from the bottom layer on the sediment surface; oxygen demand comprises consumption both in the water column and by the sediments

Kiirikki et al. (2006) as a critical point for the redox alteration of the biogeochemical fluxes. The simulated rates of sediment denitrification (10–20 mg N m⁻² d⁻¹) are higher than the rates reported from most measurements in muddy sediments of the Gulf of Finland (2–9 mg N m⁻² d⁻¹, Tuominen et al., 1998; Kiirikki et al., 2006), but are comparable to the upper limits reported from the eastern Gulf of Finland

(15–18 mg N m⁻² d⁻¹, Gran & Pitkänen, 1999), as well as to rates up to 25 mg N m⁻² d⁻¹ measured in sandy sediments of the Kiel Bight (Kähler, 1991). The sensitivity analysis has shown that a reduction in simulated sediment denitrification would result in an elevated water nitrogen pool that already now seems overestimated in the easternmost areas compared to available observations (see Table 1).

Nutrient reduction scenarios

Three rather radical scenarios of reduction in the anthropogenic nutrient load have been studied. In a “P-scenario”, the phosphorus load from St. Petersburg point sources was reduced by 90%, i.e., the total annual input to the Neva Bay was decreased by 25%, from 3751 t to 2851 t. In an “N-scenario”, considering that all the nitrogen coming in from the point sources is bioavailable, the input was reduced by 75%, i.e., the annual input of bioavailable nitrogen to the Neva Bay was decreased by 15%, from 42 360 t to 36 006 t. In an “N&P-scenario”, both nitrogen and phosphorus inputs were simultaneously reduced by the amounts indicated above.

A comparison of the annual and area integrals of biogeochemical fluxes simulated over the third year of reference and scenario runs shows a rather insignificant response of the eutrophication in the eastern Gulf of Finland to these substantial reductions of nutrient inputs (Table 3). Relative changes in the primary production, often considered as the essential indicator of eutrophication (Nixon, 1995), are readily explained by spatial gradients of limiting factors but do not exceed a few percent anywhere except in the Neva Bay. The nutrient sedimentation and regeneration rates are tightly related to the primary production and also change little outside the Neva Bay. In fact, in the offshore waters these changes are so minuscule that they would hardly be detected by an analysis of the irregular rate measurements and infrequent monitoring observations made in the region.

The rates of nitrogen fixation are the most susceptible to alterations in the nutrient loads, although these changes are quantitatively important only in the deep area, because eastward of this area

the rates themselves are negligibly small (see Table 2). In the deep area, the nitrogen fixation increases by 24% in the N-scenario and decreases by 12% in the P-scenario, while combined nitrogen and phosphorus reductions result in a 12% increase of nitrogen fixation, similar to scenario simulations for other regions of the Baltic Sea (Savchuk & Wulff, 1999; Savchuk, 2002). Thus, the benefits of reduced primary production in the transit area should be weighed against the disadvantages of increased nitrogen fixation as a proxy of cyanobacteria blooms in the deep area.

Discussion

There are several important features of the Gulf of Finland eutrophication revealed and/or highlighted by our study.

The relatively short time period of the model adaptation to recurrent boundary conditions is consistent with the relevant temporal characteristics of the Gulf of Finland that are determined by its intensive exchange with the Baltic Proper across the wide and deep entrance (Perttilä et al., 1995; Perttilä & Savchuk, 1996; Alenius et al., 1998). The highest water ages of 2 years estimated with a different three-dimensional hydrodynamic model were found in the southeastern part of the Gulf, while the water-age distribution after a 5-year simulation was almost the same as that already achieved in 3 years (Andrejev et al., 2004). Residence times of 2.0, 1.4, and 1.3 years were estimated from the long-term empirical budgets for water, total nitrogen, and total phosphorus, respectively (Savchuk, 2005). On the other hand, a short spin-up time found also in the scenario computations can partly be explained by the

Table 3 Relative differences (%) of annual-area integrals of primary production, nitrogen fixation, nitrogen sedimentation, and nitrogen recycling between scenario and reference simulations in different areas of the eastern Gulf of Finland

| | Deep | | | Transit | | | Shallow | | | Neva Bay | | |
|----|------|------|------|---------|------|------|---------|------|------|----------|------|-----|
| | P | N | P&N | P | N | P&N | P | N | P&N | P | N | P&N |
| PP | 0.3 | -0.5 | -1.3 | -0.2 | -5.4 | -3.6 | -4.3 | 0.8 | -3.4 | -20 | 0 | -21 |
| NF | -12 | 24 | 12 | -23 | 108 | 38 | -18 | 18 | 0 | - | - | - |
| SN | 0.4 | -1.7 | -1.7 | 0.3 | -4.3 | -3.4 | -4.4 | -0.1 | -4.4 | -7.8 | -9.2 | -17 |
| RN | -0.1 | 0.6 | -1.0 | -0.9 | -5.6 | -3.7 | -5.6 | 1.2 | -4.6 | -20 | -3.6 | -24 |

PP Primary production, NF nitrogen fixation, SN nitrogen sedimentation, RN nitrogen recycling

basic set-up of numerical experiments, where the long-term changes both in the waters of the Baltic Proper and in the sediments of the Gulf of Finland have been disregarded.

As appears from the comparisons of simulated to estimated biogeochemical fluxes, the seasonal dynamics of major biogeochemical fluxes are reproduced by SPBEM well enough for a confident consideration of the annual integrals of simulated fluxes. For example, simulated sediment retention of nitrogen and phosphorus (Fig. 6) nicely demonstrates the phenomenon of estuarine nutrient removal known for this and other areas from empirical and modeling studies (e.g., Pitkänen & Tamminen, 1995; Pertilä & Savchuk, 1996; Davidan & Savchuk, 1997; Prastka et al., 1998; Nedwell et al., 1999; Savchuk, 2000; Witek et al., 2003; Lehtoranta et al., 2004). Further integration of such fluxes and water flows over selected areas and boundary cross sections (see Fig. 1) allows localizing and quantifying the nutrient retention capacity of the easternmost Gulf of Finland (Fig. 7).

In the model, the Neva Estuary retains about three-quarters of the bioavailable nitrogen coming in from the St. Petersburg region, with a quarter retained in the Neva Bay and a third in the transit area. In the case of phosphorus, total retention exceeds the terrestrial load by 25%; the “deficit” is covered by the phosphorus import from the offshore waters, mainly during the algal growth season. Almost half of

this retention occurs in the shallow Neva Bay (average depth less than 4 m) due to massive sedimentation of particulate phosphorus; this is mostly allochthonous (see Table 2) because phosphorus detritus constitutes about two-thirds of the total prescribed input. Although such a high fraction of particulate phosphorus correspond to observations both in the Neva River and the Neva Bay (Davidan & Savchuk, 1997), the integral phosphorus retention in this area can be somewhat exaggerated because neither the transport of resuspended particles nor the exchange between particulate and dissolved fractions are accounted for in this model. Qualitatively, these results fully agree with earlier conclusions, although the estimates of nutrient retention are quantitatively slightly different from the previous estimates (Savchuk & Wulff, 1999; Savchuk, 2000). The very existence of the net phosphorus import into the Gulf of Finland from the Baltic Proper had been implied earlier (Pertilä et al., 1995; Kahru et al., 2000; Pitkänen et al., 2001) and its annual integral of 2934 t simulated in this study is well within the range of 1000–11 500 t yr⁻¹ estimated from empirical budgets and mathematical model (Pertilä et al., 1995; Savchuk, 2005; Kiirikki & Pitkänen, 2007).

Qualitatively, a weak response of the eutrophication indicators to considerable nutrient reductions can be explained by the high buffer capacity of the eastern Gulf of Finland owing to the geographical openness of this area to compensatory transports. The

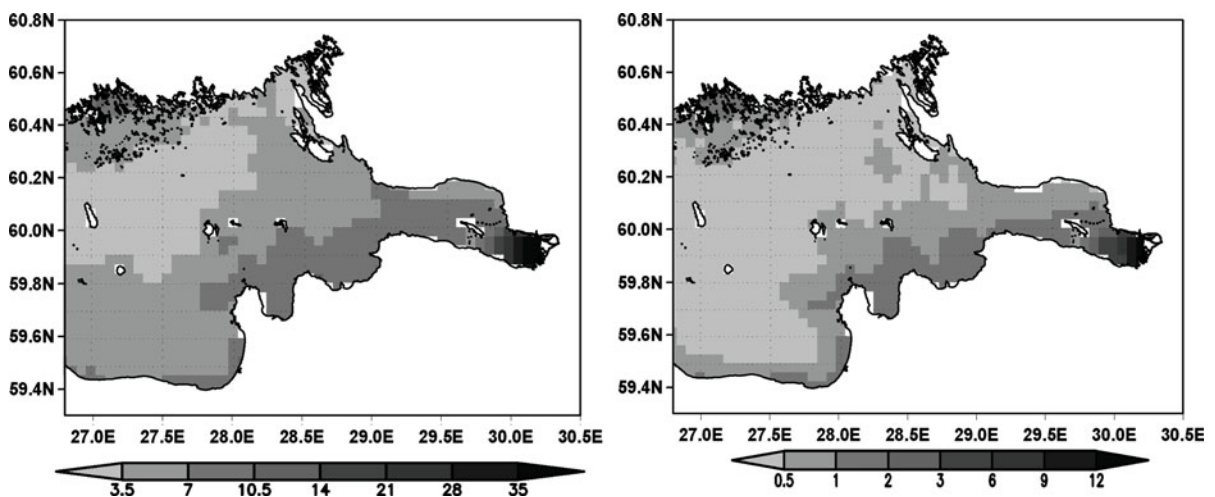


Fig. 6 Simulated sediment retention of nitrogen ($\text{g N m}^{-2} \text{yr}^{-1}$, left) and phosphorus ($\text{g P m}^{-2} \text{yr}^{-1}$, right) in the eastern Gulf of Finland calculated as the difference between the sedimentation of nutrients and their release from the bottom

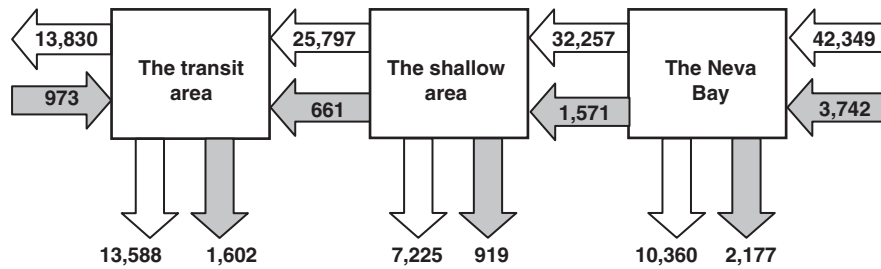


Fig. 7 Simulated annual transport and sediment retention of nitrogen (*clear*) and phosphorus (*shaded*) in the different areas of the Neva Estuary (t yr^{-1}). Areawide nutrient retentions were

obtained by integration over the distributions presented in Fig. 6 within the boundaries shown in Fig. 1

reduced 6354 t of bioavailable nitrogen and 900 t of phosphorus would constitute only 8% of the nitrogen pool and 3% of the phosphorus pool in the waters of the entire Gulf of Finland (see Fig. 3). Accounting for nutrient pools in the sediments that actively participate in biogeochemical cycling (see Table 2), these ratios between the reductions and the stocks decrease down to 1% and 0.6% for nitrogen and phosphorus, respectively. The significance of reductions is more important for the eastern Gulf of Finland, where nutrient stocks are smaller and the total ratios increase up to 2.5% for nitrogen and 1.6% for phosphorus. However, in the open system local changes of concentrations create spatial gradients and induce compensatory transports that emerged in simulations and practically counteracted the reductions (Table 4). Accordingly, the response is expectedly more pronounced only in the shallow Neva Bay with predominating westward transports, where all the annual load reductions were applied and

amounting up to 57% and 42% of nitrogen and phosphorus average stocks, respectively.

Quantitatively, these estimates are less reliable owing to certain shortcomings in the settings of the numerical experiments reported here. Because of the invariable boundary conditions at the entrance to the Gulf of Finland and yearly restoration of the sediment pools to their initial state, the model predicts only immediate, short-term effects of the load reductions. The long-term effects would be more substantial due to the overall reduction in the nutrient reserves both in the water column and in the sediments. For example, the basin-scale SANBALTS model (Savchuk & Wulff, 2007) predicted that under similar nutrient reductions (see Pitkänen et al., 2007) the inorganic nitrogen and phosphorus pools in the Baltic Proper would eventually change by +0.5% and -2%, respectively, while both the sediment pools and the annual primary production in the Gulf of Finland would decrease by 11%. The phosphorus

Table 4 Net transport (export minus import, t yr^{-1}) across the boundaries of different areas in the eastern Gulf of Finland in the reference and scenario simulations

| Scenarios | Boundary between: | | | |
|--------------------------|---|-----------------------|--------------------------|---------------------------|
| | Deep area and the rest of Gulf of Finland | Transit and deep area | Shallow and transit area | Neva Bay and shallow area |
| Net nitrogen transport | | | | |
| Reference | 5,912 | 13,830 | 25,797 | 32,257 |
| P | 6,109 | 15,064 | 27,173 | 33,173 |
| N | 4,485 | 9,828 | 20,544 | 26,980 |
| P&N | 4,523 | 10,730 | 21,864 | 27,865 |
| Net phosphorus transport | | | | |
| Reference | -2,847 | -973 | 661 | 1,571 |
| P | -3,173 | -1,264 | 380 | 1,213 |
| N | -2,577 | -835 | 661 | 1,576 |
| P&N | -2,901 | -1,155 | 374 | 1,212 |

concentration in the deep layers of the Baltic Proper in the late 1990s to early 2000s was clearly elevated (Conley et al., 2002; Savchuk & Wulff, this issue) and its possible decrease due to load reductions and climatic fluctuations could result in a lower import to the Gulf of Finland. Also, improved parameterization of sediment dynamics, including redistribution of resuspended particles, may result in either higher or lower overall nutrient retention, thus increasing or decreasing nutrient imports. Some quantitative changes in predictions may also occur from a more realistic spatial distribution and estimates of external nutrient sources that, in the experiments presented here, were all concentrated in the eastern Gulf of Finland.

Although such speculations can be continued further, experience with the sometimes counter-intuitive behavior of models prompts an evaluation of these and similar hypotheses in numerical experiments with the holistic models, rather than through verbal speculation about realistic but isolated cause-effect relationships.

Conclusion

In order to eliminate the effects of natural inter-annual variability in driving forces, a coupled three-dimensional hydrodynamic–biogeochemical model SPBEM has been run for the Gulf of Finland under recurrent seasonal forcing, including the use of invariable boundary conditions at the entrance to the Gulf and yearly restoration of sediment variables. In simulations with these settings, quasi-steady-state seasonal dynamics were reached in 3 years. The comparability of model results to observed concentrations and estimated biogeochemical fluxes was found to be reasonable enough to allow the use of the simulated values typical for the end of the last century as a reference in an analysis of nutrient reduction scenarios.

These scenarios resulted in rather small (<6%) reductions in the major biogeochemical fluxes determining eutrophication of the eastern Gulf of Finland, which would be difficult to detect in standard monitoring measurements. More significant changes were predicted for nitrogen fixation that, in offshore waters of the eastern Gulf of Finland, decreased by 12% in response to the reduction of the phosphorus load and

increased by 24% in response to the reduction of the nitrogen load. Only in the shallow Neva Bay, first subjected to scenario impacts, did the improvement in the trophic conditions amounted up to 20%.

This weak short-term ecosystem reaction to substantial load reductions is explained by a high nutrient demand (=retention capacity) of the eastern Gulf of Finland that in response to reduced input from the east induces compensatory nutrient import from the west. In other words, the reduction of nutrient loads from the St. Petersburg region would not quickly improve the local conditions outside the Neva Bay but would be useful in an eventual reduction of the nutrient pools in the waters and sediments of the Gulf of Finland and the Baltic Sea. Only lower concentrations westward of the eastern Gulf of Finland would in the long term decrease compensatory transports into the easternmost area.

Acknowledgements The authors thank Svetlana L. Basova and our colleagues from the Baltic Floating University for providing their data used in this study for model validation. We are grateful to Fredrik Wulff and two anonymous reviewers for their critical and constructive comments on this study.

References

- Alenius, P., K. Myrberg & A. Nekrasov, 1998. The physical oceanography of the Gulf of Finland: a review. *Boreal Environment Research* 3: 97–125.
- Andrejev, O., K. Myrberg & P. A. Lundberg, 2004. Age and renewal time of water masses in a semi-enclosed basin – application to the Gulf of Finland. *Tellus* 56A: 548–558.
- Bartnicki, J., A. Gusiev, K. Barret & D. A. Simpson, 2005. Atmospheric supply of nitrogen, lead, cadmium, mercury and lindane to the Baltic Sea over the period 1996–2000. *Baltic Sea Environment Proceedings* 2005: 101.
- Bashmachnikov, I. L., O. M. Johannessen, L. H. Pettersson, G. Evensen, I. A. Neelov, O. P. Savchuk, A. V. Leonov, S. Kaitala, T. Stipa, H. Kuosa & N. N. Filatov, 2005. Numerical simulations of the White Sea hydrodynamics and marine ecosystem. In Filatov, N. N., D. V. Pozdnyakov, et al. (eds), *White Sea: Its Marine Environment and Ecosystem Dynamics Influenced by Global Change*. Springer Praxis Books, Berlin: 337–438.
- BED – The Baltic Environmental Database. <http://nest.se/bed>.
- Conley, D. J., 2000. Biogeochemical nutrient cycles and nutrient management strategies. *Hydrobiologia* 410: 87–96.
- Conley, D. J., C. Humborg, L. Rahm, O. P. Savchuk & F. Wulff, 2002. Hypoxia in the Baltic Sea and basin-scale changes in phosphorus biogeochemistry. *Environmental Science & Technology* 36: 5315–5320.

- Davidan, I. N. & O. P. Savchuk (eds), 1997. "Baltica" Project. Problems of Research and Modelling of the Baltic Sea Ecosystem. Issue 5. Ecosystem Models. Assessment of the Modern State of the Gulf of Finland Ecosystem. Hydro-meteoizdat, St. Petersburg (in Russian).
- Golubkov, M. S., S. M. Golubkov & L. P. Umnova, 2005. Primary production and chlorophyll 'a' concentration as indexes of eutrophication of the Neva Estuary (Eastern Gulf of Finland). Abstracts of 5th Baltic Sea Science Congress – The Baltic Sea Changing Ecosystem, June 2005, Poland: 121–122.
- Gran, V. & H. Pitkänen, 1999. Denitrification in estuarine sediments in the eastern Gulf of Finland, Baltic Sea. *Hydrobiologia* 393: 107–115.
- HELCOM, 2004. The fourth Baltic Sea pollution load compilation (PLC-4). *Baltic Sea Environment Proceedings* 93.
- Howarth, R. W. & R. Marino, 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine systems: evolving views over three decades. *Limnology and Oceanography* 51: 364–376.
- Kähler, P., 1991. Eutrophication and sediment denitrification in coastal marine waters, the example of Kiel Bight. *Kieler Meeresforschungen. Sonderhefte* 8: 112–116.
- Kahru, M., J.-M. Leppänen, O. Rud & O. P. Savchuk, 2000. Cyanobacteria blooms in the Gulf of Finland triggered by saltwater inflow into the Baltic Sea. *Marine Ecology Progress Series* 207: 13–18.
- Kiirikki, M. & H. Pitkänen, 2007. Testing the calculation of nutrient fluxes between the Baltic Proper and the Gulf of Finland with the 3D ecosystem model. In Pitkänen, H. & P. Tallberg (eds), *Searching Efficient Protection Strategies for the Eutrophied Gulf of Finland: The Integrated Use of Experimental and Modelling Tools (SEGUE)*. Finnish Environment, Finland 15: 61–63.
- Kiirikki, M., P. Rantanen, R. Varjopuro, A. Leppänen, M. Hiltunen, H. Pitkänen, P. Ekholm, E. Moukhametshina, A. Inkala, H. Kuosa & J. Sarkkula, 2003. Cost Effective Water Protection in the Gulf of Finland – Focus on St. Petersburg. *The Finnish Environment, Finland*: 632.
- Kiirikki, M., J. Lehtoranta, A. Inkala, H. Pitkänen, S. Hietanen, P. Hall, A. Tengberg, J. Koponen & J. Sarkkula, 2006. A simple sediment process description suitable for 3D-ecosystem modeling – development and testing in the Gulf of Finland. *Journal of Marine Systems* 61: 55–66.
- Lehtoranta, J., 1998. Net sedimentation and sediment-water nutrient fluxes in the eastern Gulf of Finland (Baltic Sea). *Vie Milieu* 48: 341–352.
- Lehtoranta, J., A.-S. Heiskanen & H. Pitkänen, 2004. Particulate N and P characterizing the fate of nutrients along the estuarine gradient of the River Neva (Baltic Sea). *Estuarine, Coastal and Shelf Science* 61: 275–287.
- Leppänen, J.-M., H. Pitkänen, O. Savchuk, S. Basova, V. Drabkova, V. Gran, A.-S. Heiskanen, J. Koponen, I. Spaer & N. Silina, 1997. Eutrophication and its effects in the Gulf of Finland. In Sarkkula, J. (ed.), *Proceedings of the Final Seminar of the Gulf of Finland Year 1996*: 31–49.
- Meeuwig, J. J., P. Kaupilla & H. Pitkänen, 2000. Predicting coastal eutrophication in the Baltic: a limnological approach. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 844–855.
- Nedwell, D. B., T. D. Jickells, M. Trimmer & R. Sanders, 1999. Nutrients in estuaries. *Advances in Ecological Research* 29: 43–92.
- Neelov, I. A., T. A. Eremina, A. V. Isaev, V. A. Ryabchenko, O. P. Savchuk & R. E. Vankevich, 2003. A simulation of the Gulf of Finland ecosystem with 3-D model. *Proceedings Estonian Academy of Science, Biology, Ecology* 52: 347–359.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Perttilä, M. & O. Savchuk (eds), 1996. *The Gulf of Finland*. *Baltic Sea Environment Proceedings* 64B: 47–59.
- Perttilä, M., L. Niemistö & K. Mäkelä, 1995. Distribution, development and total amounts of nutrients in the Gulf of Finland. *Estuarine, Coastal and Shelf Science* 41: 345–360.
- Pitkänen, H., 1991. Nutrient dynamics and trophic conditions in the Eastern Gulf of Finland: the regulatory role of the Neva Estuary. *Aqua Fennica* 21: 105–115.
- Pitkänen, H. & T. Tamminen, 1995. Nitrogen and phosphorus as production limiting factors in the estuarine waters of the Eastern Gulf of Finland. *Marine Ecology Progress Series* 129: 283–294.
- Pitkänen, H., T. Tamminen, P. Kangas, T. Huttula, K. Kivi, H. Kuosa, J. Sarkkula, K. Eloheimo, P. Kauppila & B. Skalkalsky, 1993. Late summer trophic conditions in the north-east Gulf of Finland and the Neva River Estuary, Baltic Sea. *Estuarine, Coastal and Shelf Science* 37: 453–474.
- Pitkänen, H., J. Lehtoranta & A. Räike, 2001. Internal nutrient fluxes counteract decreases in external load: the case of estuarial eastern Gulf of Finland, Baltic Sea. *Ambio* 30: 195–201.
- Pitkänen, H., M. Kiirikki, O. P. Savchuk, A. Räike, P. Korpinen & F. Wulff, 2007. Searching efficient protection strategies for the eutrophied Gulf of Finland: the combined use of 1D and 3D modeling in assessing long-term state scenarios with high spatial resolution. *Ambio* 36: 272–279.
- Prastka, K., R. Sanders & T. Jickells, 1998. Has the role of estuaries as sources or sinks of dissolved inorganic phosphorus changed over time? Results of a K_d study. *Marine Pollution Bulletin* 36: 718–728.
- Savchuk, O. P., 2000. Studies of the assimilation capacity and effects of nutrient load reductions in the eastern Gulf of Finland with a biogeochemical model. *Boreal Environment Research* 5: 147–163.
- Savchuk, O. P., 2002. Nutrient biogeochemical cycles in the Gulf of Riga: scaling up field studies with a mathematical model. *Journal of Marine Systems* 32: 253–280.
- Savchuk, O. P., 2005. Resolving the Baltic Sea into 7 sub-basins: N and P budgets for 1991–1999. *Journal of Marine Systems* 56: 1–15.
- Savchuk, O. & F. Wulff, 1996. Biogeochemical transformations of nitrogen and phosphorus in the marine environment – Coupling hydrodynamic and biogeochemical processes in models for the Baltic Proper. *Systems Ecology Contributions*, 2, Stockholm University: 79 pp.
- Savchuk, O. & F. Wulff, 1999. Modelling regional and large-scale responses of Baltic Sea ecosystems to nutrient reductions. *Hydrobiologia* 393: 35–43.

- Savchuk, O. & F. Wulff, 2001. A model of the biogeochemical cycles of nitrogen and phosphorus in the Baltic. In Wulff, F., L. Rahm & P. Larsson (eds), *A Systems Analysis of the Baltic Sea*. Springer-Verlag, Berlin, Heidelberg: 373–415.
- Savchuk, O. P. & F. Wulff, 2007. Modeling the Baltic Sea Eutrophication in a decision support system. *Ambio* 36: 141–148.
- Savchuk, O. P. & F. Wulff, this issue. Long-term modelling of large-scale nutrient cycles in the entire Baltic Sea. *Hydrobiologia*.
- Seitzinger, S. P., R. W. Sanders & R. Styles, 2002. Bioavailability of DON from natural and anthropogenic sources to estuarine plankton. *Limnology and Oceanography* 47: 353–366.
- Sokolov, A. V., O. A. Andrejev, F. Wulff & M. Rodriguez-Medina, 1997. The Data Assimilation System for data analysis in the Baltic Sea. *Systems Ecology Contributions*, 3, Stockholm University: 66 pp
- Stepanauskas, R., N. O. G. Jorgensen, O. R. Eigaard, A. Zvikas, L. J. Tranvik & L. Leonardson, 2002. Summer inputs of riverine nutrients to the Baltic Sea: bioavailability and eutrophication relevance. *Ecological Monographs* 72: 579–597.
- Telesh, I. V., A. F. Alimov, S. M. Golubkov, V. N. Nikulina & V. E. Panov, 1999. Response of aquatic communities to anthropogenic stress: a comparative study of Neva Bay and the eastern Gulf of Finland. *Hydrobiologia* 393: 95–105.
- Tuominen, L., A. Heinänen, J. Kuparinen & L. P. Nielsen, 1998. Spatial and temporal variability of denitrification in the sediments of the northern Baltic Proper. *Marine Ecology Progress Series* 172: 13–24.
- Wasmund, N., M. Voss & K. Lochte, 2001. Evidence of nitrogen fixation by non-heterocystous cyanobacteria in the Baltic Sea and re-calculation of a budget of nitrogen fixation. *Marine Ecology Progress Series* 214: 1–14.
- Witek, Z., C. Humborg, O. Savchuk, A. Grelowski & E. Łysiak-Pastuszak, 2003. Nitrogen and phosphorus budgets of the Gulf of Gdańsk (Baltic Sea). *Estuarine, Coastal and Shelf Science* 57: 239–248.

Modeling eutrophication and oligotrophication of shallow-water marine systems: the importance of sediments under stratified and well-mixed conditions

Karline Soetaert · Jack J. Middelburg

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 239–254.
DOI: 10.1007/s10750-009-9777-x © The Author(s) 2009. This article is published with open access at Springerlink.com

Abstract A one-dimensional model that couples water-column physics with pelagic and benthic biogeochemistry in a 50-m-deep water column is used to demonstrate the importance of the sediment in the functioning of shallow systems, the eutrophication status of the system, and the system's resilience to oligotrophication. Two physical scenarios, a well-mixed and a stratified water column, are considered and both are run along a gradient of increasing initial pelagic-dissolved inorganic nitrogen (DIN) concentration. Where the mixed layer extends to the bottom, more nutrients and less light are available for growth. Under low to moderately eutrophic conditions (pelagic DIN $<30 \text{ mmol m}^{-3}$), this leads to higher productivity in well-mixed waters, while the stratified system is more productive under highly eutrophic conditions. Under stratification, the build-up of nitrate and depletion of oxygen below the mixed layer does not notably change the functioning of the sediment as a sink for reactive

nitrogen. In sediments underlying well-mixed waters, sedimentary denitrification, fueled mainly by in situ nitrification, is slightly more important (8–15% of total benthic mineralization) than under stratified waters (7–20%), where the influx of bottom-water nitrate is the most important nitrate source. As a consequence of this less efficient removal of reactive nitrogen, the winter DIN concentrations are higher in the stratified scenario. The model is used to estimate the long-term benefits of nutrient reduction scenarios and the timeframe under which the new steady-state condition is approached. It is shown that a 50% reduction in external nitrogen inputs ultimately results in a reduction of 60–70% of the original pelagic DIN concentration. However, as the efflux of nitrogen from the sediment compensates part of the losses in the water column, system oligotrophication is a slow process: after 20 years of reduced inputs, the pelagic DIN concentrations still remain 2.7 mmol m^{-3} (mixed) and 3.9 mmol m^{-3} (stratified) above the ultimate DIN concentrations.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

K. Soetaert (✉) · J. J. Middelburg
Netherlands Institute of Ecology (NIOO-KNAW),
Centre for Estuarine and Marine Ecology,
P.O. Box 140, 4400 NT Yerseke, The Netherlands
e-mail: k.soetaert@nioo.knaw.nl

Keywords Benthic–pelagic coupling · Model · Nitrogen cycle · Denitrification · Eutrophication · Oligotrophication

Introduction

In shallow aquatic systems, the sediments are important sites of organic matter degradation and

secondary production (Jørgensen, 1983). On the shelf, the benthic contribution to total respiration is estimated to be 40–50% (Heip et al., 1995; Wollast, 1998; Alongi, 2005), and this fraction generally increases with decreasing water depth. Through benthic mineralization, an important part of the nutrients, for example nitrogen, taken up by pelagic and benthic algae, is recycled. However, sediments not only return nutrients, but a proportion gets stored or is simply removed from the ecosystem. Denitrification is the most important benthic removal process for nitrogen, accounting for about a third to a half of the total nitrogen loss of the ocean (Christensen, 1994; Middelburg et al., 1996). Sediments also temporarily store organic matter and nutrients both in the long term and in the short term (Soetaert et al., 2000). Nitrogen is removed by the burial of refractory organic material and dissolved inorganic nitrogen through sediment accretion, and by adsorption of ammonium to the sediment.

On short time scales, transient (seasonal) storage of nutrients may cause a mismatch between water-column nutrient demand and supply, and thus directly affect pelagic production. On longer time scales, nutrients that have accumulated in the sediments because of historical eutrophication may be released when the conditions in the water column are altered, for example, by reduced nutrient loadings. Thus, sediment–water interactions in shallow coastal waters have an important effect on primary production and ecosystem dynamics (Middelburg & Soetaert, 2005).

In many shallow marine systems worldwide, increases in industrial, agricultural, and domestic activities have led to massive inputs of nutrients over the past centuries (e.g., van Raaphorst & de Jonge, 2004; Soetaert et al., 2006). Consequently, reduction in nutrient loadings has been undertaken to reverse the trend of increasing eutrophication. However, oligotrophication did not always lead to the desired reduction in the intensity of algal blooms (Kemp et al., 2005), and the effects are not necessarily the same for different nutrients (Soetaert et al., 2006).

Predicting the effects of changes in nutrient input on the ecosystem is difficult. The lack of understanding is at least partly due to limited knowledge on the impact of benthic nutrient cycling on the resilience of ecosystems to eutrophication. For lakes, it has been shown that the effects of such internal loading are to

delay the ecosystem recovery, postponing the establishment of a new equilibrium by about 10–20 years (Jeppesen et al., 2005).

In many regions of the world, where high primary production coincides with low degrees of water-column mixing, eutrophication leads to recurrent hypoxia or anoxia (Officer et al., 1984). The anoxic waters possess unique chemical and biological characteristics, such as unusually high concentrations of dissolved inorganic nutrients and reduced substances, and the occurrence of processes is typically associated with anoxic sediments, such as denitrification (Naqvi et al., 1982), Anammox (Kuypers et al., 2005) or nitrous oxide formation (de Bie et al., 2002). The peculiar characteristics of anoxic bottom waters and the loss of metazoan benthic life (Kemp et al., 2005; Rabalais, 2005) have drastic consequences on ecosystem functioning, the oxidative status of the released nutrients, and the balance of nutrient release or retention in the sediment (Kemp et al., 1990; Childs et al., 2002).

Predicting the response of ecosystems to changing nutrient loads requires the development of models with appropriate descriptions of the benthic–pelagic coupling. Such models could also serve as a basis for setting goals for nutrient reduction (e.g., Cerco & Cole, 1993) or for predicting effects of global change. By increasing the water temperature and river runoff, global change is likely to prolong stratification, reduce mixing, increase primary production, and decrease the solubility of oxygen, with important consequences on the extent of anoxic zones and benthic respiration (Justic et al., 2005). How this will affect the susceptibility of the system to eutrophication or the resilience to oligotrophication remains unclear.

This article reports on the use of a one-dimensional coupled pelagic–benthic model (Soetaert et al., 2000, 2001; Middelburg & Soetaert, 2005) to estimate the impact of sediments on the biogeochemical functioning of a shallow-water marine system. Two scenarios were run: one where the water column remains completely mixed and oxic, and the other where the water column becomes stratified during summer and anoxic conditions develop. These scenarios demonstrate how the sediment affects the resilience of the ecosystem to eutrophication and oligotrophication under these two conditions.

Materials and methods

Model description

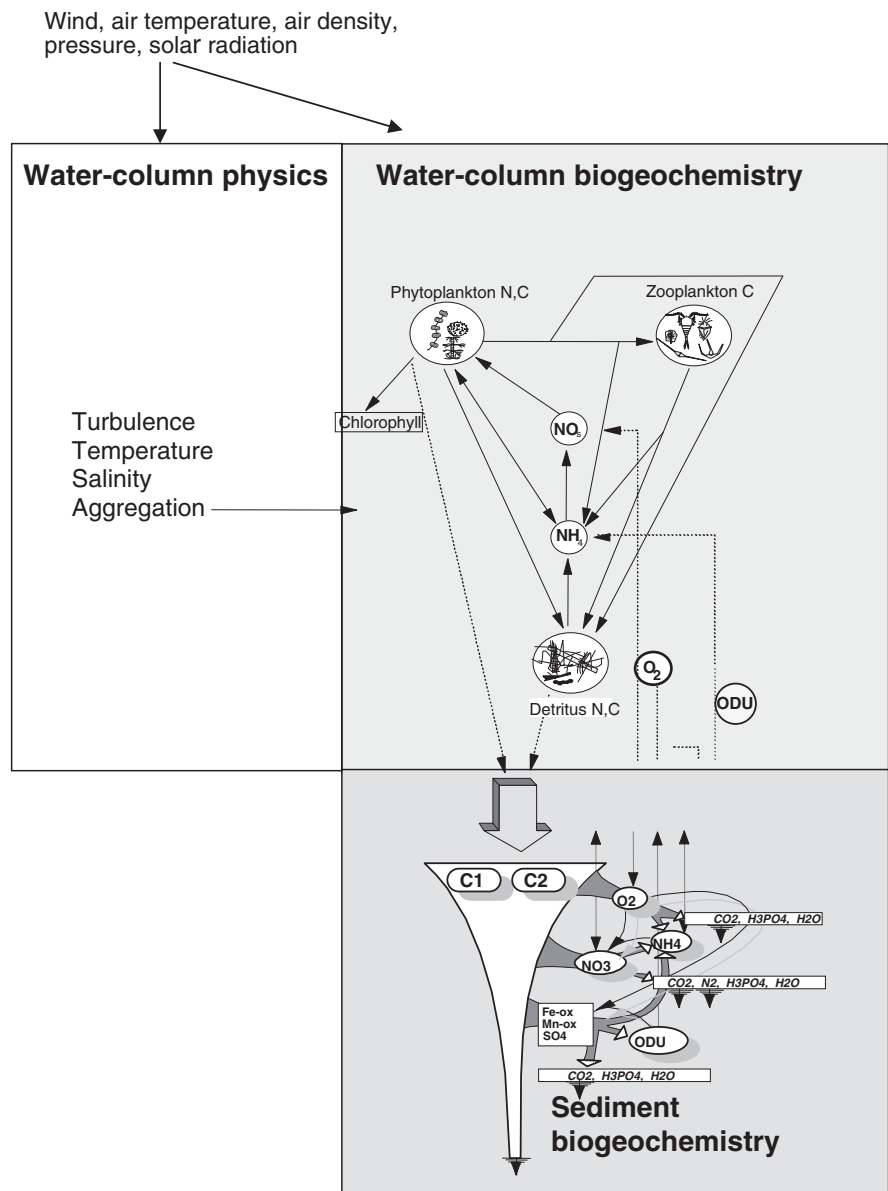
The model couples three submodels that describe the physics of the water column, the pelagic biogeochemistry, and the sediment diagenetic properties (Fig. 1). The model was described extensively by Soetaert et al. (2000, 2001) and Middelburg & Soetaert (2005); these articles should be consulted for model equations and for parameter values. The

pelagic model has 50 equally sized vertical layers in a 50-m-deep water column, with 100 layers in the sediment that progressively increase in thickness from 0.1 cm at the sediment surface to 2 cm at depth. The total sediment depth covered is about 60 cm.

Water-column turbulence and sinking

Vertical water-column mixing in the model is based on the $k-\epsilon$ turbulence-closure formulation of Gaspar et al. (1990). The air-sea heat exchange is forced by 3-h

Fig. 1 Schematic representation of the three coupled submodels describing the water-column physics (*left*), water-column biogeochemistry (*right*), and sediment biogeochemistry (*below*)



atmospheric data, as used by Soetaert et al. (2001) and Middelburg & Soetaert (2005), and representative for a temperate area (Goban spur, NE Atlantic). In order to increase the seasonality in the modeled water temperature, the seasonality in the air temperature was increased by 50%. Solar radiation was modeled as a sine wave, on which a random perturbation was added (Soetaert et al., 2000); 50% of the light was assumed to be in the infrared and the remaining in the photosynthetically active radiation (PAR).

Two physical scenarios were run:

1. The mixed scenario: Here, atmospheric conditions and the tidal pressure gradient are the only external forcing applied to the model.
2. The stratified scenario: In addition to the atmospheric and tidal forcing, input of brackish water (salinity $S^b = 25$) on top and saline water (salinity $S^s = 35.5$) near the bottom provides a stabilizing factor that reduces mixing. The extra fluxes mimic the outflow of estuarine water on the top of marine waters. It is achieved by nudging, using a velocity γ of 0.24 m d^{-1} as follows:

$$-K_z \left. \frac{\partial S}{\partial z} \right|_0 = @ - (S_0 - S^b) \cdot \gamma \quad \text{and} \quad -K_z \left. \frac{\partial S}{\partial z} \right|_{50} = -(S_{50} - S^s) \cdot \gamma,$$

where S is the salinity, z is the depth, 0 and 50 are the air–sea and sediment–water interfaces, respectively, K_z is the vertical mixing coefficient, and @ is the atmospheric salinity “flux” due to evaporation.

Pelagic biogeochemical model

As algae can store excess dissolved inorganic nitrogen, their nitrogen and carbon allocation are uncoupled (Tett & Droop, 1988), which results in an explicit description of phytoplankton carbon and nitrogen and variable nitrogen to carbon ratios. Photoadaptation is approximated by assuming that the algal chlorophyll:nitrogen (Chl:N) ratio linearly and positively depends on the algal nitrogen:carbon (N:C) ratio. One class of zooplankton, expressed in nitrogen units, is modeled. It grazes on phytoplankton, using simple Monod-type saturation. Zooplankton mortality is expressed as a second-order closure term, which increases strongly when the oxygen concentration is low. The constant stoichiometry of

the zooplankton is maintained by excess excretion or respiration to account for imbalances between nitrogen or carbon in its food and in zooplankton tissues. The bacterial loop is not resolved, but it is assumed that the mineralization of organic nitrogen and organic carbon proceeds at a first-order rate, where nitrogen decays more rapidly than carbon. Detritus mineralization proceeds unlimited by the oxygen concentration. In oxic conditions, oxygen is consumed; however, when oxygen is depleted, the mineralization process produces an undefined anoxic substance (oxygen demand units, ODU). This description is similar to that for the sediments (see below). Upon restoration of the oxic conditions, the ODU are re-oxidized, consuming one mole of oxygen per mole of ODU. Nitrate and ammonium are the only nutrients considered; ammonium is the preferred source of nitrogen for the algae. It is converted to nitrate (water column nitrification) at a low rate.

The sinking rate of detritus is calculated by the aggregation model of Kriest & Evans (1999). In contrast, algae sink with a constant velocity. All pelagic rates are temperature dependent. Photosynthetically active radiation is described by short- and long-wave radiation, with a different background extinction coefficient, to which a chlorophyll-dependent extinction is added. With the light settings used, the yearly averaged depth of the 1% level of daylight penetration is around 25 m, which is consistent with measurements in the North Sea (Visser, 1969 in Otto et al., 1990).

Oxygen is the only biogeochemical substance exchanged across the air–sea interface.

Sediment biogeochemical model

Diagenetic models (Boudreau, 1996) are biogeochemical descriptions of sedimentary systems that take into account the complexity of the sediments. The diagenetic model used, OMEXDIA (Soetaert et al., 1996), represents the sedimentary C, N, and oxygen (O_2) cycles. It combines the description of solid (two organic matter pools) and solute (oxygen, nitrate, ammonium, and ODU) species. The organic matter degradation in the sediment follows the standard two-compartment type of organic matter consumption, with orders-of-magnitude differences in decay rates and different N:C ratios. Mineralization releases ammonium which may subsequently be

re-oxidized to nitrate (nitrification), flux to bottom water, or be adsorbed onto the solid phase. From the sediment–water interface to depth into the sediment, the oxidation of organic matter proceeds first using oxygen as an electron acceptor. In deeper, low-oxygen layers, nitrate is used for mineralization (denitrification) until nitrate is exhausted, below which organic matter oxidation is driven by the reduction of manganese (Mn) and iron (Fe) oxides, sulfate (SO_4), and finally by methane fermentation. None of these anoxic processes are separately modeled; rather, it is assumed that the mineralization of organic matter via these pathways is not limited by oxidants, such that the electron acceptors that take part in anoxic mineralization need not be described. Given the high concentrations of SO_4^{2-} in marine pore waters, this is a valid assumption. However, during anoxic mineralization, Fe and Mn ions, sulfide, and methane are formed and released to the interstitial waters where they may flux upward into the oxic zone to be re-oxidized. As this oxygen consumption may constitute a significant fraction of the total oxygen consumption in the sediment, it needs to be taken into account. Thus, in OMEXDIA, the anoxic mineralization produces a reduced substance, termed ODU, which constitutes the sum of Fe and Mn ions, sulfide, and methane. When oxygen is present, the ODU is re-oxidized; one mole of oxygen is consumed for the re-oxidation of one mole of ODU. The interplay between particles and dissolved substances is incorporated by imposing a gradient in porosity (Φ), declining exponentially from 0.95 at the sediment–water interface to 0.75 at 3 cm depth, below which it remains constant; such a porosity gradient is typical for muddy sediments.

In analogy with water-column processes, sedimentary constituents are affected by mixing and advection processes. A constant sediment accretion rate is assumed. The solid phase in muddy sediments is mixed mainly by the crawling and feeding activity of animals, which is parameterized by a bioturbation coefficient (Db). As animal life is restricted to the upper layer of the sediments, the bioturbation coefficient is assumed constant in the upper, bioturbated layer, below which it declines exponentially. Solute mixing in mud occurs essentially via molecular diffusion, corrected for the tortuous path that molecules need to take, and may be increased by the ventilation activity of benthic animals. In

OMEXDIA, animal ventilation is represented by the simplest formulation, that is, as an enhanced diffusion. Thus, diffusion is increased by a multiplication factor, which varies with depth into the sediment in a similar manner as the bioturbation coefficient.

Sediment parameters

The biogeochemical rate parameters and half-saturation or inhibition constants from the original OMEXDIA model (Soetaert et al., 1996) were left unaltered. As the diagenetic model explicitly deals with biogeochemical processes, the effect of benthic animals must be parameterized. In general, benthic animals influence biogeochemical cycles in three ways: (1) through their metabolism, respiration, and excretion, they convert organic matter to inorganic form (dissolved nutrients and carbon dioxide) and consume oxygen; (2) by their crawling and feeding activity, they mix the sediment (bioturbation); and (3) some organisms pump bottom water through their burrows thus increasing the exchange rate of solutes between the water column and the sediment (bio-irrigation).

The respiration of higher organisms and bacteria was incorporated as a first-order decay rate, assuming two pools with largely varying decomposition rates, and using the parameter values as in Soetaert et al. (1996). These settings were the same between the well-mixed and the stratified scenarios. Solid-phase mixing due to animal crawling and feeding activity was modeled as a diffusive process, while the ventilation of the sediment due to burrow irrigation was described as an enhancement of sediment–water exchange over molecular diffusion. The intensity of these processes was considered to differ under well-mixed and stratified conditions. When lasting for long enough periods, anoxia may completely eradicate the larger benthic animals (e.g., Rabalais et al., 2002), and it may take years before significant populations of macrofauna return (Josefson & Widbom, 1988). In contrast, smaller organisms are less affected and may recover more rapidly upon restoration of oxic conditions (Josefson & Widbom, 1988). As larger animals contribute most to bioturbation and bio-irrigation, anoxia also affects the degree to which the sediments are mixed or to which water is pumped in the sediment. Mixing intensities in marine sediments were compiled by Middelburg et al. (1997) and they

fall roughly in the range from 1 to 100 $\text{cm}^2 \text{yr}^{-1}$ in shallow sediments; for a 50-m-deep station, the geometric and arithmetic mean of bioturbation coefficients are 5 and 25 $\text{cm}^2 \text{yr}^{-1}$, respectively. For the mixed-case scenario, with permanently oxic bottom-water conditions, an intermediate bioturbation coefficient of 15 $\text{cm}^2 \text{yr}^{-1}$ was chosen, while for the stratified case, where the water column becomes temporarily anoxic, the bioturbation rate was set an order of magnitude lower (1.5 $\text{cm}^2 \text{yr}^{-1}$). Mixed-layer depth was assumed to be 5 cm in the well-mixed scenario and 1 cm in the stratified scenario. Similarly, the ventilation, which triples the diffusion rate over molecular diffusion in the well-mixed physical scenario, is turned off under stratification.

Submodel coupling

The coupling between the physical and biogeochemical pelagic models was done off-line. The interaction between benthic and pelagic biogeochemistry is two-directional; therefore, these submodels are coupled on-line. Organic matter, settling from the water column onto the sediment, adds to sediment organic matter, assuming a fixed proportion of rapidly and slowly decaying organic matter. The pelagic model provides the bottom-water concentrations of oxygen, nitrate, ammonium, and ODU, which constitute the upper boundary condition of the sediment model. The latter then estimates the sediment–water exchange flux, which is used to adjust bottom-water concentrations. It is also assumed that the temperatures in sediment and bottom water are equal.

Sedimentary denitrification removes fixed nitrogen from the system. If the model is run in steady-state (with the same annual dynamics repeated), this nitrogen loss should be compensated by lateral influx. Soetaert et al. (2000) tested several scenarios to model this flux and concluded that the manner in which this nitrogen was effectively returned had only a marginal effect on the system's dynamics. For the runs reported here, a flux of ammonium was imposed to the bottom water, instantaneously compensating the N_2 loss.

Solving the coupled model

The coupled model was implemented in the simulation environment FEMME (Soetaert et al., 2002),

which addresses the technical aspects of the modeling.

Owing to the large differences in spatial scales, of the order of 1 m in the pelagic model, and from 1 mm to 2 cm in the sediment model, the numerical properties of pelagic and benthic submodels are very different. The pelagic model was solved by simple explicit (Euler) integration, except for transport which was approximated by a fully implicit integration scheme. The time-step chosen for the pelagic model was 7.5 min. In the benthic model, the dynamics of the different substances proceed at orders-of-magnitude different time scales, from a fraction of a second for oxygen in some sediment layers to the order of years for the low-reactive organic matter components. This leads to a stiff set of equations that was solved with an implicit integration routine that adapts the time-step to the model dynamics, in order to preserve a pre-set tolerance level (Brown et al., 1989).

The coupled model then proceeded with common time-steps of 1 h, during which the pelagic model took eight steps, and the diagenetic model took a variable number of steps (depending on the model dynamics). During the hourly integration, the average deposition of organic matter and the average bottom-water concentrations, as estimated during the previous hour from the pelagic model, were imposed onto the sediment upper boundary. Similarly, the average sediment–water nutrient, oxygen, and ODU fluxes, as estimated by the diagenetic model from the previous hourly integration, were used to update the bottom-water concentrations. This mode of coupling is very efficient, but introduces a time lag of 1 h between both submodels. As the interest of this work was mainly in much longer time scales, this artificial lag was insignificant. Reducing the common time-step did not have any noticeable effect on the model outcome.

Owing to the large difference in time scales of the benthic model, this part must be carefully initialized, to avoid a too long spin-up time before the sediment model is in equilibrium with the pelagic dynamics (and vice versa). Correct initialization of the slowest reactive fraction of organic matter is especially critical. This is the fraction of organic matter that partly escapes below the bioturbated area, where it is moved only by sediment advection. It would take many thousands of years of runtime before

concentrations of this organic matter fraction have built up in equilibrium at these depths. In order to avoid such long spin-up, the following procedure was used. First, the pelagic model was run without adding the diagenetic model. Instead, a zero-flux lower boundary was assumed (Soetaert et al., 2000), meaning that all organic matter deposited is immediately mineralized and provokes an equivalent source of nitrogen and sink of oxygen. Based on this run, an annual mean organic matter deposition rate and a mean bottom-water concentration were calculated. These were then used to initialize the diagenetic model by generating steady-state concentration versus depth profiles under these conditions. This is done by solving for the root of the 600 non-linear equations. After this initialization, the model was run for at least 5 years, and convergence to equilibrium tested. Only the results from the last year are shown.

Both physical scenarios (mixed and stratified) were applied in a range of initial concentrations of DIN in the water column (5, 10, 15, 20, 25, 30, 50, 100, and 200 mmol m^{-3}). In order to run the full model for 1 year of simulation, it takes about 2.5 min on a Pentium 4, 3 GHz personal computer.

Results

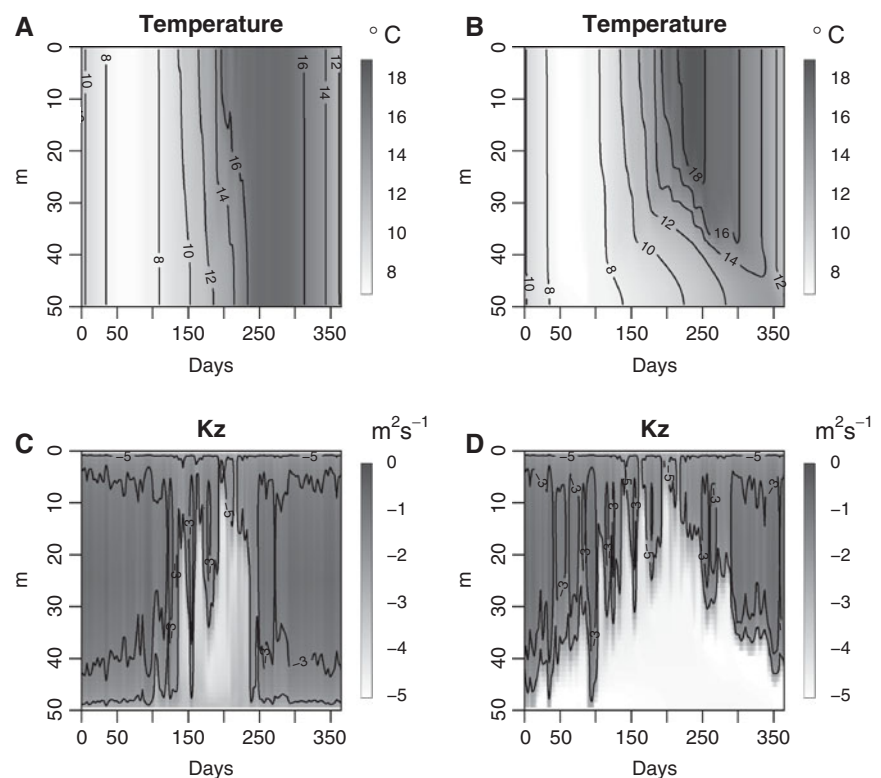
Physical properties

Figure 2 displays an annual cycle of the modeled temperature field and the turbulent diffusion coefficient as generated by the turbulence-closure model for the two physical conditions. Without extra stabilization (the mixed-case scenario), the 50-m-deep water column remains almost completely mixed, except the onset of summer where the water column stratifies slightly (Fig. 2C) and a temperature gradient of about 1°C from top to bottom emerges (Fig. 2A). In contrast, when including a brackish-water flux near the surface and a flux of saline water near the bottom, stratification starts in spring and lasts throughout summer (Fig. 2D). The maximum difference between bottom and surface temperatures is 9°C (Fig. 2B).

Water-column biogeochemistry

In Fig. 3, the changes in nitrate and oxygen concentrations during an annual cycle and vertically in the

Fig. 2 Spatiotemporal plot of water-column temperature (A, B) and diffusion coefficients (C, D) for the mixed (A, C) and stratified (B, D) conditions



water column are depicted for a water-column DIN concentration in winter of 10 mmol m^{-3} . This is a typical condition for temperate coastal systems such as the North Sea.

The development of the phytoplankton bloom, as light intensity increases in spring, depletes the mixed-layer nitrate (Fig. 3C, D) until it is entirely consumed in summer. This depletion occurs earlier (around day 100; Fig. 3D) in the stratified compared to the well-mixed case (day 130, Fig. 3C). The organic matter produced sinks toward the bottom and is mineralized at greater depth. During this process, oxygen is consumed and nutrients are released. Under stratified conditions, this first leads to a strong decrease and ultimately complete depletion of oxygen and the water column becomes anoxic below the mixed-layer depth (Fig. 3B). Also, a sub-surface nitrate peak emerges after the exhaustion of nutrients in the mixed layer (Fig. 3D). Similar results were obtained in a modeling study of the Louisiana-Texas shelf in the Gulf of Mexico (Chen et al., 1997).

If the winter nutrient concentration of the water is increased from 10 mmol m^{-3} (Fig. 3C, D) to 30 mmol m^{-3} (Fig. 4A, B), the initiation of nutrient limitation is postponed until a later date. Thus, nitrate

is completely exhausted at day 190 and 170 in the mixed and stratified waters, respectively (Fig. 4), which is 60 to 70 days later than depletion in the 10 mmol m^{-3} scenario. This trend continues as the winter DIN concentrations are increased (not shown), except the highest loadings at which exhaustion no longer occurs.

Eutrophication gradient

The annual algal uptake of DIN and primary production along a gradient of increasing pelagic DIN concentrations are shown in Fig. 5A, B. Higher nutrient concentrations enhance annual primary production by prolonging the production period, and the annual-averaged algal assimilation of DIN and primary production increase until they level off. Under the mixed-case scenario, this occurs at a winter DIN concentration of about 50 mmol m^{-3} , where primary production reaches values of $390 \text{ g C m}^{-2} \text{ yr}^{-1}$, while under stratification, the maximum production attained is higher ($510 \text{ g C m}^{-2} \text{ yr}^{-1}$) and this maximum is reached at higher mean DIN concentrations (100 mmol m^{-3}). Stratification has an opposite effect on nutrient and light limitation of

Fig. 3 Spatiotemporal plot of water-column oxygen (A, B) and nitrate (C, D) concentrations for the mixed (A, C) and stratified (B, D) conditions, as modeled for a winter water-column DIN concentration of 10 mmol m^{-3}

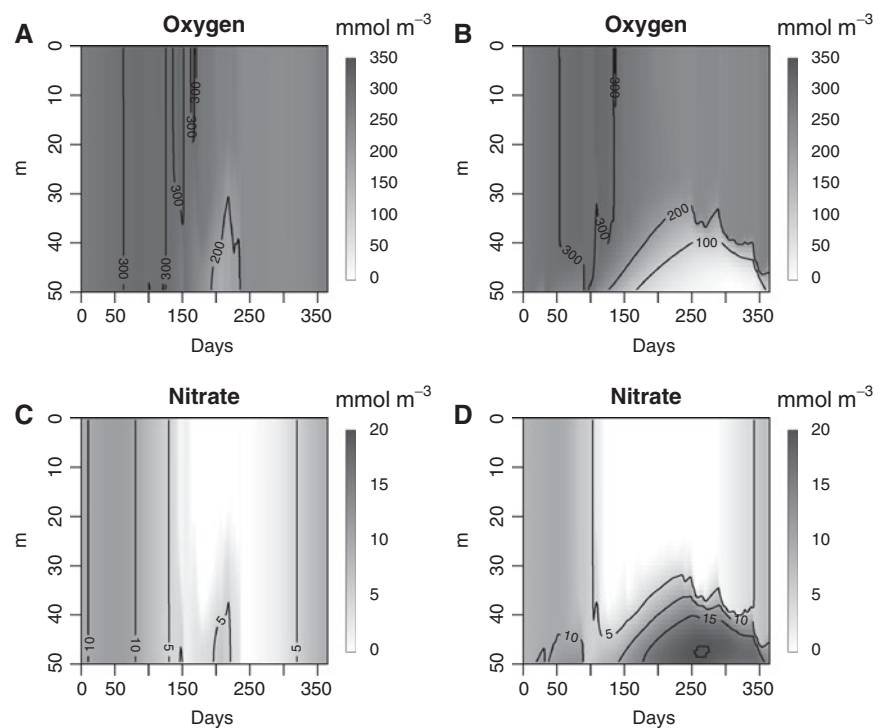


Fig. 4 Water-column nitrate concentrations for the mixed (A) and stratified (B) conditions, as modeled for a winter water-column DIN concentration of 30 mmol m^{-3}

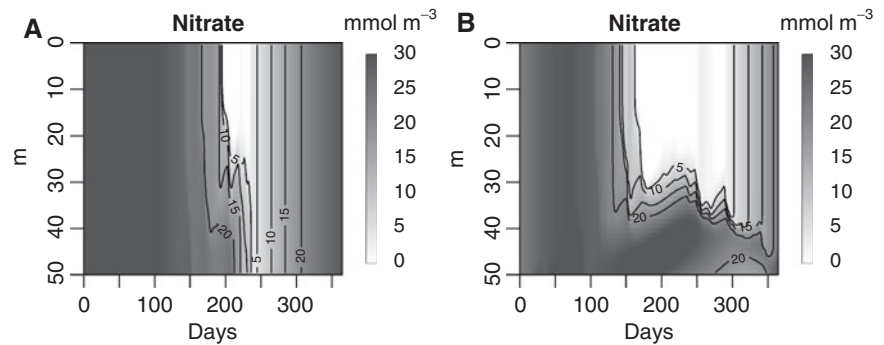
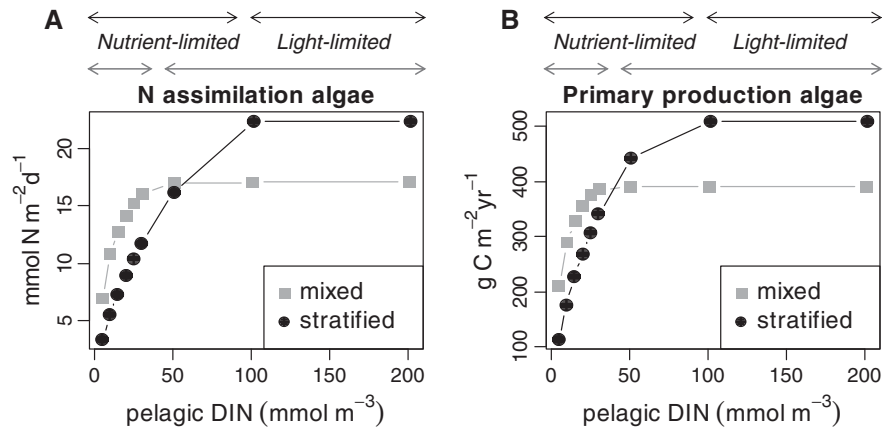


Fig. 5 Algal uptake of dissolved inorganic nitrogen (A) and primary production (B) as a function of increased mean pelagic DIN concentrations for the mixed and stratified scenarios. The top of the diagram indicates where the algae are on average “nutrient limited” or “light limited” for the well-mixed (gray line) and stratified (black line) conditions



the algae. If algae are mixed from surface to bottom, the average light experienced is less intense and the light limitation is stronger than where mixing is limited from the surface to several meters above the bottom. However, under low mixing, nutrient concentrations build up below the thermocline and remain inaccessible for algal growth.

Therefore, where winter nutrient concentrations are low, algal production in a stratified water column is lower than under well-mixed conditions. At very high winter concentrations, nutrient limitation no longer occurs, and the productivity under stratification surpasses the mixed-water case production, due to the more favorable light conditions.

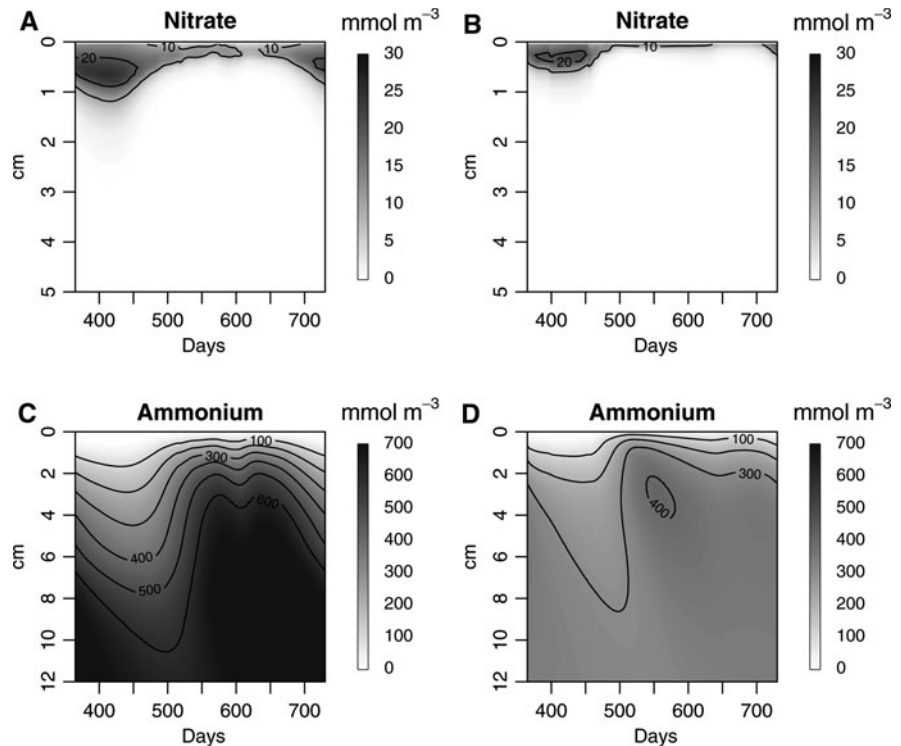
In the model presented here, the primary production of pelagic DIN concentrations ranging from 5 to 15 mmol m^{-3} varied from 210 to $325 \text{ g C m}^{-2} \text{ yr}^{-1}$ under well-mixed conditions, and from 112 to $220 \text{ g C m}^{-2} \text{ yr}^{-1}$ under stratified conditions (Fig. 5B). This compares well with the estimate of 250 – $300 \text{ g C m}^{-2} \text{ yr}^{-1}$ for coastal waters of the North Sea (Joint & Pomroy, 1993).

Sediment model

The changes in sedimentary nitrate and ammonium concentrations as determined for pelagic winter DIN concentrations of 10 mmol m^{-3} are illustrated in Fig. 6. In winter, a clear build-up of nitrate can be seen below the sediment–water interface, up to 30 mmol m^{-3} for the mixed case (Fig. 6A) and 22 mmol m^{-3} for the stratified case (Fig. 6B). As more oxygen is consumed during summer, the oxygen penetration depth shifts upward, and the nitrate peak becomes less pronounced (mixed case, Fig. 6A) or disappears (stratified case, Fig. 6B). At the same time, ammonium increases at depth in the sediment (Fig. 6C, D).

The sediments underlying the well-mixed water column have higher bioturbation and bio-irrigation rates. As these have a different effect on sediment functioning, both the ammonium and nitrate concentrations are highest under well-mixed conditions. By bio-irrigating their burrows, organisms pump water into the sediment, thus shortening diffusive pathways

Fig. 6 Spatiotemporal plot of sediment nitrate (A, B) and ammonium (C, D) concentrations for the well-mixed (A, C) and stratified (B, D) conditions, as modeled for pelagic winter DIN concentrations of 10 mmol m^{-3}



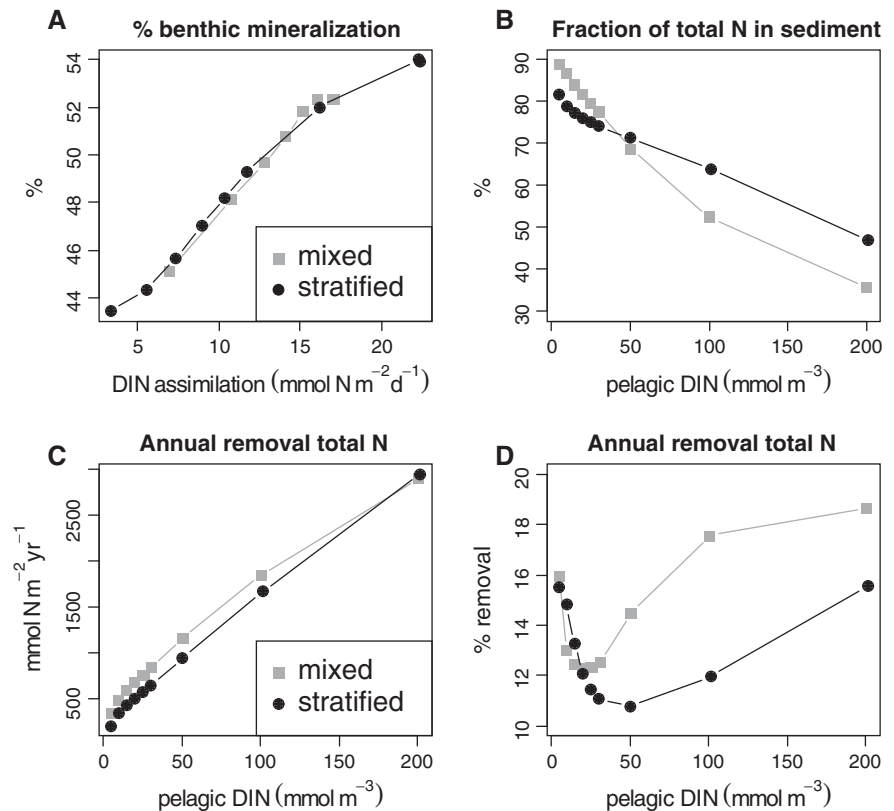
for oxygen in sediments (Meysman et al., 2006); this promotes oxygen-consuming processes such as nitrification. This provides an explanation for the higher concentrations of nitrate in the mixed-case scenario. The build-up of ammonium depends on the amount and the depth of organic matter mineralization. With increasing mixing rates, organic matter deeper into the sediment is decomposed and the pore waters become enriched in ammonium at these depths. Similar behavior has been described for the build-up of phosphate as a function of sediment advection (Van Cappellen & Berner, 1988). In contrast, as bioturbation in sediments under the stratified water column is reduced, organic matter is mineralized closer to the sediment–water interface. Thus, ammonium is released more superficially, resulting in a stronger initial gradient and therefore a larger efflux to the water column; as a consequence, lower concentrations build up.

The importance of the sediment in the biogeochemical functioning of shallow-water marine systems is illustrated by Fig. 7. A significant fraction of the organic matter decays in the sediment. The relative contribution of the sediment with respect to total system mineralization varies between 44% and 54%

(Fig. 7A). This corresponds well with literature estimates, which on average are close to a 50% contribution of benthic mineralization in shallow systems (Heip et al., 1995; Wollast, 1998; Rabouille et al., 2001; Middelburg et al., 2005). With increasing eutrophication, the share of the sediment in system mineralization increases slightly (Fig. 7A), from 44% to 54%. This is due to a lower efficiency of zooplankton grazing on the algae at high production rates, as the feeding rate of the zooplankton saturates at high algal concentrations (not shown). Under these conditions, an increasing fraction of the primary production remains ungrazed and sinks to the sediment.

The total nitrogen inventory in the sediment was estimated by integrating organic nitrogen, ammonium, and nitrate up to 60 cm depth (the total sediment depth modeled). This estimate is accurate for the reactive organic matter, which does not penetrate deeper than 15 cm, and for nitrate, which is completely consumed within the upper centimeter of the sediment. In contrast, the ammonium concentration is highest at great sediment depth, and the deeper the integration, the larger the share of total ammonium in the sedimentary nitrogen inventory will become. However, it is reasonable to assume that

Fig. 7 **A** Benthic mineralization as a percentage of total mineralization. **B** Percentage of total nitrogen residing in the upper 60 cm of the sediment. **C** Total sedimentary denitrification. **D** Annual percentage removal of total (benthic + pelagic) nitrogen due to sedimentary denitrification



deeply buried ammonium (>60 cm) will not influence the dynamics of the overlying waters and can therefore be ignored. Moreover, most of the sedimentary nitrogen resides in the organic nitrogen fraction, only 2.5–2.7% (mixed) or 2.2–2.6% (stratified) of sediment nitrogen consists of ammonium; nitrate contributes a negligible amount. Thus, even if ammonium was to be integrated to one meter depth or deeper, this would not have a large impact on the estimated sediment inventory.

Total nitrogen in the sediment ranges from 2000 to 5000 mmol N m^{-2} in the mixed-case scenario and from 1000 to 9000 mmol N m^{-2} below stratified waters (not shown). When expressed as a percentage of the total (i.e., pelagic + benthic) nitrogen inventory, the importance of the sediment as a reservoir of reactive nitrogen becomes evident (Fig. 7B). At oligotrophic to moderately eutrophic conditions (winter DIN <50 mmol m^{-3}), 70–90% of all nitrogen resides in the sediment, and this lowers consistently with increasing eutrophication. However, even at the most eutrophic conditions, 38–48% of all nitrogen is found in the sediment.

In both types of sediments, anoxic processes prevail; the contribution to total mineralization varied from 63 to 88% under a stratified water column, and from 70 to 82% under a mixed water column (not shown). The contribution of sedimentary denitrification is also similar in both cases (stratified: 8–15%; mixed: 7–20%, not shown), although the source of nitrate differed greatly. In the well-mixed case, denitrification was 100% fueled by in situ nitrification of ammonium produced during mineralization, except at the highest DIN concentrations (winter DIN >100 mmol m^{-3}), when nitrate from the overlying water also fluxed into the sediment (not shown). In contrast, for sediments underlying stratified water, the influx of bottom-water nitrate which builds up below the thermocline (Figs. 3D, 4B) is much more important. In this case, the contribution of denitrification supported by nitrification declined quasi-exponentially from 100% at the most oligotrophic conditions to <20% at pelagic DIN concentrations of 50 mmol m^{-3} (not shown). The relative contributions of the different mineralization pathways in this model compare well with literature data, which show

a more than 50% contribution of anoxic pathways in shallow sediments, and a share between 0 and 20% for the denitrification (e.g., Burdige, 2006).

The importance of the denitrification process for the nitrogen cycling in the two conditions is represented in Fig. 7C, D. The sedimentary losses vary from 200 to 3000 mmol N m⁻² yr⁻¹ (Fig. 7C), which is 10–20% of the total nitrogen per year (Fig. 7D). When expressed as a function of pelagic nitrogen, the removal varies from 25% (most eutrophic) to 140% (oligotrophic) per year (not shown). The denitrification process is more important under the well-mixed condition compared to the stratified condition because primary production, as well as benthic mineralization, is higher here (except in the most eutrophic conditions), while the contribution of denitrification to total sediment mineralization is comparable.

Nutrient reduction scenarios

Nutrient concentrations in an open system result from the balance of input and output processes. In the model described here, the output and input are, respectively, the sedimentary denitrification and the flux that compensates the losses due to denitrification (see Material and Methods, submodel coupling). Although this model approach is very simple, it closely mimics the overall nitrogen balance in shelf ecosystems, where nitrate input from external sources is balanced by denitrification (Middelburg & Soetaert, 2005).

If the nitrogen loading (input) to the water column is altered, then the system dynamics will change until the denitrification again balances the nitrogen addition, and steady-state seasonal dynamics are restored. The magnitude of denitrification depends on the nutrient concentrations in the overlying water and the rate of sedimentary organic matter mineralization, which in turn depends on the water-column nitrogen concentration. Thus, if the nutrient inputs to the system are altered, the steady-state nutrient concentrations will also be altered. Figure 8 is essentially a redrawing of Fig. 7C, but now the annual mean DIN concentration in the water column has been plotted as a function of the net nitrogen input to the system. Note that this net input is expressed per square meter of horizontal surface, thus to convert to whole ecosystem inputs, this value should be multiplied by the mean surface area of the system.

Several interesting aspects can be deduced from Fig. 8. First, as total denitrification is greater in the well-mixed scenario, total pelagic nitrogen concentrations are greater, up to about 10 mmol m⁻³, in stratified compared to well-mixed conditions, for the same nitrogen loading. Second, the effect of a certain reduction in the nitrogen load on the pelagic nitrogen concentration can be estimated. For instance, a loading of 3000 mmol N m⁻² yr⁻¹ results in a mean pelagic DIN concentration of ~200 mmol m⁻³, while at half the loading, the DIN concentration becomes 87 and 72 mmol m⁻³ under stratified and mixed conditions, respectively, a reduction of 57 and 64% of the original concentration. Halving the DIN loading of 1000 mmol N m⁻² yr⁻¹ will ultimately reduce the total DIN concentration from 52 to 22 mmol m⁻³ (a reduction of 58%) for stratified conditions and from 38 to 12 mmol m⁻³ (68%) for well-mixed conditions.

Third, Fig. 9 illustrates the results from a nutrient reduction scenario for the mixed case (Fig. 9A) and the stratified (Fig. 9B) condition. Starting from a water column with an average DIN concentration of 30 mmol m⁻³, the nutrient inputs (estimated as the annually averaged denitrification under equilibrium) were halved, and the model is run for 20 years under these altered inputs. There is a strong decline in the nutrient concentration at the onset of the nutrient loading reduction, but this is subsequently attenuated.

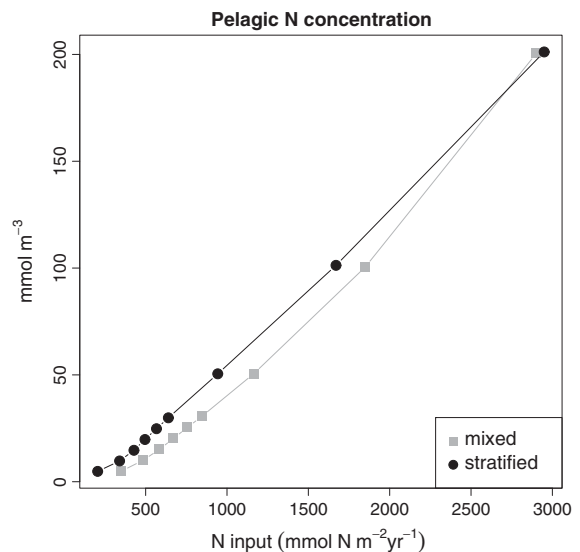
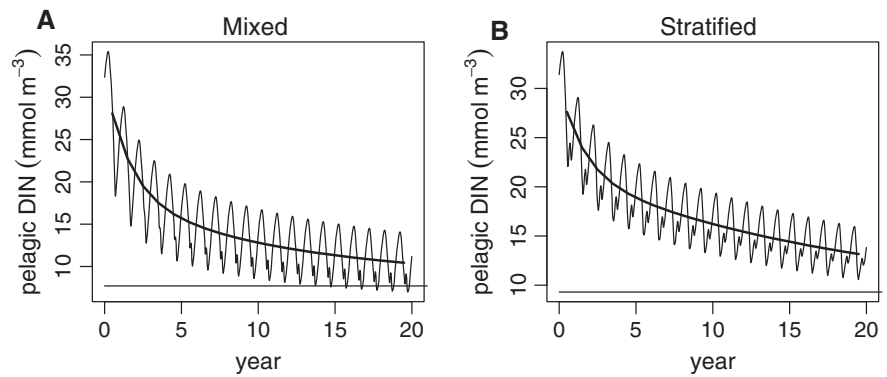


Fig. 8 Pelagic nitrogen concentration as a function of total nitrogen input

Fig. 9 Pelagic DIN concentration versus time (20 years) for the 50% reduction scenario in the mixed (A) and stratified (B) case. The *thick line* shows the annual-averaged concentrations versus time. The *thin horizontal line* denotes the ultimate equilibrium concentration



Even after the system has been subjected to the reduced nutrient inputs for 20 years, the new equilibrium DIN concentration is not yet attained, but is still 2.7 mmol m^{-3} (mixed) and 3.9 mmol m^{-3} (stratified) above those values. At that time, only 34% (stratified) and 37% (mixed) of the total system nitrogen loss has occurred in the water column. Most of the nitrogen has been lost from the sediment organic nitrogen (60 and 56% for stratified and mixed scenarios, respectively). Desorption of ammonium (5 and 6%, respectively) and loss of dissolved ammonium from the sediment (1% each) contribute only marginally to system losses.

Discussion

Estimates of denitrification rates are often lower where bottom waters become anoxic (Kemp et al., 1990; Childs et al., 2002). However, many denitrification measurements are performed in shallower regions where anoxia coincides with low nitrate concentrations. In these environments, denitrification is impeded by nitrate limitation, due to the absence of nitrification (caused by oxygen deficiency) and low concentrations of bottom-water nitrate. Compared to these situations, the simulations reported here are somewhat atypical: as the bottom water is too deep to allow algal growth, high nitrate concentrations below the mixed-layer depth are generated. Note that the modeled increase of nitrate up to about 20 mmol m^{-3} (Fig. 3D) corresponds well with findings in other coastal areas, for example, in the German Bight during the 1980s (Brockmann et al., 1990).

Consistent with the above-mentioned measurements, this model predicts (slightly) lower nitrogen

removal efficiencies under thermal stratification, albeit for different reasons. In both stratified and well-mixed conditions, the sediment functions similarly with respect to benthic removal of reactive nitrogen: compared to total sediment mineralization, the relative importance of denitrification in both scenarios is similar and varies between 8–15% and 7–20% for the stratified and mixed case, respectively. The slightly lower overall removal during stratification (Fig. 7C) arises because, for similar pelagic DIN concentrations, algal production is lower (Fig. 5A); therefore, the supply of organic matter and thus sedimentary mineralization and denitrification are also lower.

That the denitrification is so similar under the very different bottom-water conditions in the oxic and anoxic environments relates to the contrasting sediment communities. The organic-rich sediments underlying oxic bottom waters (as in the well-mixed scenario) are populated by long-lived, relatively large benthic animals. By ventilating their burrows, they facilitate oxygen transfer to the sediment (Aller, 1980; Meysman et al., 2006), which enhances the re-oxidation of ammonium (nitrification) and promotes denitrification of the nitrate formed in situ. In contrast, sediments exposed to low-oxygen or anoxic bottom waters are either devoid of life or populated by smaller, opportunistic species that do not ventilate the sediments. In the absence of oxygen, nitrate that has built up in the bottom water is the only source for sedimentary denitrification, but the rate at which this nitrate can flux into the sediment is limited due to the reduced sediment–water exchange rates. The overall effect of these two types of benthic ecosystems is that denitrification contributes similarly and significantly to coastal nitrogen cycling under both oxic and anoxic conditions.

Marine systems are essentially three-dimensional environments, driven by inflows and outflows of pelagic constituents (via currents and turbulence) and by internal biogeochemical processes. When the external driving variables remain constant over a sufficiently long period of time, or are recurring with a certain (e.g., annual) periodicity, steady-state seasonal dynamics will evolve, whereby inputs exactly balance outputs. For nitrogen, this means that the net input of external nitrogen, resulting from three-dimensional hydrodynamics, will balance the sedimentary denitrification, by far the most important, if not the only, loss term. In order to obtain such a steady-state by dynamically running a model, however, is computationally costly. This is especially true when sedimentary processes are explicitly modeled, due to the very long time scales involved; the residence time of the less labile organic matter in the sediment is of the order of tens to hundreds of years, and it will take several times this amount of time before the steady-state will be achieved. Nevertheless, as this fraction of organic matter plays an important role in a system's resilience, it cannot simply be ignored (Soetaert et al., 2000).

Whereas it is now possible to couple relatively complex sediment models that accurately describe the benthic biogeochemical complexity to water column models, this is only rarely done in practice (Soetaert et al., 2000—but see Luff & Moll, 2004 for an exception). Moll & Radach (2003) reported that most of the three-dimensional models applied to the North Sea either included no benthic module or only a crude parameterization of the benthic–pelagic coupling. The potential artifacts that this omission or simplification may create, and some simple models describing sediment–water interactions, have been discussed by Soetaert et al. (2000).

In the modeling application described here, the three dimensions were compressed into one vertical dimension. Due to that, the ability to *explicitly* describe the net import of new, external nitrogen due to hydrodynamic processes is lost, such that it has to be imposed. The omission of seasonality in hydrodynamic inputs, however, allows the emphasis to shift toward a more detailed representation of benthic processes, and thus to assess their impact on ecosystem functioning.

If it is assumed that nitrogen losses are immediately compensated by lateral inputs, the model is a

computationally very effective tool for estimating ecosystem dynamics in which pelagic and benthic processes are in (a recurrent) equilibrium. By applying the model over a eutrophication gradient, a relationship is obtained between the net lateral input of external nitrogen and the pelagic DIN concentration, representing a proxy for the eutrophication status of shallow-water systems (Fig. 8).

Comparing the results for stratified waters, where net removal is lower, with the well-mixed scenario shows that, as less nitrogen is removed, total pelagic nitrogen concentrations are higher at the same loadings. At intermediate inputs, the difference is largest, with pelagic DIN concentrations up to about 10 mmol m^{-3} higher for the same nitrogen loading (Fig. 8). This supports the idea that the more efficient nitrogen regeneration under water-column anoxia promotes the eutrophication potential of the waters and vice versa, as has been postulated for nitrogen (Childs et al., 2002) and phosphorus (Ingall & Jahnke, 1997; Wallmann, 2003).

The long-term effect of a reduction in nitrogen load on the pelagic nitrogen concentration can also be estimated. Depending on the initial conditions, a 50% reduction in loadings ultimately results in new equilibrium conditions which have more than a proportionate reduction (57–68%) in pelagic DIN concentration. The benefit is slightly larger at lower initial eutrophication conditions (Fig. 8).

Of the studies that model the response of shallow systems to oligotrophication (e.g., Skogen et al., 2004), few include the detailed response of the sediment (Lancelot & Billen, 1985). In accordance with the latter study, the model simulations here of 50% reductions in nutrient loading clearly show that internal loading of nitrogen from the sediments needs to be considered when restoring these systems from severe eutrophication. The storage of nutrients in the sediments significantly delays the system's recovery, but significant changes in the eutrophication status already take place at a time scale of 10–20 years. A similar delay in system recovery to reduced phosphorus loading by internal loading in the sediment has been shown for certain lakes (Jeppesen et al., 2005).

Finally, although the modeling results presented here are hopefully informative and thought provoking, the authors do not expect or pretend them to be generally valid. By restricting the model to a 50-m-

deep water column, by using specific parameter values focusing on nitrogen (excluding phosphorus and silicate), and by omitting certain processes that may be important in some regions (i.e., N_2 formation in the water column), the model mainly serves to demonstrate the importance of benthic–pelagic coupling under these specific conditions. However, it is hoped that this study will provide impetus for the inclusion of more detail on the benthic compartment when modeling eutrophication and oligotrophication in shallow waters, in particular regarding the intimate feedbacks between nutrient regeneration efficiency and benthic animals.

Conclusions

In shallow areas, sediments have a large effect on water-column nitrogen dynamics. They determine the ultimate DIN concentration in the water column in winter and slow down the long-term effects of nutrient reduction scenarios. In order to realistically reproduce these processes in simulation models, an adequate representation of sediment–water exchange is necessary.

Acknowledgments The authors thank the organizing committee, and especially Jesper H. Andersen, for inviting them to The International Symposium on Research and Management of Eutrophication in Coastal Ecosystems. This is publication no. 4085 of the Netherlands Institute of Ecology, Centre for Estuarine and Marine Ecology (NIOO-CEME). Two anonymous reviewers are thanked for constructive criticism.

Open Access This article is distributed under the terms of the Creative Commons Attribution Noncommercial License which permits any noncommercial use, distribution, and reproduction in any medium, provided the original author(s) and source are credited.

References

Aller, R. C., 1980. Quantifying solute distribution in the bioturbated zone of marine sediments by defining an average microenvironment. *Geochimica Et Cosmochimica Acta* 44: 1955–1965.

Alongi, D. M., 2005. Ecosystem types and processes. In Robinson, A. R. & K. Brink (eds), *The Sea*, Vol. 13, *The Global Coastal Ocean: Multiscale Interdisciplinary Processes*. Harvard University Press, Cambridge: 317–351.

Boudreau, B. P., 1996. *Diagenetic Models and Their Implementation. Modelling Transport and Reactions in Aquatic Sediments*. Springer, Berlin.

Brockmann, U. H., R. W. P. M. Laane & H. Postma, 1990. Cycling of nutrient elements in the North Sea. *Netherlands Journal of Sea Research* 26: 239–264.

Brown, P. N., G. D. Byrne & A. C. Hindmarsh, 1989. VODE: a variable coefficient ODE solver. *SIAM Journal of Scientific and Statistical Computing* 10: 1038–1051.

Burdige, D. J., 2006. *Geochemistry of Marine Sediments*. Princeton University Press, Princeton.

Cerco, C. F. & T. Cole, 1993. 3-Dimensional eutrophication model of Chesapeake Bay. *Journal of Environmental Engineering-ASCE* 119: 1006–1025.

Chen, C., D. A. Wiesenburg & L. Xie, 1997. Influences of river discharge on biological production in the inner shelf: a coupled biological and physical model of the Louisiana-Texas shelf. *Journal of Marine Research* 55: 293–320.

Childs, C. R., N. N. Rabalais, R. E. Turner & L. M. Proctor, 2002. Sediment denitrification in the Gulf of Mexico zone of hypoxia. *Marine Ecology Progress Series* 240: 285–290.

Christensen, J. P., 1994. Carbon export from continental shelves, denitrification and atmospheric carbon dioxide. *Continental Shelf Research* 14: 547–576.

de Bie, M. J. M., J. J. Middelburg, M. Starink & H. J. Laanbroek, 2002. Factors controlling nitrous oxide at the microbial community and estuarine scale. *Marine Ecology Progress Series* 240: 1–9.

Gaspar, P., Y. Grégoris & J.-M. Lefevre, 1990. A simple eddy-kinetic energy model for simulations of the oceanic vertical mixing: tests at Station Papa a long-term upper ocean study site. *Journal of Geophysical Research* 95(C9): 16179–16193.

Heip, C. H. R., N. K. Goosen, P. M. J. Herman, J. Kromkamp, J. J. Middelburg & K. Soetaert, 1995. Production and consumption of biological particles in temperate tidal estuaries. *Oceanography and Marine Biology—An Annual Review* 33: 1–149.

Ingall, E. & R. Jahnke, 1997. Influence of water-column anoxia on the elemental fractionation of carbon and phosphorus during sediment diagenesis. *Marine Geology* 139: 219–229.

Jeppesen, E., M. Sondergaard, J. P. Jensen, K. E. Havens, O. Anneville, L. Carvalho, M. F. Coveney, R. Deneke, M. T. Dokulil, B. Foy, D. Gerdeaux, S. E. Hampton, S. Hilt, K. Kangur, J. Kohler, E. H. H. R. Lammens, T. L. Lauridsen, M. Manca, M. R. Miracle, B. Moss, P. Noges, G. Persson, G. Phillips, R. Portielje, C. L. Schelske, D. Straile, I. Tatrai, E. Willen & M. Winder, 2005. Lake responses to reduced nutrient loading—an analysis of contemporary long-term data from 35 case studies. *Freshwater Biology* 50: 1747–1771.

Joint, I. & A. Pomroy, 1993. Phytoplankton biomass and production in the southern North Sea. *Marine Ecology Progress Series* 99: 169–182.

Jørgensen, B. B., 1983. Processes at the sediment–water interface. In Bolin, B. & R. B. Cook (eds), *The Major Biogeochemical Cycles and their Interactions*. SCOPE, Stockholm: 477–515.

Josefson, A. B. & B. Widbom, 1988. Differential response of benthic macrofauna and meiofauna to hypoxia in the Gullmar-Fjord basin. *Marine Biology* 100: 31–40.

Justic, D., N. N. Rabalais & R. E. Turner, 2005. Coupling between climate variability and coastal eutrophication:

- evidence and outlook for the northern Gulf of Mexico. *Journal of Sea Research* 54: 25–35.
- Kemp, W. M., P. Sampou, J. Caffrey, M. Mayer, K. Henriksen & W. R. Boynton, 1990. Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnology and Oceanography* 35: 1545–1563.
- Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M. Glibert, J. D. Hagy, L. W. Harding, E. D. Houde, D. G. Kimmel, W. D. Miller, R. I. E. Newell, M. R. Roman, E. M. Smith & J. C. Stevenson, 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303: 1–29.
- Kriest, I. & G. T. Evans, 1999. Representing phytoplankton aggregates in biogeochemical models. *Deep-Sea Research I* 46: 1841–1859.
- Kuypers, M. M. M., G. Lavik, D. Woebken, M. Schmid, B. M. Fuchs, R. Amann, B. B. Jørgensen & M. S. M. Jetten, 2005. Massive nitrogen loss from the Benguela upwelling system through anaerobic ammonium oxidation. *Proceedings of the National Academy of Sciences of the United States of America* 102: 6478–6483.
- Lancelot, C. & G. Billen, 1985. Carbon–nitrogen relationship in nutrient metabolism of coastal marine ecosystem. *Advances in Aquatic Microbiology* 3: 263–321.
- Luff, R. & A. Moll, 2004. Seasonal dynamics of the North Sea sediments using a three-dimensional coupled sediment–water model system. *Continental Shelf Research* 24: 1099–1127.
- Meysman, F. J. R., O. Galaktionov, B. Gribsholt & J. J. Middelburg, 2006. Bioirrigation in permeable sediments: advective pore water flow induced by burrow ventilation. *Limnology and Oceanography* 51: 142–156.
- Middelburg, J. J. & K. Soetaert, 2005. The role of sediments in shelf ecosystem dynamics. In Robinson, A. R. & K. Brink (eds), *The Sea, Vol. 13, The Global Coastal Ocean Multiscale Interdisciplinary Processes*. Harvard University Press, Cambridge: 353–373.
- Middelburg, J. J., K. Soetaert, P. M. J. Herman & C. H. R. Heip, 1996. Denitrification in marine sediments: a model study. *Global Biogeochemical Cycles* 10: 661–673.
- Middelburg, J. J., K. Soetaert & P. M. J. Herman, 1997. Empirical relationships for use in global diagenetic models. *Deep-Sea Research I* 44: 327–344.
- Middelburg, J. J., C. M. Duarte & J. P. Gattuso, 2005. *Respiration in coastal benthic communities*. In del Giorgio, P. A. & P. J. B. Williams (eds), *Respiration in Aquatic Ecosystems*. Oxford University Press, New York: 206–224.
- Moll, A. & G. Radach, 2003. Review of three-dimensional ecological modelling related to the North Sea shelf system—Part 1: models and their results. *Progress in Oceanography* 57: 175–217.
- Naqvi, S. W. A., R. J. Noronha & C. V. G. Reddy, 1982. Denitrification in the Arabian Sea. *Deep-Sea Research A* 29: 459–469.
- Officer, C. B., R. B. Biggs, J. L. Taft, L. E. Cronin, M. Tyler & W. R. Boynton, 1984. Chesapeake Bay anoxia: origin, development, and significance. *Science* 223: 22–27.
- Otto, L., J. T. F. Zimmerman, G. K. Furnes, M. Mork, R. Saetre & G. Becker, 1990. Review of the physical oceanography of the North Sea. *Netherlands Journal of Sea Research* 26: 161–238.
- Rabalais, N. N., 2005. Eutrophication. In Robinson, A. R. & K. Brink (eds), *The Sea, Vol. 13, The Global Coastal Ocean: Multiscale Interdisciplinary Processes*. Harvard University Press, Cambridge: 821–866.
- Rabalais, N. N., R. E. Turner & W. J. Wiseman, 2002. Gulf of Mexico hypoxia, aka ‘The dead zone’. *Annual Review of Ecology and Systematics* 33: 235–263.
- Rabouille, C., F. T. Mackenzie & L. M. Ver, 2001. Influence of the human perturbation on carbon, nitrogen, and oxygen biogeochemical cycles in the global coastal ocean. *Geochimica Et Cosmochimica Acta* 65: 3615–3641.
- Skogen, M. D., H. Soiland & E. Svendsen, 2004. Effects of changing nutrient loads to the North Sea. *Journal of Marine Systems* 46: 23–38.
- Soetaert, K., P. M. J. Herman & J. J. Middelburg, 1996. A model of early diagenetic processes from the shelf to abyssal depths. *Geochimica Et Cosmochimica Acta* 60: 1019–1040.
- Soetaert, K., J. J. Middelburg, P. M. J. Herman & K. Buis, 2000. On the coupling of benthic and pelagic biogeochemical models. *Earth-Science Reviews* 51: 173–201.
- Soetaert, K., P. M. J. Herman, J. J. Middelburg, C. H. R. Heip, C. L. Smith, P. Tett & K. Wild-Allen, 2001. Numerical modelling the shelf break ecosystem: reproducing benthic and pelagic measurements. *Deep-Sea Research II* 48: 3141–3177.
- Soetaert, K., V. deClippele & P. Herman, 2002. Femme, a flexible environment for mathematically modelling the environment. *Ecological Modelling* 151: 177–193.
- Soetaert, K., J. J. Middelburg, C. Heip, P. Meire, S. Van Damme & T. Maris, 2006. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, the Netherlands). *Limnology and Oceanography* 51: 409–423.
- Tett, P. & M. R. Droop, 1988. Cell quota models and planktonic primary production. In Wimpenny, J. W. T. (ed.), *CRC Handbook of Laboratory Model Ecosystems for Microbial Ecosystems*. CRC Press, Florida: 177–233.
- Van Cappellen, P. & R. A. Berner, 1988. A mathematical model for the early diagenesis of phosphorus and fluorine in marine sediments—apatite precipitation. *American Journal of Science* 288: 289–333.
- van Raaphorst, W. & V. N. de Jonge, 2004. Reconstruction of the total N and P inputs from the IJsselmeer into the western Wadden Sea between 1935 and 1998. *Journal of Sea Research* 51: 109–131.
- Wallmann, K., 2003. Feedbacks between oceanic redox states and marine productivity: a model perspective focused on benthic phosphorus cycling. *Global Biogeochemical Cycles* 17: art. No. 1084.
- Wollast, R., 1998. Evaluation and comparison of the global carbon cycle in the coastal zone and in the open ocean. In Brink, K. H. & A. R. Robinson (eds), *The Sea, The Global Coastal Ocean, Processes and Methods*. Wiley, New York: 213–252.

Reference conditions for phytoplankton at Danish Water Framework Directive intercalibration sites

Peter Henriksen

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 255–262.
DOI: 10.1007/s10750-009-9767-z © Springer Science+Business Media B.V. 2009

Abstract Phytoplankton is one of the biological quality elements included in the EU Water Framework Directive (WFD). Classification of water quality according to the WFD is based on the deviation of the present conditions from reference conditions. Given the lack of data from pristine conditions, this study used approximately 100-year-old measurements of Secchi depths from Danish waters in combination with relationships between Secchi depth and chlorophyll *a* (as a proxy for phytoplankton biomass) obtained from recent monitoring to calculate ‘historical’ or reference chlorophyll *a* (Chl-*a*) concentrations. Historical Secchi depth data were available for 9 out of the 11 Danish WFD intercalibration sites. At eight of the sites, reference summer (May–September) Chl-*a* concentrations were in the range 0.7–1.2 $\mu\text{g l}^{-1}$. At one site, west of Bornholm in the western Baltic Sea, historical Secchi depth measurements date back to only the late 1950s corresponding to a calculated Chl-*a* concentration of

1.3 $\mu\text{g l}^{-1}$. This value cannot be considered representative of reference conditions.

Keywords Reference conditions · Water Framework Directive · Chlorophyll · Secchi depth · Phytoplankton

Phytoplankton is one of the Water Framework Directive (WFD) biological quality elements. The classification of water quality by means of phytoplankton should be based on phytoplankton biomass, composition, and abundance and, in addition, phytoplankton blooms frequency and intensity. For all of these phytoplankton parameters, the deviation from reference or undisturbed conditions is a measure of the water quality.

Danish waters have been heavily impacted by human activity for a long time. Thus, it is not possible to obtain reference conditions from data obtained from the present marine-monitoring programs, which were generally initiated during the 1970s or later. Some early records of phytoplankton from Danish waters are available (e.g., Petersen, 1903; Ostefeld, 1913). However, the phytoplankton data presented are qualitative, or at best semi-quantitative, and collected using plankton net samples rather than modern techniques, thus precluding a direct comparison with recent data.

In contrast to data on phytoplankton parameters, a number of approximately 100-year-old measurements

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems, 20–23 June 2006, Nyborg, Denmark

P. Henriksen (✉)
Department of Marine Ecology, National Environmental Research Institute, Frederiksborgvej 399, 4000 Roskilde, Denmark
e-mail: pet@dmu.dk

of Secchi depth are available for Danish waters. Secchi depth, or water clarity, is dependent on the abundance or biomass of phytoplankton, commonly expressed by the concentration of Chl-*a*, present in the water. However, the relationship between phytoplankton Chl-*a* and Secchi depth is not linear owing to the influence of, for example, suspended particles and dissolved organic matter (DOM) on the water transparency (Kirk, 1994).

This study aimed at establishing preliminary reference conditions for phytoplankton biomass, expressed as Chl-*a*, using historical Secchi depth measurements and relationships between Secchi depth and Chl-*a* obtained from recent monitoring data from the geographical areas included as Danish sites in the EU WFD intercalibration. Due to the long history of human activity and dense population in Denmark, conditions 100 years ago cannot be considered pristine. However, data from this time period represent the oldest pelagic measurements available and inputs of nutrients to the marine areas were minor in comparison to the post-World War Two period of intensified agriculture and use of fertilizers.

Data on Secchi depth and Chl-*a* concentrations (in 1 m depth) were collected from the Danish intercalibration sites (Fig. 1) as part of the Danish National Aquatic Monitoring and Assessment Programme (Conley et al., 2002). While comparison of Secchi depth with Chl-*a* concentrations in integrated samples representing a large part of the upper water column would be optimal, a comparison with samples taken at 1 m depth was chosen to follow procedures determined for the EU WFD intercalibration. For each intercalibration site, data from several monitoring stations were included. Time periods covered by the monitoring program are shown in Table 1.

Secchi depth measurements taken from Danish waters during the early 1900s (Aarup, 2002; data available from the ICES database at <http://www.ices.dk/Ocean/project/secchi/>) were used for comparison with the relationships found in recent monitoring data. The geographical distribution of these measurements is shown in Fig. 1. In addition, series of Secchi depth measurements taken from lightships in northern Kattegat and in the Great Belt during the 1960s and 1970s were available for comparison with recent data (Table 1).

For all intercalibration sites, recent monitoring data were used to establish Secchi depth–Chl-*a*

relationships (Fig. 2). In addition to Chl-*a*, suspended particles and DOM will affect light absorption in the water column and thus Secchi depth (Kirk, 1994). Therefore, the relationships between Secchi depth and Chl-*a* showed a very large scatter with a range of Chl-*a* values corresponding to each Secchi depth, in particular at the numerically low Secchi depths. To compensate for the lack of complementary data on factors other than Chl-*a* that influence Secchi depth, relationships were established using boundary functions describing the upper bounds of the distributions (Blackburn et al., 1992; Krause-Jensen et al., 2000). The rationale behind this approach was that the higher the Chl-*a* values at an individual Secchi depth, the higher the contribution from Chl-*a* to the total light attenuation and thereby influence on Secchi depth. Furthermore, analyses were only performed on summer samples (May–September) to reduce the likelihood of strong wind events, which potentially lead to heavy resuspension of sediment. During this time period, the growth of phytoplankton is predicted to be limited by the availability of nutrients rather than light, and Chl-*a* is expected to be a major contributor to light attenuation in the water column.

In shallow areas where the cover and depth limit of benthic vegetation has changed from the time of the historical Secchi depth measurements until now, the contribution from resuspended particles to total water column light attenuation may have changed. Such potential changes have not been included in this study.

Corresponding measurements of Chl-*a* concentrations and Secchi depths were grouped in bins representing 1 m Secchi depth intervals. For each 1-m Secchi depth interval, the 90th percentile of the corresponding Chl-*a* measurements was calculated. Subsequently, correlations between the average depth of each Secchi depth bin (0.5 m, 1.5 m, 2.5 m, etc.) and the 90th percentiles of Chl-*a* concentrations were established from regression analyses assuming exponential relationships (Fig. 2).

For all intercalibration sites, a significant correlation was found between the Secchi depth and the 90th percentiles of Chl-*a* concentrations found within each Secchi depth interval (Fig. 2). For areas with historical Secchi depth data, ‘historical Chl-*a* values’ were calculated using the 90th percentile correlations and averages of the historical Secchi depth measurements (Tables 1, 2). Thus, the calculated ‘historical’ Chl-*a*

Fig. 1 *Top panel* WFD intercalibration sites in Danish waters where Chl-*a* and Secchi depth data are available. 1: Inner Wadden Sea, 2: Outer Wadden Sea, 3: Hirtshals, 4: Northern Kattegat, 5: Århus Bay, 6: North of Funen, 7: Dybsø Fjord, 8: Hjelm Bay, 9: Fakse Bay, 10: Northern and central part of the Sound, 11: West of Bornholm. Each intercalibration site includes several monitoring stations. *Bottom panel* Historical Secchi depth measurements from Danish waters (Aarup, 2002). Each marker may represent several measurements

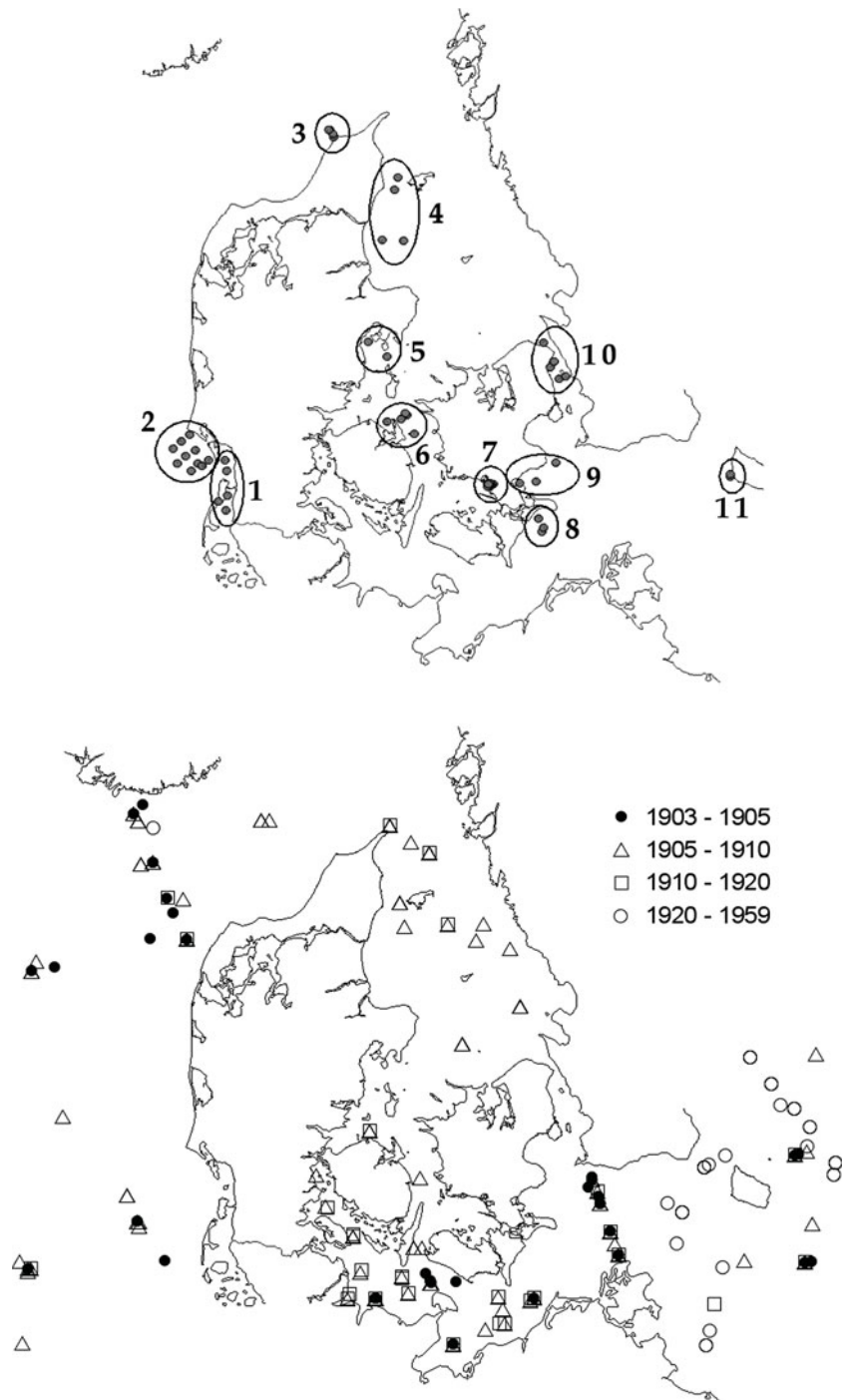


Table 1 Data series used for the calculation of reference summer (May–September) Chl-*a* concentrations at the Danish WFD intercalibration sites

| Site | Recent data series ^a | Historical data series | <i>n</i> | Historical Secchi depths (m) | | | | 90th percentile model |
|-------------------|---------------------------------|--|----------|------------------------------|------|------|-----|-------------------------------------|
| | | | | Min. | Max. | Avg. | SD | |
| West of Bornholm | 1988–2003 | West of Bornholm (1958–1959) | 5 | 8 | 10 | 8.8 | 0.8 | Chl- <i>a</i> = 7.501e-0.128*Secchi |
| Fakse Bay | 1972–2003 | Falsterbo-Rügen transect (1904–1911) | 30 | 7 | 14 | 10.1 | 1.8 | Chl- <i>a</i> = 11.04e-0.207*Secchi |
| North of Funen | 1974–2003 | North of Funen (1907–1911) | 3 | 7 | 10 | 8.7 | 1.5 | Chl- <i>a</i> = 8.517e-0.119*Secchi |
| | | North of Funen (Ostenfeld, 1908) ^b | | | | 12.8 | | |
| | | Great Belt (1909) | 1 | 10 | 10 | 10.0 | | |
| | | Little Belt (1907) | 1 | 9 | 9 | 9.0 | | |
| | | l/s Halskov Rev (1960s) | 41 | 5 | 11 | 7.1 | 1.6 | |
| | | l/s Halskov Rev (1970–1971) | 12 | 5 | 8 | 6.3 | 0.9 | |
| Hirtshals | 1983–2003 | Skagerrak (1907–1909) | 3 | 10 | 12 | 11.3 | 1.2 | Chl- <i>a</i> = 31.19e-0.272*Secchi |
| | | Hanstholm-Norway transect (1904–1911) | 30 | 8 | 28 | 14.8 | 4.2 | |
| | | Estimated historical Secchi depth ^c | | | | 10.9 | | |
| Hjelm Bay | 1974–2004 | Western Baltic–Femern (1903–1912) | 19 | 8 | 13 | 10.4 | 1.6 | Chl- <i>a</i> = 10.16e-0.181*Secchi |
| | | Falsterbo-Rügen transect (1904–11) | 30 | 7 | 14 | 10.1 | 1.8 | |
| Northern Kattegat | 1972–2004 | Northern Kattegat (1908–1911) | 17 | 5 | 14 | 10.5 | 2.0 | Chl- <i>a</i> = 23.78e-0.275*Secchi |
| | | l/s Anholt Knob (1960s) | 74 | 6.5 | 18.5 | 10.2 | 2.4 | |
| | | l/s Laesoe Rende (1960s) | 31 | 5 | 13 | 8.0 | 2.1 | |
| | | l/s Aalborg Bay (1960s) | 36 | 4.7 | 10.7 | 7.8 | 1.4 | |
| | | l/s Anholt Knob (1970s) | 24 | 6.5 | 14.5 | 9.4 | 2.0 | |
| | | l/s Aalborg Bay (1970s) | 10 | 4.7 | 9.7 | 6.8 | 1.7 | |
| Outer Wadden Sea | 1982–2003 | Outer Wadden Sea (1904–1910) | 8 | 9 | 18 | 13.0 | 2.8 | Chl- <i>a</i> = 30.90e-0.296*Secchi |
| | | Estimated historical Secchi depth ^d | | | | 8.5 | | |
| Århus Bay | 1971–2003 | Central Kattegat (1908–1910) | 3 | 11 | 15 | 12.3 | 2.3 | Chl- <i>a</i> = 10.15e-0.142*Secchi |
| | | Southern Kattegat (1907–1910) | 3 | 12 | 16 | 13.7 | 2.1 | |

90th percentile models were derived from exponential relationships between Secchi depths and Chl-*a* concentrations in the recent data. Lightships are denoted l/s

^a Some years are missing from the data series

^b Based on observations of depth limit of eelgrass (Ostenfeld, 1908) converted to Secchi depth according to Nielsen et al. (2002)

^c Historical Secchi depths originate further offshore than the Hirtshals intercalibration site. From Aarup (2002), it is estimated that Secchi depths at the Hirtshals site will be ~75% of those further offshore

^d Historical Secchi depths originate further offshore than the Outer Wadden Sea intercalibration site. From Aarup (2002), it is estimated that Secchi depths at the Outer Wadden Sea site will be ~65% of those further offshore

values represent estimates of the 90th percentile in the historical data. ‘Historical’ summer averages were calculated from the ‘historical’ 90th percentiles assuming a similar ratio between averages and 90th percentiles in historical and recent (monitoring) data (Table 3).

Historical Secchi depths used for the Hirtshals and outer Wadden Sea intercalibration sites originated further offshore than the intercalibration sites. Based

on Aarup (2002), historical Secchi depths from these two sites were assumed to have been approximately 75% and 65%, respectively, of the offshore values.

For all areas, the calculated historical Chl-*a* concentrations were lower than recent values. Thus, the averages of recent (2000–2005) Chl-*a* measurements were from 1.2-fold (North of Funen) to 5.4-fold (outer Wadden Sea) higher than the calculated historical values. The historical Secchi depth

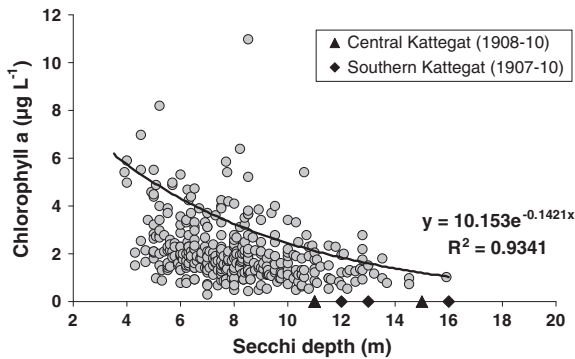


Fig. 2 Corresponding summer (May–September) values of Secchi depth and Chl-*a* concentration at the EU WFD intercalibration site Århus Bay. The regression model is based on 90th percentile values of Chl-*a* within 1-m Secchi depth intervals. Additional markers indicate historical Secchi depth measurements from Aarup (2002); these markers of historical Secchi depths are placed on the *x*-axis as no corresponding Chl-*a* values were available

measurements available for the area north of Funen ranged from 7 to 10 m (Table 1). However, data on the depth limit of eelgrass (*Zostera marina*) from the same area and time period showed the growth of eelgrass down to approximately 10.4 m depth (Ostenfeld, 1908). Thus, the few (five) historical Secchi depth measurements available for that area seem to underestimate the Secchi depth resulting in too high a Chl-*a* reference condition for that area. This is further supported by the much deeper ‘historical’ Secchi depths in Århus Bay (Table 1) located close to the area north of Funen and a general similarity in the calculated Chl-*a* reference conditions for the other intercalibration sites showing similar Secchi depth–Chl-*a* relationships (northern Kattegat, northern part of the Sound, Hjelm Bay and Fakse Bay, Table 1). Using the relation between depth limit of eelgrass and Secchi depth given by Nielsen et al.

Table 2 Recent and historical summer (May–September) averages of Secchi depth at the Danish EU WFD intercalibration sites

| | Secchi depth (m) | | | | | | |
|-------------------------------|--------------------|-----------------------|-----------------------|-----------------------|-----------------------|-----------------------|--------------------------------------|
| | Historical average | Average of 1960s data | Average of 1970s data | Average of 1980s data | Average of 1990s data | Average of 2000s data | 2000s Data/historical concentrations |
| Northern Kattegat | 10.5 | 9.1 | 8.7 | 6.7 | 7.6 | 8.5 | 0.8 |
| Fakse Bay | 10.1 | | | 6.7 | 6.9 | 6.7 | 0.7 |
| Hjelm Bay | 10.4 | | | 5.9 | 6.8 | 6.7 | 0.6 |
| Hirtshals ^a | 10.9 | | | | 7.6 | 7.8 | 0.7 |
| Århus Bay | 13.0 | | 6.7 | 6.8 | 8.2 | 8.9 | 0.7 |
| West of Bornholm ^b | 8.8 | | | 7.4 | 11.4 | 11.4 | 1.3 |
| Outer Wadden Sea ^c | 8.5 | | | 3.0 | 3.4 | 3.4 | 0.4 |
| North of Funen | 12.8 ^d | | 7.0 | 6.4 | 6.7 | 7.7 | 0.6 |
| Northern part of the Sound | | | | 7.2 | 7.3 | 8.1 | |
| Dybsø Fjord | | | | 1.6 | 1.8 | 2.2 | |
| Inner Wadden Sea | | | | | 1.7 | 1.8 | |

Historical measurements are taken from Aarup (2002), while recent values originate from the modern monitoring data at the individual sites

^a Derived from historical offshore Secchi depths and the assumption that Secchi depths at the intercalibration site are 75% of those offshore

^b Derived from data collected during 1958–1959

^c Derived from historical offshore Secchi depths and the assumption that Secchi depths at the intercalibration site are 65% of those offshore

^d Based on observations of the depth limit of eelgrass (Ostenfeld, 1908) converted to Secchi depth according to Nielsen et al. (2002)

Table 3 Recent and calculated 'historical' summer (May–September) averages of Chl-*a* concentrations and reference conditions for summer Chl-*a* derived from the 10th and 20th percentiles of monitoring data at the Danish EU WFD intercalibration sites

| Chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$) | | | | | | | | | |
|---|-------------------------------|--|--|-----------------------|-----------------------|-----------------------|-----------------------|--------------------------------------|--|
| | Calculated historical average | Reference conditions based on 10th percentile of monitoring data | Reference conditions based on 20th percentile of monitoring data | Average of 1970s data | Average of 1980s data | Average of 1990s data | Average of 2000s data | 2000s Data/historical concentrations | |
| Northern Kattegat | 0.7 | 0.5 | 0.5 | 1.2 | 2.4 | 2.0 | 1.1 | 1.6 | |
| Fakse Bay | 0.8 | 0.9 | 1.1 | 2.2 | 1.9 | 2.4 | 1.9 | 2.4 | |
| Hjelm Bay | 1.0 | 0.8 | 1.0 | | | 2.2 | 1.6 | 1.6 | |
| Hirtshals ^a | 0.9 | 0.5 | 0.5 | | | | 3.1 | 3.4 | |
| Århus Bay | 0.9 | 0.9 | 1.2 | 1.3 | 2.4 | 2.2 | 1.8 | 2.0 | |
| West of Bornholm ^b | (1.3) | 0.6 | 0.6 | | 1.6 | 1.8 | 1.4 | 1.1 | |
| Outer Wadden Sea ^c | 1.1 | 2.0 | 3.0 | | 4.8 | 8.5 | 5.9 | 5.4 | |
| North of Funen | 1.2 ^b | 1.3 | 1.6 | 2.0 | 3.9 | 2.5 | 2.2 | 1.2 | |
| Northern part of the Sound ^e | 0.9 | 0.7 | 0.9 | 1.8 | 2.3 | 2.1 | 1.5 | 1.7 | |
| Dybsø Fjord | | 0.6 | 0.9 | | 2.5 | 2.3 | 1.3 | | |
| Inner Wadden Sea | | 3.3 | 4.5 | | 6.2 | 9.6 | 8.4 | | |

'Historical' averages were derived from calculated 'historical' 90th percentiles by multiplying the 90th percentiles with the ratio 'average' to '90th percentile' of the modern monitoring data at the individual site

^a Derived from historical offshore Secchi depths and the assumption that Secchi depths at the intercalibration site are 75% of those offshore

^b Derived from data collected during 1958–1959. Not comparable to other 'historical' values

^c Derived from historical offshore Secchi depths and the assumption that Secchi depths at the intercalibration site are 65% of those offshore

^d Based on observations of depth limit of eelgrass (Ostenfeld, 1908) converted to Secchi depth according to Nielsen et al. (2002)

^e Based on a modeled 'pristine' scenario (Øresundsvandsamarbejdet, 2004)

(2002), a depth limit of 10.4 m would correspond to a Secchi depth of 12.8 m and subsequently a 'historical' average Chl-*a* concentration of $1.2 \mu\text{g l}^{-1}$. This value is in better agreement with reference values from the other sites in the Kattegat area and is therefore suggested as a better estimate of Chl-*a* reference conditions north of Funen (Tables 2, 3).

Secchi depth measurements obtained from lightships in the northern Kattegat and the Great Belt during the 1960s and 1970s illustrate a reduction in water clarity from about 1900 until the 1960s and a further reduction from the 1960s until the 1970s (Table 2). Data from the intercalibration sites show that water clarity has improved since the 1980s and present conditions in the northern Kattegat are comparable to those in the 1970s (Table 2). Eutrophication leading to increased biomass of phytoplankton is expected to be a major cause for the changed Secchi depth regimes. While Chl-*a* measurements are available only from the 1980s and onwards, the early measurements of primary production made by Steemann Nielsen in the Kattegat during the 1950s support the hypothesis that the reduced water clarity is coupled to eutrophication. Thus, Richardson & Heilmann (1995) calculated a two to threefold increase in primary production in the Kattegat from the 1950s until the period 1984–1993.

Historical Secchi depth measurements were not available for the northern part of the Sound. A modeled 'pristine' scenario for the Sound (Øresundsvandsamarbejdet, 2004) suggested that recent summer (June–August) Chl-*a* concentrations are approximately 75–90% higher than those 150 years ago. Assuming a similar development for the period May–September and a 70% increase over the last century, it is possible to estimate a 'historical' average Chl-*a* concentration of $0.9 \mu\text{g l}^{-1}$ for this area.

No historical data were available for the Dybsø Fjord or Inner Wadden Sea intercalibration sites and therefore reference conditions could not be established. At the site west of Bornholm, historical Secchi depth data date back to only the late 1950s and thus are not directly comparable to reference conditions established for the other areas.

In a recent project on the development of tools for the assessment of eutrophication in the Baltic Sea, Finland estimated the boundary between high and good ecological status according to the EU WFD as the 10th percentile of recent monitoring data

(Andersen et al., 2006). For comparison, the 10th and 20th percentiles of the Chl-*a* measurements from the monitoring programs have been included in Table 3. For most of the Danish intercalibration sites, the 10th or 20th percentiles of monitoring data were in good agreement with estimated reference conditions. However, at Hirtshals and west of Bornholm, the 10th and 20th percentiles were much lower than the estimated reference conditions, and in the outer Wadden Sea the 10th and 20th percentiles were approximately 2- and 3-fold higher than the estimated reference Chl-*a* concentration (Table 3). It should be emphasized that even though the 10th or 20th percentile method showed reasonable agreement with estimated reference Chl-*a* concentrations at several of the Danish intercalibration sites, this approach should not be taken in more enclosed and heavily eutrophied areas where all recent Chl-*a* measurements will be expected to exceed reference conditions.

This study provides a first step in establishing reference conditions for phytoplankton in Danish waters. Chl-*a* is the most commonly used proxy for phytoplankton biomass and it was chosen as the first phytoplankton metric for the EU WFD intercalibration process. However, in addition to Chl-*a*, phytoplankton species composition, abundance, and bloom frequency/intensity should be included in the future assessment of water quality. At present, reference conditions are not available for these indicators.

References

- Aarup, T., 2002. Transparency of the North Sea and Baltic Sea—a Secchi depth data mining study. *Oceanologia* 44: 323–337.
- Andersen, J. H., J. Aigars, U. Claussen, B. Håkanson, H. Karup, M. Laamanen, E. Lysiak-Pastuszak, G. Martin & G. Nausch, 2006. Development of tools for assessment of eutrophication in the Baltic Sea. *Baltic Sea Environment Proceedings* No. 104, Helsinki Commission.
- Blackburn, T. M., J. H. Lawton & J. N. Perry, 1992. A method of estimating the slope of upper bounds of plots of body size and abundance in natural animal assemblages. *Oikos* 65: 107–112.
- Conley, D. J., S. S. Markager, J. Andersen, T. Ellermann & L. M. Svendsen, 2002. Coastal eutrophication and the Danish national aquatic monitoring and assessment program. *Estuaries* 25: 706–719.
- Kirk, T. O., 1994. *Light and photosynthesis in aquatic ecosystems*. Cambridge University Press, London.

- Krause-Jensen, D., A. L. Middelboe, K. Sand-Jensen & P. B. Christensen, 2000. Eelgrass, *Zostera marina*, growth along depth gradients: upper boundaries of the variation as a powerful predictive tool. *Oikos* 91: 233–244.
- Nielsen, S. L., K. Sand-Jensen, J. Borum & O. Geertz-Hansen, 2002. Depth colonisation of eelgrass (*Zostera marina*) and macroalgae as determined by water transparency in Danish coastal waters. *Estuaries* 25: 1025–1032.
- Øresundsvandsamarbejdet, 2004. Jämförelse mellan “urtida” och “nutida” näringsnivåer i Öresund.—beräkningar utförda med MIKE 3-modell. Öresundsvattensamarbetet, <http://www.oresundsvand.dk> (in Swedish).
- Ostenfeld, C. H., 1908. Ålegræssets (*Zostera marina*)s udbredelse i vore farvande. In Petersen, C. G. J. (ed.), Beretning til Landbrugsministeriet fra den danske biologiske station XVI. Centraltrykkeriet, Copenhagen: 1–61 (in Danish).
- Ostenfeld, C. H., 1913. De danske Farvandes Plankton i Årene 1889–1901. Phytoplankton og protozoer I. Kgl. Danske Vidensk. Selsk. Skr. 7. Række. Naturv. og Math. Afd. IX. 2: 1–478 (in Danish).
- Petersen, C. G. J., 1903. De danske Farvandes Plankton i Årene 1898–1901. Kgl. Danske Vidensk. Selsk. Skrifter, 6. Række. Naturv. og Math. Afd. XII.3. Bianco Lunos Bogtrykkeri. Kjøbenhavn (in Danish).
- Richardson, K. & J. P. Heilmann, 1995. Primary production in the Kattegat: past and present. *Ophelia* 41: 317–328.

Coastal eutrophication research: a new awareness

Carlos M. Duarte

Originally published in the journal *Hydrobiologia*, Volume 629, No. 1, 263–269.
DOI: 10.1007/s10750-009-9795-8 © Springer Science+Business Media B.V. 2009

Abstract An analysis of the contents and conclusions of the papers contained in this issue (*Hydrobiologia* Volume xxx) suggests that a new vision is taking shape that may correspond to an emerging new paradigm in the way we understand and manage coastal eutrophication. This new paradigm emphasizes its global dimension and the connections with other global environmental pressures, and re-evaluates the targets of remedial actions and policies. Eutrophication research must evolve toward a more integrative, ecosystem perspective which requires that it be extended to include impacts beyond primary producers and to examine possible cascading effects and feedbacks involving other components of the ecosystem. A quantitative framework that incorporates the interacting top-down and bottom-up effects in eutrophication models must be urgently developed to guide

diagnostics and establish targets to mitigate coastal eutrophication. The required macroscopic view must also be extended to the managerial and policy frameworks addressing eutrophication, through the development of policies that examine activities in the environment in an integrative, rather than sectorial, manner. Recent evidence of complex responses of coastal ecosystems to nutrient reduction requires that management targets, and the policies that support them, be reconsidered to recognize the complexities of the responses of coastal ecosystems to reduced nutrient inputs, including non-linear responses and associated thresholds. While a predictive framework for the complex trajectories of coastal ecosystems subject to changes in nutrient inputs is being developed, the assessment of managerial actions should be reconsidered to focus on the consideration of the status achieved as the outcome of nutrient reduction plans against that possibly derived from a ‘do nothing’ scenario. A proper assessment of eutrophication and the efforts to mitigate it also requires that eutrophication be considered as a component of global change, in addressing both its causes and its consequences, and that the feedbacks between other components of global change (e.g., climate change, overfishing, altered biogeochemical cycles, etc.) be explicitly considered in designing eutrophication research and in managing the problem.

Guest editors: J. H. Andersen & D. J. Conley
Eutrophication in Coastal Ecosystems: Selected papers from
the Second International Symposium on Research and
Management of Eutrophication in Coastal Ecosystems, 20–23
June 2006, Nyborg, Denmark

C. M. Duarte (✉)
IMEDEA (CSIC-UIB), Miquel Marqués 21,
07190 Esporles, Spain
e-mail: carlosduarte@ifisc.uib.es

Keywords Eutrophication · Oligotrophication ·
Global change · Nutrient · Coastal ecosystems ·
Management targets

Introduction

About 15 years have elapsed between the first conference on coastal eutrophication convened in Denmark, in 1993, and the Second International Symposium on Research and Management of Eutrophication in Coastal Ecosystems (Nyborg, Denmark, 20–23 June, 2006), the results of which are collected in this special issue. The symposium addressed a range of topics (Andersen & Conley, 2009) and promoted active discussion among the 200 participants. Whereas the first conference included the definition of ‘eutrophication’ as a noun (Nixon, 1995), the definition now coined by Scott Nixon is that for ‘oligotrophication,’ the antonym of eutrophication (Nixon, 2009). This is symptomatic of the evolution of eutrophication science and the management of eutrophied coastal ecosystems.

In 1993, we were creating an awareness of the spread of eutrophication as a threat to coastal ecosystems, lagging about two decades behind the recognized impacts of eutrophication in lakes. We now have a better appreciation of the scale of problem (e.g., Nixon, 2009), a better understanding of its processes and dynamics (e.g., Cloern, 2001; Boesch, 2002; Howarth & Marino, 2006; Conley et al., 2009; Duarte et al., 2009; Soetaert & Middelburg, 2009), improved diagnostic tools and indicators (e.g., Eyre & Ferguson, 2009; Jaanus et al., 2009), and have taken action to mitigate it (e.g., Nørring & Jørgensen, 2009; Petersen et al., 2009; Savchuk & Wulff, 2009), as clearly exemplified by the papers compiled in this issue. Collectively, these papers portray a major development of the field over the past decade. Although less explicitly, a critical analysis of the contents and conclusions of these papers, along with recent papers published elsewhere, suggests that a new vision is taking shape that may correspond to an emerging new paradigm in the way we understand and manage coastal eutrophication.

In this summary, I will not attempt to list the insights derived from each individual contribution to this special issue, which can best be obtained through an examination of the papers themselves. I will, instead, focus on what these papers collectively inform as the challenge for coastal eutrophication research in the future. I argue that these challenges accrete to conform a new vision of coastal eutrophication that emphasizes its global dimension and connection with other global

environmental pressures, and reconsiders the targets for remedial actions and policies.

Challenges for coastal eutrophication research

Eutrophication research has focused mainly on phytoplankton, but clearly eutrophication impacts on all components of coastal ecosystems. The diagnostics of these impacts require that robust indicators be developed that help assess the responses of coastal ecosystems to eutrophication (e.g., Jaanus et al., 2009). The search for reliable ecosystem-level indicators of eutrophication impacts is not driven by an academic interest in better understanding eutrophication, but by the requirements of legislation and policy to identify and abate coastal eutrophication, as exemplified by the EU Water Framework Directive and the OSPAR Convention (e.g., Claussen et al., 2009; Duarte et al., 2009; Henriksen, 2009). Long-lived components of coastal ecosystems sensitive to degraded water quality, such as seagrass, can be particularly useful because their response integrates variability in nutrient concentrations and water quality, offering a robust diagnostic of change (Duarte, 1995, 2002; Olivé et al., 2009). Such an ecosystem approach to understanding and managing eutrophication requires that research be extended to include impacts beyond primary producers and to examine possible cascading effects and feedbacks involving other components of the ecosystem.

The role of changes in the upper levels of the food web in facilitating eutrophication has received limited attention, but research on eutrophied coral reef ecosystems has clearly shown that the integrity of the fish community can greatly affect the resilience and resistance of coral reefs to eutrophication (Dulvy et al., 2004; Mumby et al., 2006). Indeed, a recent meta-analysis of the role of fish removal in the loss of ecosystem resistance to eutrophication has provided compelling evidence that the depletion of top predators from coastal waters may have increased the vulnerability of coastal ecosystems to eutrophication (Heck & Valentine, 2007). Heck & Valentine (2007) showed that the widespread overharvesting of large consumers, which once played pivotal roles in regulating ecosystem structure and function (Jackson et al., 2001), may trigger indirect effects that may enhance those of eutrophication. Indeed, overfishing and eutrophication

have strong interactive effects. Breitburg et al. (2009) argue that nutrient enrichment and habitat degradation causes reduced yields of fisheries, whereas fisheries exploitation can affect the ability of estuarine systems to process nutrients, enhancing eutrophication symptoms. Whereas general relationships between nutrient inputs and concentrations, and chlorophyll *a* concentrations are available (e.g., Smith, 2006), there is no predictive framework that integrates the interactive top-down and bottom-up forces in the development of eutrophication symptoms. A quantitative framework that incorporates the interacting top-down and bottom-up effects in eutrophication models must be urgently developed to guide diagnostics and establish targets to mitigate coastal eutrophication.

The development of a broader ecosystem approach to coastal eutrophication often stumbles on the difficulties of defining the extent of coastal ecosystems. Coastal ecosystems have clearly defined, objective boundaries inland but their offshore extent is generally unclear, and its delineation remains critical to properly manage eutrophication problems (Kratzer & Tett, 2009; Lessin et al., 2009). Moreover, whereas most approaches focus on land-derived influences (i.e., nutrient inputs) on eutrophication dynamics, forcing derived from interactions with the adjacent offshore waters may determine significant departures from the expected dynamics (e.g., Cloern et al., 2007; Savchuk et al., 2009). This requires that forcing by both inland (human-dominated) and offshore (climate-dominated) processes be considered in predicting and understanding eutrophication dynamics in the coastal zone. For example, long-term analysis of Narragansett Bay provided evidence of climate-induced oligotrophication in the absence of any significant reduction in nutrient concentrations or inputs (Fulweiler & Nixon, 2009).

The above plea for a more integrative ecosystem approach to eutrophication research may be interpreted to reinforce the tradition to study and manage eutrophication at the individual ecosystem level. However, it is becoming increasingly evident that eutrophication must be examined in a much broader context. Nixon (2009) makes a strong case, drawing on advice he received from his mentor, the late H. T. Odum, to examine eutrophication issues at a scale larger than the individual ecosystem, applying a macroscope to better capture the dimensions and controls of the forces that shape eutrophication

problems. This case can be forcefully made by considering the problems of eutrophication-driven hypoxia in the Gulf of Mexico, the causes and dynamics of which have been traced back to agricultural practices in the Corn Belt, thousands of kilometers away from the coastal region where the problem is manifested (Burkart & James, 1999). Moreover, the application of such a macroscope does not need to be restricted to understand the environmental processes, excess nutrient use and export that our concept of eutrophication identifies as the cause of the problem, but it may also incorporate the underlying social dynamics. For example, it has recently been argued that the rapid growth of biofuel crops in the Corn Belt affects the prospect for decreasing the hypoxic area in the Gulf of Mexico, as it is likely to lead to increased fertilizer application in the Corn Belt (Hill et al., 2006).

Indeed, agricultural, fisheries, water quality and even energy policies all affect coastal eutrophication, even though these policies do not recognize such connections and are often issued as if they bore no relationship to the eutrophication problem. Recognizing that excess fertilizer and manure application to agricultural soils is a main driver of coastal eutrophication, many nations have invested important resources to reduce nitrogen effluents. These efforts are currently championed by Denmark, which implemented a number of different Action Plans against water pollution focused on the agricultural sector that have successfully led to a reduction in the total loss of nitrogen from farmland by approximately 50% compared to the mid-1980s (Nørring & Jørgensen, 2009). While the analysis of nitrogen inputs to coastal waters has focused on activities on land, the importance of the fisheries sector in removing nutrients from coastal ecosystems has only recently been identified. Indeed, Maranger et al. (2008) recently reported that fisheries mobilize almost as much nitrogen, bound in the N-rich tissues of marine organisms, from the oceans back to land as that delivered from land to the oceans. The large amount of nitrogen removed with the harvesting of marine animals has led Lindahl & Kollberg (2009) to propose that mussel farming, which has been recognized by Swedish environmental authorities as a possible measure to improve coastal water quality, be subsidized by sectors that emit nutrients, such as agriculture, through the trading of emission quotas. Specifically, Lindahl & Kollberg (2009) argue that

because most of the nutrient supply to coastal waters in many areas of Europe derives from agricultural operations, funding from the EU agro-environmental program should subsidize mussel farming enterprises in the same way that agricultural farmers are supported to reduce nutrient leakage from their farmlands. Again, this requires the development of policies that examine activities in the environment in an integrative, rather than sectorial, manner.

Most importantly, recent evidence of complex responses of coastal ecosystems to nutrient reduction requires that management targets, and the policies that support them, be reconsidered. An examination of the responses of coastal ecosystems to reduced nutrient inputs shows that these responses often deviate from the simple decline in chlorophyll *a* concentrations that is expected based on the underlying assumption of a direct relationship between nutrient inputs and eutrophication symptoms (Kemp et al., 2005; Duarte et al., 2009). Failure to observe the expected improvements in response to the reductions in nutrient inputs can be very frustrating for the public and managers and can undermine their willingness to undertake future reductions; this must be urgently addressed (Duarte et al., 2009). For instance, in Denmark, arguably the country that has led efforts to reduce nutrient inputs, the 43% reduction in the nitrogen load to the marine environment in the period 1989–2003 has resulted in only minor improvements at the ecosystem level, generating the perception of a disproportion between the cost of nutrient reduction plans and the improvements achieved (Nørring & Jørgensen, 2009). Indeed, whereas increased nutrient inputs have clearly led to increased coastal eutrophication, the reverse, oligotrophication (i.e., a reduced production of organic matter; Nixon, 2009), does not appear to be a necessary, nor prevalent, outcome of reduced nutrient inputs.

We have long understood the complexity of factors involved in the eutrophication process (Cloern, 2001; Kemp et al., 2005) but, these complexities have not been adequately conveyed to managers and legislators, who have received a simplified view of a direct control of chlorophyll *a* concentration, as a metric of water quality, by nutrient inputs (Duarte et al., 2009). A new paradigm is emerging that recognizes the complexities of the factors and dynamics involved in the response of coastal ecosystems to reduced nutrient inputs (e.g., Conley et al., 2009; Duarte et al., 2009).

Only now, three decades after eutrophication problems were realized as a major problem affecting coastal ecosystems are data becoming available to show the trajectories of ecosystems following increased and subsequently reduced nutrient inputs (Duarte et al., 2009). Examination of these trajectories reveals that the response of coastal ecosystems to reduced nutrient inputs appears to display complex, non-linear dynamics. A first driver of such non-linearities is the occurrence of thresholds of nutrient inputs beyond which abrupt changes in the response of the ecosystem occur (e.g., Duarte et al., 2009). Such threshold responses have been identified and analyzed in the case of the decline in oxygen in response to increased nutrient concentrations (Conley et al., 2009), whereby the occurrence of hypoxia creates a number of buffers (loss of benthic organisms, build up of sulfides, and others) that render ecosystems who have experienced hypoxia more prone to experience subsequent hypoxia even if nutrient concentrations are reduced (Conley et al., 2009). A second factor is the occurrence of time lags in the responses owing to the accumulation and release of nutrients in the sediments (Soetaert and Middelburg, 2009) or the long time spans involved in the recovery of some inherently slow-growing organisms, such as seagrass (Duarte, 1995). These time lags result in apparent hysteresis in response to nutrient reductions that appear as non-linearities in the trajectories of ecosystem status during increased and reduced nutrient concentrations (Duarte et al., 2009). Finally, non-linearities may derive from shifting baselines altering the relationship between chlorophyll *a* and nutrient inputs over time (Duarte et al., 2009). The trajectories of coastal ecosystems from the onset of eutrophication to the achievement of nutrient reductions occur typically over two to three decades, sufficiently long for other relevant forcing factors, including top predators in the system, climate, hydrology, land use in the catchment, atmospheric carbon dioxide (CO₂) concentrations, etc., to have changed significantly (Duarte et al., 2009), possibly affecting the yield of chlorophyll, or the extent of eutrophication, per unit nutrient in the system. Shifting baselines along the eutrophication-to-oligotrophication transition may result in an apparent failure to achieve the extent of reduction in chlorophyll *a* concentration sought by nutrient reduction programs.

The awareness of the importance of abrupt changes in coastal ecosystems as nutrient inputs

increase beyond critical tipping points is largely a consequence of the increasing availability of time series data extending over three decades to encompass the eutrophication and oligotrophication periods. Indeed, the availability of these time series occurs at a time when new statistical techniques have become available to search for and detect abrupt changes and thresholds (Andersen et al., 2008), providing an opportunity to learn more about the non-linearities affecting the relationship between chlorophyll *a* concentration and changes in nutrient inputs in coastal ecosystems.

While we struggle to understand and develop a predictive framework for the complex trajectories of coastal ecosystems from eutrophication to subsequent oligotrophication, it is imperative that we reassess our advice to managers and, particularly, the targets of nutrient reduction plans. Focusing on the improvements, although unavoidable, can be discouraging. The expectation that coastal ecosystems will return to an original status prior to significant eutrophication, following nutrient reduction efforts has recently been criticized as naïve, because it implies the return to a past situation that may be precluded by major concurrent changes in the global environment, the shifting baselines (Duarte et al., 2009). Hence, a better framework for action may involve consideration of the status achieved as the outcome of nutrient reduction plans against that possibly existing under a 'do nothing' scenario, which may include acceptable deviations from the status prior to eutrophication. Indeed, this is the approach adopted for other environmental problems, such as climate change, where the ambition is to stop further warming of the planet, thereby avoiding the associated impacts, rather than cooling the planet back to the pre-industrial climate. The alternative approach proposed here is, however, precluded by the lack of appropriate frameworks to assess what would have been the trajectories of ecosystems if nutrient inputs had not been reduced, which is the correct, albeit elusive, context in which the outcome of efforts to reduce nutrient inputs should be assessed. The development of such predictive frameworks should have a high priority in the future of eutrophication research. Moreover, the resistance of eutrophied and hypoxic ecosystems to become more oligotrophic upon reductions in nutrient inputs (Conley et al., 2009; Duarte et al., 2009) should alert managers and policy-

makers to the importance of preventing eutrophication because the costs of mitigating eutrophication may be far higher than those of avoidance measures.

Coastal eutrophication as a component of global change

Once again, we find ourselves in need of the macroscopic view when evaluating the research challenges. Effective management requires that we integrate policies across sectors within and beyond the domains of the coastal ecosystems and that we move away from oversimplified views on the controls of eutrophication to consider the operation of factors beyond nutrient inputs, including overfishing, the occurrence of invasive species and climate change, among others. Most important, a look through the global microscope readily reveals that coastal eutrophication is not a process affecting individual ecosystems, but is a global phenomenon both in its global spread and in the relative synchrony of this spread. The forces that shape coastal eutrophication include human population growth, changes in land use, massive production of N in fertilizers through the Haber reaction, increased anthropogenic emissions of reactive nitrogen species to the atmosphere, increased atmospheric CO₂, climate change and overfishing (Duarte et al., 2009; Nixon, 2009), all recognized as forces affecting global change.

Moreover, it may be argued that coastal eutrophication may have the potential to impact the function of the Earth System at the regional and, possibly, global level through changes affecting the trophic dynamics and biogeochemical cycling of the coastal ocean. For example, the increase in hypoxia is clearly a global phenomenon affected by global warming, but also by widespread eutrophication of coastal ecosystems (Díaz & Rosenberg, 2008; Vaquer-Sunyer & Duarte, 2008; Conley et al., 2009). Hence, eutrophication should be considered as a component of global change, in addressing both its causes and its consequences (Duarte et al., 2009), and as such incorporated into major international collaborative programs to address global change. Seeing eutrophication in the macroscopic view of global change is important to better understand and manage the phenomenon (Duarte et al., 2009; Nixon, 2009).

In conclusion, whereas our understanding of coastal eutrophication has progressed in parallel with efforts to

reduce nutrient inputs to coastal ecosystems, major challenges are still ahead of us in the development of a refined understanding that allows prediction of the dynamics of coastal ecosystems subject to simultaneous changes in nutrient inputs and other global forcing factors that lead to a continuous shift in baselines. Most importantly, we must urgently resume a dialog with managers and legislators that more humbly acknowledges the complexities of the problem and the limitations of our capacity to predict the trajectories of managed coastal ecosystems. We must develop a new vision of the effectiveness of efforts to mitigate eutrophication that focuses not only on the changes relative to the initial conditions, but also on the benefits relative to a 'do nothing' policy. Realization of the difficulties of reversing eutrophication, owing to the multiple mechanism that lead to a resistance of coastal ecosystems to reverse the eutrophied state once entered, must also reinforce policies and efforts to avoid eutrophication, the costs of which are likely to be far lower than those of remedial actions.

Acknowledgements This is a contribution to the THRESHOLDS and WISER project, funded by the FP6 and FP7 programs of the EU (contracts # 003933-2 and 226273, respectively). I thank the guest editors of this issue (Jesper H. Andersen and Daniel J. Conley) for the invitation to the conference and to contribute this article.

References

- Andersen, J. H. & D. J. Conley, 2009. Eutrophication in coastal marine ecosystems: Towards better understanding and management strategies. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9758-0.
- Andersen, T., J. Carstensen, E. Hernández-García & C. M. Duarte, 2008. Ecological thresholds and regime shifts: approaches to identification. *Trends in Ecology and the Environment* 24: 49–57.
- Boesch, D. F., 2002. Challenges and opportunities for science in reducing nutrient over-enrichment of coastal ecosystems. *Estuaries* 25: 886–900.
- Breitburg, D. L., J. K. Craig, R. S. Fulford, K. A. Rose, W. R. Boynton, D. C. Brady, B. J. Ciotti, R. J. Diaz, K. D. Friedland, J. D. Hagy III, D. R. Hart, A. H. Hines, E. D. Houde, S. E. Kolesar, S. W. Nixon, J. A. Rice, D. H. Secor & T. E. Targett, 2009. Nutrient enrichment and fisheries exploitation: interactive effects on estuarine living resources and their management. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9762-4.
- Burkart, M. R. & D. E. James, 1999. Agricultural-nitrogen contributions to hypoxia in the Gulf of Mexico. *Journal of Environmental Quality* 28: 850–859.
- Claussen, U., W. Zevenboom, U. Brockmann, D. Topcu & P. Bot, 2009. Assessment of the eutrophication status of transitional, coastal and marine waters within the OSPAR. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9763-3.
- Cloern, J., 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.
- Cloern, J. E., A. D. Jassby, J. K. Thompson & K. A. Hieb, 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. *Proceedings of the National Academy of Sciences* 104: 18561–18565.
- Conley, D. J., J. Carstensen, R. Vaquer-Sunyer & C. M. Duarte, 2009. Ecosystem thresholds with hypoxia. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9764-2.
- Díaz, R. J. & R. Rosenberg, 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321: 926–929.
- Duarte, C. M., 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41: 87–112.
- Duarte, C. M., 2002. The future of seagrass meadows. *Environmental Conservation* 29: 192–206.
- Duarte, C. M., D. J. Conley, J. Carstensen & M. Sánchez-Camacho, 2009. Return to Neverland: shifting baselines affect eutrophication restoration targets. *Estuaries and Coasts* 32: 29–36.
- Dulvy, N. K., R. P. Freckleton & N. V. C. Polunin, 2004. Coral reef cascades and the indirect effects of predator removal by exploitation. *Ecology Letters* 7: 410–416.
- Eyre, B. D. & A. J. P. Ferguson, 2009. Denitrification efficiency for defining critical loads of carbon in shallow coastal ecosystems. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9765-1.
- Fulweiler, R. W. & S. W. Nixon, 2009. Responses of benthic-pelagic coupling to climate change in a temperate estuary. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9766-0.
- Heck, K. L. Jr. & J. F. Valentine, 2007. The primacy of top-down effects in shallow benthic ecosystems. *Estuaries and Coasts* 30: 371–381.
- Henriksen, P., 2009. Reference conditions for phytoplankton at Danish Water Framework Directive intercalibration sites. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9767-z.
- Hill, H., E. Nelson, D. Tilman, S. Polasky & D. Tiffany, 2006. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proceedings of the National Academy of Sciences of the United States of America* 103: 11206–11210.
- Howarth, R. W. & R. Marino, 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over 3 decades. *Limnology and Oceanography* 51: 364–376.
- Jaanus, A., K. Toming, S. Hällfors, K. Kaljurand & I. Lips, 2009. Potential phytoplankton indicator species for monitoring Baltic coastal waters in the summer period. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9768-y.
- Jackson, J. B. C., M. X. Kirby, W. H. Berger, et al., 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629–638.
- Kemp, W. M., W. R. Boynton, J. E. Adof, et al., 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303: 1–29.

- Kratzer, S. & P. Tett, 2009. Using bio-optics to investigate the extent of coastal waters: a Swedish case study. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9769-x.
- Lessin, G., V. Ossipova, I. Lips & U. Raudsepp, 2009. Identification of the coastal zone of the central and eastern Gulf of Finland by numerical modeling, measurements, and remote sensing of chlorophyll *a*. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9770-4.
- Lindahl, O. & S. Kollberg, 2009. Can the EU agro-environmental aid program be extended into the coastal zone to combat eutrophication? *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9771-3.
- Maranger, R., N. Caraco, J. Duhamel & M. Amyot, 2008. Nitrogen transfer from sea to land via commercial fisheries. *Nature Geosciences* 1: 111–113.
- Mumby, P. J., C. P. Dahlgren, A. R. Harborne, C. V. Kappel, F. Micheli, D. R. Brumbaugh, K. E. Holmes, J. M. Mendes, K. Broad, J. N. Sanchirico, K. Buch, S. Box, R. W. Stoeffle & A. B. Gill, 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. *Science* 311: 98–101.
- Nixon, S. W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia* 41: 199–219.
- Nixon, S. W., 2009. Eutrophication and the macroscope. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9759-z.
- Nørring, N. P. & E. Jørgensen, 2009. Eutrophication and agriculture in Denmark: 20 years of experience and prospects for the future. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9772-2.
- Olivé, I., M. P. García-Sánchez, F. G. Brun, J. J. Vergara & J. L. Pérez-Lloréns, 2009. Interactions of light and organic matter under contrasting resource simulated environments: the importance of clonal traits in the seagrass *Zostera noltii*. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9773-1.
- Petersen, J. D., N. Rask, H. B. Madsen, O. T. Jørgensen, S. E. Petersen, S. V. Koch Nielsen, C. B. Pedersen & M. H. Jensen, 2009. Odense Pilot River Basin: implementation of the EU Water Framework Directive in a shallow eutrophic estuary (Odense Fjord, Denmark) and its upstream catchment. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9774-0.
- Savchuk, O. P. & F. Wulff, 2009. Long-term modeling of large-scale nutrient cycles in the entire Baltic Sea. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9775-z.
- Savchuk, O. P., T. R. Eremina, A. V. Isaev & I. A. Neelov, 2009. Response of eutrophication in the eastern Gulf of Finland to nutrient load reduction scenarios. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9776-y.
- Smith, V., 2006. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51: 377–384.
- Soetaert, K., & J. J. Middelburg, 2009. Modeling eutrophication and oligotrophication of shallow-water marine systems: the importance of sediments under stratified and well-mixed conditions. *Hydrobiologia* (this issue). doi:10.1007/s10750-009-9777-x.
- Vaquer-Sunyer, R. & C. M. Duarte, 2008. Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of Sciences* 105: 15452–15457.