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Assessment Criteria for Eutrophication Areas -Emphasis German Bight-

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16. Kurzfassung

In diesem Bericht werden Literatur-Ergebnisse und Datenanalysen zusammengefasst, mit dem Ziel, Bewertungskriterien für die Eutrophierung in der Deutschen Bucht zu entwickeln. Diese Untersuchung wurde auf die Arbeiten in OSPAR bezogen, wo eine „Common Procedure“ entwickelt wurde, nach der für das gesamte OSPAR Konventionsgebiet in Bezug auf Eutrophierungskriterien eine Einstufung als ein „problem-freies“ Gebiet, ein „potentielles Problemgebiet“ oder ein „Problemgebiet“ erfolgen wird. In dieser Untersuchung sind die unterschiedlichen Parameter berücksichtigt, die von OSPAR für eine Gesamtbewertung genannt werden. Basierend auf den hohen Nährstoffeinträgen, und hohen Konzentrationen im Winter, sowie auf den Eutrophierungseffekten im Frühjahr und Sommer (z.B. Problemblüten und Sauerstoffmangel im Bodenwasser), wurde die Deutsche Bucht als „Problemgebiet“ bewertet. In einer regionalen Differenzierung wurde diese Bewertung auf den inneren Hauptteil der Bucht eingeschränkt. Allerdings ist zu beachten, dass die Deutsche Bucht für Eutrophierungseffekte ein von Natur aus sensitives Gebiet ist, da lange Aufenthaltszeiten der Wassermassen, die enge Nachbarschaft von Flußfahnen und geschichteten Gebieten, sowie die Anreicherung von partikulärem Material aus benachbarten Gebieten, Eutrophierungseffekte begünstigen. Abnehmende Phosphatkonzentrationen wurden nur während des Winters beobachtet, verbunden mit einem Anstieg der N/P und Si/P Verhältnisse. Die Datenlage ist für eine vollständige und umfassende Bewertung besonders während des Sommers nicht ausreichend, um die dann kurzzeitigen Eutrophierungsereignisse zu erfassen.

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16. Abstract

In this report the results of a literature study and data analysis are compiled, aiming at developing assessment criteria for the eutrophication in the German Bight. The work was related to activities in the framework of the OSPAR Common Procedure through which the whole OSPAR Convention Area will be designated as either Non-Problem, Potential Problem or Problem Area with regard to the eutrophication status. In this study the different parameters of the checklist of OSPAR for a holistic assessment will be considered. Based on the high nutrient discharges and winter concentrations and on the eutrophication effects during spring and summer (e.g. harmful blooms, oxygen depletion in the bottom water), the German Bight is assessed as a Problem Area with respect to eutrophication. A regional differentiation restricts this classification to the main inner part of the German Bight. However, the German Bight is a sensitive area concerning eutrophication effects, due to long residence times, the close vicinity of river plumes and stratified areas, and trapping of particulate material from adjacent areas. Decreasing phosphate concentrations were only observed during winter in the German Bight, coupled with an increase of N/P and Si/P ratios. Data are not sufficient for a complete holistic assessment during summer to detect short time lasting eutrophication events.

17. Keywords: German Bight, Coastal Water, Eutrophication, OSPAR Common Procedure

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Abbreviations

AlgFes	Algen Früherkennungssystem
ARGE	Arbeitsgemeinschaft
AWI	Alfred Wegner Institut für Meeres – und Polarforschung , Bremerhaven
BAH	Biologische Anstalt Helgoland (now combined with AWI)
BFA-Fi	Bundesforschungsanstalt für Fischerei, Hamburg
BfG	Bundesanstalt für Gewässerkunde, Koblenz
BMFT, BMBF	Federal Ministry of Science and Technology
BSH	Bundesanstalt für Seeschiffahrt und Hydrographie
CCW	Continental Coastal Water
CW	Coastal Water
CNSW	Central North Sea Water
COCOA	Continental Coastal Application
COMP	Common Procedure
COMPP	Comprehensive Procedure (OSPAR)
CPR	Continuous Plankton Recorder
CSTT	Comprehensive Studies Task Team
DHI	Deutsche Hydrographisches Institut (now BSH)
DIN	Dissolved inorganic nitrogen
DOC	Dissolved organic carbon
DOD	Deutsches Ozeanographisches Datenzentrum
DON	Dissolved organic nitrogen
DOM	Dissolved organic matter
DOP	Dissolved organic phosphorus
DSP	Diarrhetic Shellfish poisoning
EPA	Environmental Protection Agency
ERSEM	European Regional Seas Ecosystem Model
EUT	OSPAR Ad Hoc Working Group on Eutrophication
HAMSOM	Hamburg Shelf-Ocean Model
FTZ	Forschungs- und Technologiezentrum Westküste, Büsum
IfBMHH	Institut für Biogeochemie und Meereschemie, Hamburg
ICES	International Council for the Exploration of the Sea
ICWS	International Centre of Water Studies
IKSR	Internationale Kommission zum Schutze des Rheins
KUSTOS	"Coastal mass and energy fluxes – the land-sea transition in the south-eastern North Sea" Project
LANU	Landesamt für Natur und Umwelt, Kiel
LV	Light Vessel
MAR-GIS	Marines Geo – Informationssystem (BMBF- Projekt)
MOCADOB	Monte Carlo Dobson Smith radiation model
N	Nitrogen
NAO	North Atlantic Oscillation

NERC	National Environment Research Council, Plymouth
NEUT	Working Group on Nutrients and Eutrophication
NH_3, NH_4	Ammonium
NLÖ	Niedersächsisches Landesamt für Ökologie, Norderney
NLWA	Niedersächsisches Landesamt für Wasser und Abfall, Hildesheim
NOWESP	The North-West European Shelf Programme
NO_2	Nitrite
NO_3	Nitrate
NO_x	Nitrate + Nitrite + NO_2 and intermediates
NPA	Nationalparkamt Töning
NTU	Nephelometric turbidity units
O_2	Oxygen
OSPAR	OSPAR Commission (formerly Oslo and Paris Commissions)
P	Phosphorus
PAR	Photosynthetically Active Radiation
PN	Particulate nitrogen
POC	Particulate organic carbon
POM	Particulate organic matter
PON	Particulate organic nitrogen
PO_4	Phosphate
PP	Particulate phosphorus
PPC	Phytoplankton carbon
PRISMA	Prozesse in Schadstoffkreislauf Meer-Atmosphäre
Q	Freshwater discharge rate (m^3/s)
RPW	River Plume Water
S	Salinity
SeaWifs	Sea-Viewing Wide Field of view sensor
Si	Silicate
SPM	Suspended particulate matter
SST	Sea Surface Temperature
SYKON	Synthesis and new conception of North Sea research (BMBF- Project)
SYNDWAT	Synoptische Wattenmeeruntersuchungen (BMBF- Projekt)
T	Temperature
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TRANSWATT	Transport, Transfer und Transformation von Biomasseelementen in Wattgewässern (BMBF-Projekt)
TUVAS	Transport, Umsatz und Variabilität von Schadstoffen in der Deutschen Bucht
UBA	Umweltbundesamt (Federal Environmental Agency)
WFD	Water Framework Directive
WRRL	Wasser Rahmen Richtlinie
WSW	Wadden Sea Water
ZISCH	Zirkulation und Schadstofftransport in der Nordsee
ZMK	Zentrum für Meeres- und Klimaforschung der Universität Hamburg

Assessment criteria for eutrophication areas
- emphasis German Bight –

ABSTRACT

The German Bight is characterised by mean residence times of 33 days in the inner Bight and of 67 days in the larger German Bight area. The shallow Bight (mostly < 40 m depth) receives large amounts of nutrients from the rivers Elbe, Weser and Ems, from transboundary transports along with the coastal current and from the atmosphere. Nutrient river loads, reached annual means of 280 % for TN and of 450 % for TP of the natural background concentrations in the period 1990 – 1997 . The main discharge occurs during winter / spring, mostly leaving the area before the growing season starts. Especially phosphate discharges were reduced significantly since the early 1990s, causing phosphate decreases in the total German Bight and within all salinity regimes during winter. However, during summer there was no decreasing phosphate trend, and at the Elbe river mouth an increase was detected, probably due to trapping of particulate material.

Stratification during summer enhances primary production in the mixed layer, which is fed by the nutrient discharges of the rivers. High suspended matter concentrations (> 50 mg/L) in the shallowest parts of the inner German Bight cause light limitation for primary production. As a consequence, nutrients transported by the spreading river plumes often reach the mixed layer of deeper (> 20 m), stratified areas where the phytoplankton receives sufficient light. Frequently, 20 µg/L chlorophyll a are reached as peak concentrations which is significantly above reference values (10 µg/L in the Dutch coastal water). However, the chlorophyll gradients are very variable, especially during spring and near the coast.

Following nutrient exhaustion, the phytoplankton will sink down to the bottom layer, reaching concentrations of more than 15 µg/L chlorophyll a at some locations. The long residence times of the bottom water allow long-lasting degradation processes causing extended oxygen depletion (< 2 mg/L O_2), as was observed by chance in 1981, 1982, 1983 1989, 1994. This oxygen depletion may last several weeks and extend over the complete bottom layer beneath the primary densicline, as has been found in July/August 1994. In connection with the oxygen depletion, zoobenthos and fish mortality has been observed.

During winter, high nitrogen discharges cause high N/P ratios (10-50 as means) and N/Si ratios (2-6). During summer, N/P ratios of > 60 in the central German Bight were observed, due to faster phosphate utilisation. N/Si ratios of 4 – 10 during summer indicate longer lasting fixation of Si. These conditions enhance the development of flagellates, of which some toxic species have been detected, such as *Dinophysis acuminata* in the 1980s and in 1998. Also, new harmful species have been detected in 1997 and 1998. Since the late 1970s flagellates were detected as the dominant phytoplankton group near Helgoland, in the centre of the German Bight, however, without clear evidence for an anthropogenic effect.

For a consistent and complete holistic assessment of all parameters listed by OSPAR, the data quality is not sufficient, especially not during summer, when short time eutrophication effects

are assumed to occur. During summer, more high-frequency measurements are needed with sufficient spatial resolution. Measurements of TN and TP have to be included, considering the phase transfer of the elements N and P. To detect changes of the ecosystem by nonanthropogenic eutrophication, comparisons with the trends in the open North Sea should be enabled, including all components to be assessed.

Following the OSPAR criteria, the inner German Bight has to be assessed as a "problem area" concerning nutrient enrichment and eutrophication effects. However, the German Bight has to be addressed as a sensitive area for eutrophication effects by nature due to (i) the close vicinity of still nutrient rich (light inhibition by high natural turbidity) river plumes and thermohaline stabilised mixed layers promoting the formation of phytoplankton blooms in combination with long residence times, (ii) even longer residence times of bottom water during thermohaline stratification, allowing longer lasting trapping and degradation of sedimented organic material coupled with oxygen depletion, and (iii) trapping of particulate material in estuaries and tidal flats by lateral density gradients and asymmetric tides causing also a permanent import from the sea.

1. Problem definition

1.1 Background

The **COM**mon **P**rocedure (COMP) for the Identification of the Eutrophication Status of the Maritime Area of the OSPAR Convention is an integral part of the Strategy to Combat Eutrophication in the marine environment of the north-east Atlantic and comprises a stepwise procedure.

The first step in the OSPAR Common Procedure is the Screening Procedure, the second is the Comprehensive Procedure, both having the purpose of enabling a classification of the marine region in terms of problem areas, potential problem areas and non-problem areas with regard to eutrophication. Germany, as a contracting party to OSPAR, is obligated to report its intended procedure on the basis of the approach to the Working Group on Nutrients and **EU-T**rophication (NEUT), as outlined at annex 1 and 2 of NEUT 98/5/5.

To protect the marine environment, the Second International North Sea Conference in 1987 agreed to reduce anthropogenic nutrient inputs into the North Sea to 50 % of the 1985 level by 1995. This decision was reaffirmed in 1990 by the Third International North Sea conference, and in 1992 the OSPAR Ministerial Meeting endorsed the Action Plan, which aims at the establishment of measures for achieving the 50 % reduction goal. However, the results are not yet satisfactory, as will be described below.

Following the principles of the Comprehensive Procedure a set of assessment criteria should be linked to form a holistic assessment of the eutrophication status of the maritime area (EUT (2) 97/2/Info.1-E). The biological, chemical and physical assessment criteria are divided into five categories:

1. causative factors (e.g. nutrient enrichment)

2. supporting environmental factors (e.g. light availability or stratification or residence time)
3. direct effects of nutrient enrichment (e.g. increased phytoplankton biomass or shifts in species composition)
4. indirect effects (e.g. oxygen deficiencies or changes in benthic community structures) and
5. other possible effects of nutrient enrichment.

The single factors and effects have to undergo a critical scientific evaluation for their applicability as eutrophication assessment criteria for the North Sea. Since information for all these criteria are not available for the different areas of the German territorial waters in a comparable quality, this study will take into account the priorities of effects as well as the availability of information.

The goal of the present study is the compilation of eutrophication assessment criteria, at first for the German territorial waters then, in a second step, more generally applied to the North Sea.

1.2 Definition of the term "eutrophication"

There is a wide variety of definitions for the term eutrophication, but most of them focus on nutrient enrichment and increased plant growth. The most frequent use of the term is the increase in an input of mineral nutrients, primarily nitrogen and phosphorus, to a particular water body.

The meaning of the two Greek roots of the word eutrophication – eu and trope – are a "good nourishment". "Eutrophic" has the meaning of "well supplied with nutrients" as distinguished from "oligotrophic" ("poor in nutrients") and "hypertrophic" ("oversupplied with nutrients") (Gerlach 1990, p. 1). In the field of medicine it means "healthy or adequate nutrition" (Nixon 1995, p. 200). In the latter context it has a clearly positive sense.

Jørgensen and Richardson (1996) followed that "eutrophication can be defined as the process of changing the nutritional status of a given water body by increasing the nutrient resources".
It is important to note that in the definitions given above already the pure enrichment with inorganic nutrient salts may be classified as eutrophication, even if this process is not followed by enhanced formation of organic matter. In contrast to this view, Nixon (1995, p. 201) proposed the following, more general definition: "Eutrophication – an increase in the rate of supply of organic matter to an ecosystem."

Sommer (1998, p. 401) defines the term "eutrophication" through the trophic state and refers to the anthropogenic origin of eutrophication: "Eutrophication is understood as the increase in trophic state of a water through anthropogenic influences." (translation).

Vollenweider et al. (1992, p. 3) suggested a definition which is suitable for fresh and marine waters: "Eutrophication – in its most generic definition that applies to both fresh and marine waters – is the process of enrichment of waters with plant nutrients, primarily nitrogen and phosphorus, that stimulates aquatic primary production and in its most serious manifestations leads to visible algal blooms, algal scum, enhanced benthic algal growth and, at times, to massive growth of submersed and floating macrophytes." In all the definitions mentioned above, the process of nutrient enrichment is not necessarily coupled with undesirable effects due to eutrophication, but Vollenweider‘s suggestion yet alludes to some of the negative outcomes in connection with massive algal production.

Decreasing oxygen levels in the water of deep basins in the Baltic Sea in the 1960s have been related to the increased discharge of untreated organically loaded wastewater (Gerlach 1990, p. 1). Some authors therefore suggest to talk about hypertrophication instead of eutrophication (Buchwald 1990, p. 324; Radach et al. 1990a, p. 48). Nevertheless, the term “eutrophication” has been established to describe the enrichment of waters by nutrients followed by undesirable effects more in the sense of a burden for an ecosystem than as “well supplied with nutrients”.

In the present investigation the term “eutrophication” is used as proposed in the “OSPAR Strategy to Combat Eutrophication” (NEUT 98/5/1, annex 1):
“Eutrophication means the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of the organisms present in the water and to the quality of the water concerned, and therefore refers to the undesirable effects resulting from anthropogenic enrichment by nutrients as described in the Common Procedure.”

The definition proposed in the OSPAR Strategy to Combat Eutrophication includes not only the process of increasing the nutritional status of water but also encompasses the undesirable effects of this enrichment. In this context it is important to stress that the term “anthropogenic enrichment by nutrients” is persistent. Natural eutrophication or anthropogenic nutrient enrichment which are followed by desirable effects on the marine environment, e.g. stabilisation of the food web resulting in a higher amount of commercial fish stocks, are explicitly not taken into consideration, although some of these effects are suggested by a few authors (e.g. Beukema and Cadée 1997; SRU 1980, p. 154).

A drawback of the OSPAR definition is that it does not take into account a direct input of organic matter. In view of the fact that organic material input to a system from outside can result in a serious, although more local deterioration of the ecosystem and of the water quality it is herewith proposed to complete the OSPAR definition correspondingly[1]. In the present study the term eutrophication will merely be used in the sense of this enlarged OSPAR definition. Some definitions of eutrophication are discussed in a document of the European commission (Mac Garvin et al. 2001).

1.3 Eutrophication aspects relevant for the North Sea and the German Bight

Until the 1980s, anthropogenic eutrophication in the sense of an increase in nutrient supply into the waters of the German Bight was not considered as an important threat to the health of

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[1] Enlarged OSPAR definition of eutrophication: “Eutrophication means the enrichment of water by nutrients **and organic material** causing an accelerated growth of **heterotrophic microorganisms**, algae and higher forms of plant life to produce an undesirable disturbance to the balance of the organisms present in the water and to the quality of the water concerned, and therefore refers to the undesirable effects resulting from anthropogenic enrichment by nutrients **and organic material** as described in the Common Procedure.”

the coastal ecosystem of the North Sea (SRU 1980, p. 156). Nevertheless, already in the mid 1970s reduced oxygen concentrations were observed in a small area of the German Bight between the island of Helgoland and the Elbe estuary (see Rachor 1980, 1982). This was close to the sewage sludge dumping area.

Besides many other possible consequences of eutrophication, oxygen deficiency due to the microbial degradation of organic matter and subsequent formation of hydrogen sulphide is considered as the most negative effect of eutrophication processes because of its lethal impact on the bottom fauna. Recurrent oxygen deficiency (< 4 mg O_2/L) below the pycnocline has been observed in large areas of the German Bight since 1981 (DHI 1983; Gerlach 1984; Rachor 1983, 1985; Rachor and Albrecht 1983; Brockmann and Eberlein 1986).

Generally, it has to be considered that effects of discharges of nutrients and organic matter are linked to the phase distribution, mobility and effects of most of the discharged contaminants (Brockmann 1992; Brockmann et al. 1994). For this reason any modification of the eutrophication status must be assessed in relation to the present load of contaminants.

Beside some less obvious effects, especially in fragile ecosystems (Gabric and Bell 1993), a variety of causative factors and effects of eutrophication have been observed in the North Sea. The most important eutrophication aspects relevant for the North Sea and, especially, for the German Bight can be stepwise arranged in a simplified way as outlined below. More detailed interactions of eutrophication effects are given by Zevenboom (1994). In her article, the author defines problem areas and potential problem areas for the entire region of the North Sea, thereby designating the German Bight as a problem area according to the different eutrophication symptoms.

The following changes can occur during events, seasonally, from year to year or as a long term trend. Locally, regionally or in extended areas of the North Sea the ecosystem will be affected reversibly or irreversibly depending on the intensity, extent and duration of the stress and the resilience of the system.

Tab. 1.3.1: Simplified eutrophication cascade
(OSPAR Comprehensive Procedure (COMPP), checklist in EUT (2) 97/2/Info.1 modified)

1. Discharges or imports above the natural background level of
        (i) inorganic nutrients
        (ii) ammonium (also included in (i))
        (iii) organic dissolved and particulate material
2. Concentrations in coastal waters
    2.1 Increase of concentrations of
        (i) nutrients
        (ii) ammonium
        (iii) organic dissolved and particulate material
    2.2 Shift in nutrient ratios
3. Increase of
        (i) primary production

        (ii) ammonium oxidation (nitrification)
        (iii) decomposition processes
4. Changes in phytoplankton biomass
    4.1 more frequent and/or longer lasting blooms
    4.2 changes in species composition
    4.3 decrease in species diversity
    4.4 formation of nuisance blooms
    4.5 shift in annual biomass cycling
5. Disconnection of food webs
6. Increased sedimentation of biomass
7. Increased oxygen consumption
8. Oxygen depletion in bottom water and in sediment surface layers
9. Shift in benthic species composition
10. Dying or emigration of benthic organisms and fish
11. Changes of sediment structure

Most of the eutrophication aspects listed in Tab.1.3.1 have been observed in the German Bight and in the North Sea and will be briefly discussed in chapter 2.

In addition to this simplified cascade of eutrophication causes and effects which is more or less covered by COMPP, process-related aspects have to be considered for a more detailed assessment, such as

- discharge rates of fresh water, nutrients and organic material to defined areas,
- residence times in these areas,
- growth rates of abundant phytoplankton,
- grazing rates of abundant herbivorous and other zooplankton,
- sedimentation rates of particulate organic material,
- accumulation, trapping, conversion and mobilisation of material at morphological sinks, fronts, densiclines, and other interfaces,
- decomposition rates of dissolved and particulate organic material,
- remobilisation rates from the sediment,
- turbulence,
- diffusive and advective long lasting imports and exports of nutrients and organic material (also at low concentrations),
- phasing of processes,
- release rates of gases accumulating organic material at the sea surface.

These processes are mainly controlled by external factors like temperature, light climate, tidal and residual currents, freshwater discharges, wind forcing and precipitation.

Thus, an assessment based exclusively on concentrations or discharge rates can only reveal very rough information on the eutrophication status of a specific area. Moreover, data normally used for comparisons in the North Sea are generally on an annual scale and hence not sufficient to characterise problem or non-problem-areas, because the time scales for most eutrophication events are in the order of weeks. For this reason, at least monthly values and maximum discharges have to be considered.

For a more realistic assessment, worst case scenarios based on the main processes controlling eutrophication events or trends have to be considered.

1.4 Relevance for the area of investigation

Until the 1980s the known anthropogenic eutrophication in the sense of an increase in nutrient supply into the waters of the German Bight was not considered as an important threat to the health of the coastal ecosystem of the North Sea (SRU 1980, p. 156). However, reduced oxygen contents were already detected in a small area of the German Bight between the island of Helgoland and the Elbe estuary in the mid 1970s (see Rachor 1980, 1982). This was close to the sewage sludge dumping area .

Besides many other possible consequences of eutrophication, oxygen deficiency due to the microbial degradation of organic matter and subsequent formation of hydrogen sulphide is considered as the most negative effect of eutrophication processes because of the lethal consequences for the bottom fauna. Unexpected oxygen deficiency (< 4 mg O_2/L) below the pycnocline was observed in large areas of the outer German Bight in 1981 (DHI 1983; Gerlach 1984; Rachor 1983, 1985; Rachor and Albrecht 1983; Brockmann and Eberlein 1986).

1.5 Research projects focussing on eutrophication in the area of investigation

In the aftermath of extended oxygen deficiencies in the German Bight in 1981 several national research programmes were initiated in order to investigate these phenomena.
Within an interdisciplinary project called "Eutrophication of the North Sea and the Baltic" supported by the UBA (Federal Environmental Agency) a compilation and discussion of information concerning eutrophication in these areas was performed in the years from 1984 to 1990. The final report "Nitrogen, Phosphorus, Plankton and Oxygen Deficiency in the German Bight and in Kiel Bay" (Gerlach 1990) was based on results documented in 21 final reports of the single sub-projects. In this report enhanced phytoplankton production was held responsible for the oxygen consumption at the seabed (Gerlach 1990, p. 304).

The investigations of Hickel et al. (1989) on "Oxygen deficiency in the south-eastern North Sea: Sources and biological effects" covered the south-eastern North Sea, comprising the German Bight and the coastal waters off Jutland as far north as the Limfjord (ibid., p. 4) and were supported by the Federal Ministry of Science and Technology (BMFT). During the time of investigation from April 1984 to March 1987, reduced oxygen contents in the bottom water (3-5 mg/L) were observed, however, critical values were not recorded during this period. Consequently, the bottom fauna re-established in those areas where it had died off during the period from 1981 to 1983. This project again showed that the high amount of particulate organic matter in the German Bight could be a major cause of oxygen consumption, and it was assumed that this organic seston was mainly produced by phytoplankton (ibid., p. 67).

In order to get more insight into the processes leading to oxygen depletion in the coastal North Sea and to elucidate the possible effects on the benthic communities, a follow-up investigation called "Oxygen reduction in the bottom water of the German Bight" (original title: Sauerstoff-Zehrung im Bodenwasser der Deutschen Bucht) was initiated by the BMFT (see above) in 1987 (von Westernhagen et al. 1989). As a main result it was concluded that it is not merely the variable organic load, but also temperature that controls the annual cycle of oxygen reduction in the south-eastern North Sea (von Westernhagen et al. 1989, p. 61). However, in the final report of this study it is not stated that the amount of oxygen depletion in the area is exclusively dependent on water temperature as a result of meteorological variability and that the organic matter concentration does not control the extent of oxygen reduction.

Frey and Becker (1987) used historical data from the German Bight to investigate whether extraordinary hydrological stratification during the summertime in the years 1981 to 1983 could have been the cause of the alarming oxygen deficiencies. In contrast to results from the 1960s and 1970s, they found no clear-cut connection between the spatial extent of the oxygen deficiency area and the spatial extent of the thermohaline stratified area nor extraordinarily stable hydrographic conditions. But the oxygen deficiency area was closely related to the spreading of the nutrient-rich Elbe river plume. It seems certain that similar hydrographic conditions are not exceptional and will occur again, which in combination with the high nutrient levels may lead to serious oxygen deficiencies in the German Bight.

Gillbricht (1988) analysed data from the Helgoland Roads time series and concluded that nutrient supply to the German Bight is always high enough to satisfy even the demands of large phytoplankton blooms and that neither nitrogen nor phosphate play a limiting role for phytoplankton production in the area (ibid., p. 435). In the case of the intense *Ceratium* bloom in 1981, the author didn’t find any depletion of the nutrient pool. This was explained by fast nutrient recycling and uptake of nutrient compounds not registered by the routine analyses (ibid., p. 452), such as dissolved organic nutrients.

Contrary to the reporting of Gillbricht, other authors found indications for temporary phosphate limitation during the phytoplankton spring bloom (Bauerfeind et al. 1990) or limitation of the *Phaeocystis* bloom by phosphorus (Veldhuis et al. 1987) or by nitrogen (Lancelot et al. 1986).

The literature study of Nelissen and Stefels (1988) tried to unravel some of the tangled eutrophication problems and covered many aspects of marine eutrophication. Among these are,

(i) the possible progression of seasonal accumulation of organic matter in problem areas;
(ii) the discussion about when and where what nutrients limit primary production if at all;
(iii) the influences upon the North Sea nitrogen budget to be expected from denitrification, and
(iv) the role of the “small food web“ in recycling of nutrients which should be included in further investigations.

Hanslik et al. (1998) examined the seasonal dynamics of bloom-forming and toxic algae at the seaward border of the East Frisian Wadden Sea along the coast of Lower Saxony (Niedersachsen, German Bight) since 1982 and revealed a changed trend of orthophosphate for this area. In contrast to previous years with highest concentrations prevailing in winter, orthophosphate now attains maxima in summer. The seasonal dynamics of phosphate suggest an influence of eutrophication. The same phenomenon of a summer maximum in orthophosphate concentrations has been observed in the southern North Friesian Wadden Sea since 1989 (Hesse et al. 1992). In the Dutch Wadden Sea, summer maxima in orthophosphate have occurred since the 1950s. They have been explained by excess remineralisation of an increasing load of particulate organic matter entering the Wadden Sea from the adjacent coastal water (Postma 1984). In turn, the increase in the formation and subsequent import of particulate organic matter from the coastal water was interpreted as a result of coastal eutrophication.

UBA (see above) supported a project focussing on long-term measurements of nutrients and phytoplankton in the German Bight (Hickel et al. 1997). Hickel et al. evaluated three decades of long-term series in order to look in detail for changes in nutrient concentrations and phytoplankton stocks in the Helgoland region. The authors emphasise the importance of nitrogen supply in comparison with phosphate. Besides an observed trend towards elevated nutrient concentrations in the waters around Helgoland, a shift in nutrient inputs changed the nutrient ratios. The observed changes in phytoplankton stock were due to an increase of nanoflagellates at the end of the 1970s.

Results from investigations supported by the Danish Environmental Protection Agency (Danish EPA) underline that oxygen reduction in Danish marine waters occurring in some years is primarily determined by the actual meteorological conditions, whereas the general increase in the frequency and extent of episodes of oxygen deficit is due to a general increase in organic matter production (Christensen 1998, p. 20).

There are some new projects related to eutrophication in the German Bight, especially in connection with the Water Framework Directive. Time series of nutrients from the North Sea have been compiled by NOWESP (Laane et al. 1996) and in a nutrient atlas (Brockmann and Topcu 2001). Many other compilations have been involved in the SYCON publication series (e.g. Damm et al. 2001).

2. German Bight

The present study evaluates phenomena related to the anthropogenic eutrophication process in German territorial waters with emphasis on the German Bight. The geographical extension of the area under investigation reaches from 53° to 56° North and 03° to 09° East in order to cover the entire German territorial waters. Although mapped, the German Wadden Sea will not be considered in the present study. This area is covered by the trilateral project “Development of Wadden Sea Specific Eutrophication Criteria”, which was jointly initiated by the Wadden Sea states Denmark, Germany and the Netherlands. The final report “Wadden Sea Specific Eutrophication Criteria” (van Beusekom et al. 2001) is available as publication of the common Wadden Sea Secretariat (Wadden Sea Ecosystem No. 14 – 2001).

2.1 Stratification and residence times

The description of the hydrodynamic situation in the German Bight is presented primarily in terms of supporting factors for eutrophication. That means that the physical characteristics, e.g. of stratification or flushing times, are discussed mainly in the view of their possible influence on the primary production or the sinking of organic material.

2.1.1. Temperature and Salinity

These two parameters are discussed based on the simulation results of the HAMSOM model (Pohlmann 1996; Pohlmann and Puls 1994) covering 14 years in total (82 – 95). The focus of this assessment is not on the description of the temperature and salinity distribution but on the presentation of the vertical gradients between surface and bottom water, resulting in separation of the two water masses by thermo-haline stratification.

In order to represent the seasonal dynamics within the German Bight, the following figures show mean distributions for the months a) January and February, b) March and April, c) July and August and d) September and October. In Fig. 2.1.1.1, 2 the salinity gradient between surface and bottom water is shown as mean over 14 years of simulations. Negative values indicate areas with surface water of lower salinity compared to the bottom water. The strongest salinity gradients, also in combination with the strongest horizontal extension, covering almost the entire German Bight region, can be observed in the summer months July and August. These gradients increase steadily starting from the beginning of the year and cause an estuarine like circulation, accumulating particulate material, also from the sea. The close vicinity between river plumes and stratified areas allows a continuous primary production during summer.

While there is a low salinity gradient of 0.1 near the coast in January and February, these salinity gradients reach a wider horizontal coverage in the following months March and April. This extension covers the region of the Elbe glacial valley but reaches maximum values of 0.2 only near the coast. In July and August, maximum values of 0.4 and 0.5 are reached. However, towards the coast the salinity gradient declines towards lower values. In the same area of the Elbe glacial valley, these values of about 0.2 remain stable throughout the months September and October.

In contrast to the salinity distribution the temperature gradient is oriented rather parallel to the coast (Fig. 2.1.1.3, 4). The strongest gradients can also be observed during the summer months July and August. Positive values represent sites with warmer surface water and colder bottom water. During the other months, there are almost no regions with stronger temperature gradients. In September and October a maximum gradient of 0.4 °C was observed in the northern area of the German Bight. This value corresponds to the lowest value during July and August, found near the coast, increasing at that time to a maximum of 6.8 °C in the open water of the German Bight.

When comparing the distribution of the maximal density gradient (Fig. 2.1.1.5, 6) with those for the temperature and salinity gradients, the structure of the salinity gradient is found to be dominating the maximum density gradient. This holds especially for spring (March and April) and autumn (September and October), mainly because the distribution of the temperature gradient (Fig. 2.1.1.5, 6) shows no clear structure. In accordance with the distribution of the salinity gradients (Fig. 2.1.1.1, 2) and the temperature gradients (Fig. 2.1.1.3, 4), the highest values and the widest horizontal extension for the maximum density gradient can be observed in summer during the months July and August. However, for this part of the year the influence of the temperature gradient seems to be dominant since the maximum density gradient reveals a structure which is oriented rather parallel to the coast in comparison to the salinity gradient. In the vicinity of the mouths of the rivers Elbe and Weser, the density contour lines are rather oriented towards the distribution of the salinity gradient, but further offshore the distribution of the maximum density gradient is again mainly influenced by the temperature gradient.

This change in the influence on the maximum density gradient (kg/m^4) can clearly be seen when looking at the contributions of the temperature and salinity gradients to the development of the density gradient (Fig. 2.1.1.7 -10). The vertical density gradient has the unit kg/m^4 by combining the density (kg/m^3) with the water depth (m^{-1}). The contribution of the temperature to the density gradient is nearly zero in all months except for July and August, when the mean values express significant contributions. During these months, the temperature related influence on the density gradient causes values near the coast of 0.01 kg/m^4 near the coast, while offshore maximum values greater 0.05 kg/m^4 are reached. For a wide area, values of about 0.03 kg/m^4 can be observed. For the same months, the contribution of the salinity to the density gradient yields values of 0.01 and 0.02 kg/m^4, but these values are restricted to a small region within the German Bight. This shows that in the months July and August for a wide region of the German Bight the temperature is the main trigger for the development of a density gradient and as a consequence for a separation of surface and bottom water. During all other months the salinity dominates this process, but at a lower level (between 0.01 and 0.02 kg/m^4). Therefore, there is not only a change in the dynamics of the triggering of the density gradient from salinity to temperature, but it can be shown that the influence of the temperature on the density during this time is stronger than any other process during the whole year.

When looking at the depth of the maximum density gradient (Fig. 2.1.1.11, 12), a decrease of this depth can be observed, starting at the beginning of the year and reaching a maximum during the months July and August. Afterwards, the maximum density gradient is again getting into deeper layers. At the beginning of the year, only a small-scale structure can be observed, mainly because of the existing small density gradient. Towards the summer months July and August, this is transformed into a structure where the contour lines are oriented parallel to the coast. This goes along with a decrease of the depth of the maximum density gradient from 35 m on a small spatial scale in January and February to a depth of about 20 m covering a wide area during July and August. In autumn, depths of 30 m are found for the maximum density gradient in offshore areas.

A similar depth decrease in summer (July and August) is observed for the mixed layer, which is calculated based on the minimum of the vertical exchange coefficient A_v from hydrodynamic model simulations (Fig. 2.1.1.13, 14). In winter (January and February), the mixed layer extends to the bottom, then slowly rises to a mean depth of 20 m in March and April in the offshore areas within the German Bight. The minimum of about 13 m is reached during the summer months July and August for the entire German Bight. In September and October the mixed layer again reaches greater depths because of the increased storm activity, finally leading to the winter distribution in January and February. It should be noted that the depth of the mixed layer shows the same structure as the depth of the maximum density gradient for the months March and April as well as September and October. In the summer months, mainly because of the missing gradient in the vertical exchange coefficient A_v, a horizontal structure was not observed. Therefore, the absence of the vertical exchange in the summer months is responsible for the dominance of the temperature in the influence on the maximum density gradient.

2.1.2. Flushing Times

The flushing time is calculated as the volume of an area (box) divided by the total gross inflow into this area. The interpretation of the flushing times is simply the time needed to fill the given area by the inflowing water, expressed in days. Another way of looking at the exchange for a certain area is given by the residence time. The calculation is based on tracer studies, with a statistical representation of how long an ensemble of tracers released within a certain area needs to leave the given area. Usually, the residence times provide higher estimates compared to the flushing times for the same area of interest, since single tracers are capable of circulating within the area for a longer time span, therefore leading to a higher mean residence time of the whole ensemble released. In contrast, the flushing times represent an integrated approach for the total transport through the area.

Literature values on the flushing times for the German Bight can be found within the framework of the division of the North Sea into the ICES-boxes (Lenhart and Pohlmann, 1997; OSPAR Commission, 2000). For a volume of the German Bight of 602 km^3 (Box 5A), a mean flushing time of 33 days was calculated. For this value, a range between a minimum flushing time of 10 days and a maximum flushing time of 56 days is given. In comparison, for the Danish coast a mean value of 11 days represents a much higher flushing of this area. Even for the Dutch coast with a mean flushing time of 28 days the exchange of the water mass is higher. Since the German Bight represents an area of lower exchange, this results in a longer residence time for material that is introduced, e.g. by the River Elbe, as well as a higher possibility of material which has been transported into the German Bight region of sinking and accumulating on the bottom.

Values for the flushing times for the German Bight can be found in the literature already within the framework of the division of the North Sea into the ICES-boxes (Lenhart and Pohlmann, 1997; OSPAR Commission, 2000). For a volume of the German Bight of 602 km^3 (Box 5A) a mean flushing time of 33 days was calculated. For this value a range between a minimum flushing time of 10 days and a maximum flushing time of 56 days is given. In comparison, for the Danish coast a mean value of 11 days represents a much higher flushing rate. Even for the Dutch coast, with a mean flushing time of 28 days, the exchange of the water mass is higher. Since the German Bight represents an area of lower exchange, this results in a longer residence time for material that is introduced, e.g. by the River Elbe, as well as a higher possibility for material which has been transported into the German Bight region to sink and accumulate at densiclines or at the bottom.

In order to get a regionally differentiated picture, 6 subregions were defined for the German Bight for which additional flushing time calculations were carried out (Tab. 2.1.2.1). The boxes were taken from the ecosystem model ERSEM in the box configuration for the COCOA (Continental Coastal Application) simulations (Lenhart 2001, 1999; Lenhart et al. 1997). The division of the boxes is presented in Fig. 2.1.1.8. One should add that box 7 is defined as the sum of all smaller boxes 1 to 6. The hydrodynamic transport which enters the calculations is based on the data set by Pohlmann (1996) , which also forms the basis for the discussion of the horizontal distribution of the temperature and salinity gradients as outlined above.

Tab. 2.1.2.1: Calculated flushing times for sub regions within the German Bight

(see Fig. 2.1.2.1)

BOX	Vol. (km^3)	Flushing times (days) Min.	Max.	Mean
1 Inner German Bight	58	18	60	33
2 East Frisian Coast	136	5	25	15
3 Outer German Bight	291	5	26	15
4 Southern Danish Coast	117	4	25	12
5 Offshore German Bight	159	2	18	7
6 Middle Danish Coast	246	2	25	9
7 (Total)	246	15	67	38

The values for the flushing times of the subregions within the German Bight are based on 2 years of hydrodynamic transport information (1988 and 1989). There is a clear structure of the flushing times for the different boxes, except for box 7 which covers the complete German Bight area as the sum of all other boxes. The longest flushing time can be found for box 1, which represents the mouth of the River Elbe. For the surrounding boxes 2, 3 and 4 the mean values range between 12 and 15 days. In contrast, the offshore boxes 5 and 6 have mean values for the flushing time of only 7 to 9 days. It can be concluded that these offshore boxes are influenced by the coastal current to a higher degree than the more nearshore boxes. This is especially true for box 1, where the flushing time is twice as long compared to the coastal boxes. Based on these values it can be concluded that a substance which is introduced into box 1 will remain in this box twice as long in relation to the other coastal boxes. For the offshore boxes 5 and 6 the residence time for a substance can be estimated again to be twice as long as for the coastal boxes 2, 3 and 4.

Finally, some information should be given for box 7. With a volume of 1007 km^3 this box covers a larger region than the ICES box 5A which represents the German Bight region within the ICES subdivision of the North Sea (Lenhart and Pohlmann 1997; OSPAR Commission 2000). Therefore the slightly higher flushing time of 38 days is realistic, in comparison to the 33 days given for the ICES box 5A. Small differences can also be related to the data which enter the flushing time calculation. While the calculation for the ICES box 5A could be based on 12 years of daily transport information, the flushing time for box 7 is only based on 2 years of transport data. In comparison to box 1, the mean and the maximum flushing time for box 7 has shifted to higher values. However, the shortest flushing time for the overall box 7 of 15 days is still less than the shortest flushing time in box 1 with 18 days, despite the smaller volume. Finally, it should be pointed out that all flushing times show a clear seasonal signal, with shorter flushing times in winter because of an increased transport during this time of the year.

2.1.3. Light and suspended matter concentrations

In addition to the transport information which formed the basis for the flushing times calculations, the ecosystem model ERSEM (Lenhart 1999, 2001, Lenhart et al. 1997) was used for further parameters like irradiance at the sea surface, underwater light and suspended matter concentration (Fig. 2.1.3.1; see also Fig. 2.3.7.3). The irradiance at the sea surface, which is used within ERSEM as forcing for the primary production, was calculated with the MOCADOB model for the years 1988 and 1989 as examples (Pätsch 1994). In this model, the global radiation is modified by the actual cloud cover. In the time series of the irradiance for the different boxes the seasonal cycle of the irradiance can clearly be seen, which is modified by the cloud cover within a certain range. Because of the low geographic extension of the boxes the time series show individual differences in the local representation of the irradiance under the influence of the cloud cover, but the curves of the time series are very similar for all boxes with a maximum value of about 300 Watt/m^2 in summer.

The corresponding underwater light is mainly influenced by the suspended matter concentration in the water. The information on the suspended matter concentration for the North Sea is derived from the simulations of a suspended matter model by Puls (Puls et al. 1997), which were aggregated for the ERSEM boxes. There is a general similarity between the boxes for the suspended matter concentration and the underwater light, which could already be seen for the flushing times. The inner German Bight box 1 shows by far the highest suspended matter concentration. The time series of the suspended matter concentration displays a strong seasonal pattern with high values in winter, which can be related to a higher storm activity in winter, resulting in a higher vertical mixing and resuspension of sediment particles. The related values for the underwater light are extremely low, reaching at most about 15 Watt/m^2 in summer (see also chapter 2.3.7). In all other boxes suspended matter concentration is much lower, also revealing a strong seasonal cycle with the highest values in winter. In box 5, the box furthest from the coast, these winter maxima can be observed as strong peaks in the time series, while in the other boxes there are generally higher concentrations for suspended matter during the winter months. Because of the low suspended matter concentration, the coastal boxes 3 and 4 reach values of 100 Watt/m^2 in summer. For the offshore boxes 5 and 6 the summer values are even higher with up to 150 Watt/m^2. The summer maximum for the overall box 7 gives an integrated value for the all boxes of 100 Watt/m^2 for the underwater light.

Taking also into account the information from the flushing times (Tab. 2.1.2.1) it can be concluded that nutrients which are introduced into box 1 will mostly not be transferred into biological production here. In contrast, in the coastal boxes with mean flushing times of 12 and 15 days and an underwater light of about 100 to 150 Watt/m^2 the chances are much better that nutrients which have been introduced into these boxes are utilised for biological production. On the other side, the potential risk of dead organic material sinking to the bottom which, in turn, can lead to problems with the oxygen availability because of the remineralisation processes, is much higher in box 1. In the offshore boxes 5 and 6 this risk is reduced by the flushing times, allowing only for about half the residence time within this subregion of the German Bight compared to the coastal boxes 2 to 4.

2.2 Input of nutrients and organic matter

A sea area can receive external nutrient supplies (including nutrient elements fixed in organic material) by advection (adjacent areas), from land runoff (via rivers, coastal runoff and direct discharge) and from the atmosphere (Radach et al. 1990 b). Internal sources are mineralisation in the water column and remineralisation from the sediment, which is temporarily a source and/or a sink. The relation between man-made nutrient inputs and concentrations in the sea, however, is far from simple (Nelissen and Stefels 1988).

The four major rivers directly influencing the German Bight are the Elbe, the Weser, the Ems and the Eider. According to the ratios between the discharge volumes of about 20:9:2.5:1, the riverine inputs are mainly compiled for the Elbe and the Weser. However, a complete compilation was recently published (Lenhart and Pätsch 2001).

The 50 % reduction goal with respect to nutrient inputs, agreed by the North Sea Conferences in London (1987) and in The Hague (1990), was successful for phosphorus until now, but not for nitrogen. Therefore, increased N/P ratios and related shifts in the species composition of algae can cause further problems.

The more or less permanent import from the southern North Sea at a low concentration level but in large volumes in comparison to the river discharges will also contribute to the eutrophication in this sensitive area. By the import of dissolved and particulate organic material seasonal deposition will be increased. This material, decomposed during longer time periods, can quickly be remobilised in the shallow bight and contribute to eutrophication events.

Discharges or imports have to be assessed in relation to natural background levels. Several attempts have been made to define natural background levels of nutrients for river discharges and for the coastal water (ARGE Elbe 1995; Laane 1992). Similar data are used as references within OSPAR in deriving definitions of elevated concentrations (Zevenboom 1994).

Elbe river

Discharge calculations differ significantly due to the method and data base used. Examples are given for the Elbe (Tab. 2.2.1). Differences reach up to 20 % for total N and surpass 50 % for total P. There is a need to evaluate calculation methods taking into account the different estuarine processes.

These differences are also involved in international comparisons of discharge rates of different rivers (Tab. 2.2.2). Besides data handling and analyses, also sampling, sample processing and analysis have to be harmonised.

Tab. 2.2.1: Annual loads of N and P of the River Elbe, calculated by different methods from 1984 to 2001

Year	Parameter	ICWS, 1990 1) kt/a	Lenhart and Pätsch 2001 2) kt/a	ARGE Elbe 3) kt/a	QSR, 1993 kt/a	ARGE Elbe: input to the North Sea, kt/a	Calculations Teufelsbrück 4) kt/a	Calculations Seemannshöft 5) kt/a	Calculations Cuxhaven 6) kt/a
1984	Total N	151.2	138.3	150.0			147.8		
1985	Total N	128.6	140.3	150.0		150.0	150.3		
1986	Total N	212.8	168.6	200.0		200.0	168.1		
1987	Total N	306.4	255.2	280.0		280.0	282.3	275.3	
1988	Total N		195.3					212.1	
1989	Total N		119.6					130.4	
1990	Total N		97.7		120.0	130.0		106.5	
1991	Total N		105.8					75.9	
1992	Total N		72.0			130.0		111.3	87.4
1993	Total N		94.3					98.1	92.4
1994	Total N		175.2					185.5	214.1
1995	Total N		165.3					175.0	181.8
1996	Total N		121.5					129.3	136.0
1997	Total N		109.3					114.0	121.9
1998	Total N		110.0					112.5	113.8
1999	Total N		116.7					114.1	115.9
2000	Total N		103.5					115.7	108.9
2001	Total N							118.2	99.0
1984	Total P	9.5	8.8	12.0			9.4		
1985	Total P	8.9	8.9	9.0		11.0	9.5		
1986	Total P	12.3	8.6	10.0		10.0	8.8		
1987	Total P	12.8	9.2	11.0		11.0	10.0	10.5	
1988	Total P		8.2					9.9	
1989	Total P		5.7					6.8	
1990	Total P	4.1	4.5		7.9	8.0		5.1	
1991	Total P		3.3					3.5	
1992	Total P	4.7	4.7			7.0		4.7	3.2
1993	Total P	4.1	4.1			11.0		4.4	4.4
1994	Total P		6.1			7.0		6.1	9.3
1995	Total P		6.2					5.7	7.9
1996	Total P		4.8					4.6	5.2
1997	Total P		4.2					3.8	5.0
1998	Total P		4.6					4.2	6.9
1999	Total P		4.6					4.2	7.0
2000	Total P		4.2					4.2	5.0
2001	Total P							4.3	3.6

1) measurements of the ARGE Elbe station Glückstadt – km 675.5 - (weekly mixed samples), Q from Neu Darchau (multiplied with a factor 1.109); mean flows differ from those which are presented in the Wassergütedaten of the ARGE Elbe. This is caused by the fact that this investigation only takes into account flows of the days when samples were taken and not the daily flows. So the flows which are used in this investigation are, with the exception of 1985, higher than the flow used in the Wassergütedaten (the differences ranges from –9 % to +18 %)

2) based on measurements (weekly) of the Elbe stations Teufelsbrück (until 1991) – km 630.1 - and Seemannshöft – km 628.9 -, silicate measurements from the station Grauerort (from 1988) – km 660.5 -, calculated without a factor for the difference between Neu Darchau and the measurement-station

3) calculations for the years 1985 and 1986 based on only 11 to 12 measurements and for 1987 on 8 measurements at the Elbe station Glückstadt

4) calculations based on measurements of the Elbe station Teufelsbrück (weekly samples); the nutrient concentration was multiplied with Q (Neu Darchau) and a given factor from the ARGE Elbe (1.07 for Teufelsbrück) to take into account the additional drainage from the area downstream of Neu Darchau

5) calculations based on measurements of the Elbe station Seemannshöft (weekly samples); the nutrient concentration was multiplied with Q (Neu Darchau) and a given factor from the ARGE Elbe (1.077 for Seemannshöft) to take into account the additional drainage from the area downstream of Neu Darchau

6) calculations based on measurements of the Elbe station Cuxhaven (weekly samples); the nutrient concentration was multiplied with Q (Neu Darchau) and a given factor from the ARGE Elbe (1.21 for Cuxhaven) to take into account the additional drainage from the area downstream of Neu Darchau

Tab. 2.2.2: Annual riverine inputs of nutrients, total N and P to the German Bight, compilation of data from different sources for 1985 and 1990

Parameter	River	Time	Load [kt/a]	Reference
NH_4^+	Elbe	1985	27.1 kt N/a	ICWS, 1990
NH_4^+	Elbe	1985	31.5 kt N/a	Wulffraat et al.1), 1993
NH_4^+	Elbe	1990	13.0 kt N/a	Wulffraat et al., 1993
NH_4^+	Elbe	1985	49.7 kt N/a	Lenhart et al., 1996
NH_4^+	Elbe	1990	18.9 kt N/a	Lenhart et al., 1996
NH_4^+	Weser	1985	1.1 kt N/a	ICWS, 1990
NH_4^+	Weser	1985	1.1 kt N/a	Wulffraat et al., 1993
NH_4^+	Weser	1990	2.2 kt N/a	Wulffraat et al., 1993
NH_4^+	Ems	1985	2.4 kt N/a	ICWS, 1990
NH_4^+	Ems	1985	6.5 kt N/a	Wulffraat et al., 1993
NH_4^+	Ems	1990	1.3 kt N/a	Wulffraat et al., 1993
NO_3^-	Elbe	1985	78.3 kt N/a	ICWS, 1990
NO_3^-	Elbe	1985	92.0 kt N/a	Wulffraat et al., 1993
NO_3^-	Elbe	1990	58.0 kt N/a	Wulffraat et al., 1993
NO_3^-	Elbe	1985	64.0 kt N/a	Lenhart et al., 1996
NO_3^-	Elbe	1990	65.8 kt N/a	Lenhart et al., 1996
NO_3^-	Weser	1985	49.2 kt N/a	ICWS, 1990
NO_3^-	Weser	1985	47.5 kt N/a	Wulffraat et al., 1993
NO_3^-	Weser	1990	51.5 kt N/a	Wulffraat et al., 1993
NO_3^-	Ems	1985	17.8 kt N/a	ICWS, 1990
NO_3^-	Ems	1985	27.0 kt N/a	Wulffraat et al., 1993

Parameter	River	Time	Load [kt/a]	Reference
NO_3^-	Ems	1990	22.0 kt N/a	Wulffraat et al., 1993
NO_3^-		1990	140 kt N/a by Ger-	QSR, 1993
Total N	Elbe	1985	155.4 kt N/a	de Vries et al.2), 1990
Total N	Elbe	1985	128.6 kt N/a	ICWS, 1990
Total N	Elbe	1985	140.3 kt N/a	Lenhart et al., 1996
Total N	Elbe	1990	97.4 kt N/a	Lenhart et al., 1996
Total N	Elbe	1990	120.0 kt N/a	QSR subregion 5, 1993
Total N	Weser	1985	70.0 kt N/a	de Vries et al., 1990
Total N	Weser	1985	50.3 kt N/a	ICWS, 1990
Total N	Weser	1990	45.0 kt N/a	QSR subregion 5, 1993
Total N	Ems/Dollard	1985	37.9 kt N/a	de Vries et al., 1990
Total N	Ems	1985	26.6 kt N/a	ICWS, 1990
Total N	Ems	1990	17.0 kt N/a	QSR subregion 5, 1993
Total N		1985	740 kt/a to the North	ICWS
Total N		1990	190 kt by Germany;	QSR, 1993
PO_4^-	Elbe	1985	3.7 kt P/a	ICWS, 1990
PO_4^-	Elbe	1985	3.8 kt P/a	Wulffraat et al., 1993
PO_4^-	Elbe	1990	1.5 kt N/a	Wulffraat et al., 1993
PO_4^-	Elbe	1985	4.3 kt P/a	Lenhart et al., 1996
PO_4^-	Elbe	1990	2.2 kt P/a	Lenhart et al., 1996
PO_4^-	Weser	1985	4.0 kt P/a	Wulffraat et al., 1993
PO_4^-	Weser	1990	1.1 kt P/a	Wulffraat et al., 1993
PO_4^-	Ems	1985	0.40 kt P/a	ICWS, 1990
PO_4^-	Ems	1985	0.95 kt P/a	Wulffraat et al., 1993
PO_4^-	Ems	1990	0.25 kt P/a	Wulffraat et al., 1993
PO_4^-		1990	3.8 kt P/a by Germany,	QSR, 1993
Total P	Elbe	1985	11.1 kt P/a	de Vries et al., 1990
Total P	Elbe	1985	8.9 kt p/a	ICWS, 1990
Total P	Elbe	1985	10.0 kt P/a	Wulffraat et al., 1993
Total P	Elbe	1990	6.0 kt P/a	Wulffraat et al., 1993
Total P	Elbe	1985	8.9 kt P/a	Lenhart et al., 1996
Total P	Elbe	1990	4.1 kt P/a	Lenhart et al., 1996
Total P	Elbe	1990	7.9 kt P/a	QSR subregion 5, 1993
Total P	Weser	1985	5.0 kt P/a	de Vries et al., 1990
Total P	Weser	1985	3.6 kt P/a	ICWS, 1990
Total P	Weser	1985	4.9 kt P/a	Wulffraat et al., 1993
Total P	Weser	1990	4.7 kt P/a	Wulffraat et al., 1993
Total P	Weser	1990	2.4 kt P/a	QSR subregion 5, 1993
Total P	Ems/Dollard	1985	9.1 kt P/a	de Vries et al., 1990
Total P	Ems	1985	0.89 kt P/a	ICWS, 1990
Total P	Ems	1985	1.8 kt P/a	Wulffraat et al., 1993
Total P	Ems	1990	1.1 kt P/a	Wulffraat et al., 1993
Total P	Ems	1990	0.4 kt P/a	QSR subregion 5, 1993
Total P		1990	11 kt by Germany;	QSR, 1993
Total Si	Elbe	1985	55.2 kt/a	de Vries et al., 1990

Parameter	River	Time	Load [kt/a]	Reference
Si	Elbe	1990	57.0 kt/a	Lenhart et al., 1996
Total Si	Weser	1985	27.3 kt/a	de Vries et al., 1990
Total Si	Ems/Dollard	1985	11.4 kt/a	de Vries et al., 1990

1) mean values from minimum and maximum loads
2) based on PARCOM-nut inquiry (1989)

Nitrogen compounds
In the River Elbe there was an increase of nitrate loads calculated for Teufelsbrück/Seemannshöft at km 630.1/628.8 until 1989 (see Fig. 2.2.1 for the different Elbe stations), followed by a decrease (Fig. 2.2.2). This decrease was much more significant for ammonium (Fig. 2.2.3). However, as shown for loads calculated at Cuxhaven (km 725.2), the discharges were controlled by the freshwater discharge (Figs. 2.2.4 - 2.2.6, 2.2.8) and any trend calculation has to be normalised concerning the variable flow rates.

Looking at the concentrations and neglecting the dependence of nutrient concentrations on freshwater discharge rates (ARGE), at Cuxhaven a slight decrease of nitrate can be assumed, starting in 1992 (Fig. 2.2.6). The natural background concentrations are assumed to be 1 mg/L (ARGE Elbe 1991). This means that due to the anthropogenic input this value will be reached only during summer. On the other hand, an increase of ammonium was estimated until the end of 1999. Unfortunately, the analytical method was changed at that time, causing a significant decrease (Fig. 2.2.7). However, correlations with salinity (not presented) reveal for both methods and periods an increase of ammonium with increasing salinity. The natural background concentrations of 0.1 mg/L were strongly surpassed in most cases and only after the changing of the analytical method were values in the range of natural background concentrations.

The ammonium increase probably reflects an increasing turnover occurring in the estuary, caused by decreasing toxic compounds and remobilisation from adjacent sediments which accumulate particulate organic matter. These processes have to be considered for a selection of data for flux calculations.

Discharges of ammonium should be distinguished from general nutrient discharges due to the fact that significant amounts of oxygen will be consumed by nitrification of ammonium, which has caused severe oxygen depletion in the Elbe estuary downstream of Hamburg. Recently, ammonium loads in the Elbe have been reduced due to changes in the use of agricultural fertilisers and improved waste water cleaning (Fig. 2.2.3). In spite of reduced nutrient concentrations in the River Elbe, an increased algal growth was monitored in summer during the 1990’s. Although the high nutrient content in the 1970’s and 1980’s should have resulted in algal mass occurrences in the Elbe, this was not the case, since this was prevented by inhibition through toxic substances. These toxic substances also inhibited bacteria for ammonium decomposition. Therefore, the increase of the algal population in the river is not nutrient-related but related to the decrease of harmful substances.

Due to temperature limitation with respect to the process of nitrification, ammonium river discharges are higher during winter. The imported load will than be oxidised mainly in the coastal water.

Phosphorus compounds

The load of total phosphorus at Cuxhaven was mainly controlled by the freshwater discharge, varying between 5 and 70 t P/d (Fig. 2.2.8).

Similar to ammonium, phosphate concentrations at Cuxhaven showed an increasing trend from 1992 to the end of 1999 from 0.11 mg/L to about 0.17 mg/L (Fig. 2.2.9). After this period, estimated concentrations dropped to about 0.1 mg/L, due to changed analytical method. The same holds for total phosphorus, which increased from 0.18 mg/L to 0.28 mg/L in 1999, followed by concentrations of 0.17 mg/L (Fig. 2.2.10). As for ammonium, for both phosphate data sets of an increase with increasing salinity was also observed (not shown). This indicates remobilisation from trapped particulate material in the lower estuary of the Elbe.

In spite of a significant phosphorus reduction during the last decade, concentrations were far above the natural background concentrations of phosphate (0.02 mg P/L) and total phosphorus (0.05 mg P/L), which were assumed for freshwater. Due to the fact that nutrient concentrations in the North Sea normally decrease from the estuaries towards the sea, it could be assumed that in the lower estuaries, besides the accumulation of matter in the maximum turbidity zones, the accumulation of particulate material in the adjacent Wadden Sea will also affect concentration gradients here.

Organic matter

Organic compounds discharged by rivers or precipitated from the atmosphere are often neglected in the eutrophication discussion, mainly due to the lack of data. However, these substances contribute significantly to the oxygen consumption in coastal waters.

Since particulate material will be trapped by the maximum turbidity zones, which is mostly located at Brunsbüttel (km 693) for the Elbe river, data from this location cannot be used for calculations of discharges of matter to the sea. High discharges will shift this zone downstream, affecting also measurements at Cuxhaven (Fig. 2.2.11). For example, there were extraordinarily high floods in winter/spring 1994, 1999 and 2000 (Fig. 2.2.4), shifting the maximum turbidity zone downstream and causing an increase of POC concentrations at Cuxhaven.

Again, differences are evident depending on the methods of estimation. Concentrations of POC are either calculated as difference between TOC and DOC or measured (since 1994) directly. However, by both methods a permanent decrease in the discharges of POC was measured at Cuxhaven in 1998, from 500 t/d in 1994 (at high freshwater discharges, see Fig. 2.2.4), down to 100 t/d in 1998.

The dominant fraction of organic material discharged to the sea is DOC (Fig. 2.2.12). This phase has been assumed to be refractory and to behave conservatively. However, recent research considering the different size classes of dissolved organic compounds revealed that significant conversion already starts in the estuary (Volk et al. 1997). For this reason the discharge of organic material to the coastal water has to be considered when assessing the eutrophication situation.

Atmospheric deposition

Concerning atmospheric deposition, time series for a coastal station in southern Jutland (Gerlach 1990) showed a threefold increase in the deposition of ammonium and nitrate between 1955 and 1979. Calculations of Bennekom and Wetsteijn (1990) have shown that in the southern North Sea atmospheric deposition of nitrogen compounds is, per annum, less than 10% of the river loads. For the coast of Lower Saxony and the area of the North Frisian Wadden Sea, estimates are much higher (Schulz pers. comm.), reaching calculated values of 50 % (Beddig et al. 1997). Thus the proportion of atmospheric to riverine input strongly depends on the geographical position of the investigated area.

However, during nitrogenlimited conditions, atmospheric deposition can significantly contribute to the nutrient load of the North Sea. The deposition of ammonium and NO_x into the North Sea was calculated by models and reached for the total North Sea 200 000 t/a for ammonium and 400 000 t/a for NOx compounds (Krell and Schlünzen 2001, p. 47). Asman et al. (1994) calculated a deposition of 7 600 t/a for the southern North Sea (south of 56° N) . Rendell et al. (1993) reported a calculated deposition rate of total nitrogen of 228 000 t/a for the southern North Sea and 412 000 t/a for the total North Sea.

In addition to the NO_x and NH_3 deposition, atmospheric input of organic material such as DON may also be relevant for nutrient budget calculations, because significant portions of these compounds are rapidly taken up by the plankton (Peierls et al. 1997).

Atmospheric deposition of nutrients was studied explicitly for the German Bight area, revealing highly variable deposition rates, depending on the direction of the wind trajectories (Schwikowski et al. 1988). Detailed studies in the German Bight were performed by Schulz et al. (1999), showinging weekly deposition rates of 2 – 4 mg/m^2d nitrate and ammonium each at selected stations.

2.3 Increase of nutrients and organic matter in the coastal water

There have been several studies in the North Sea, analysing the long term trends of nutrients on the basis of mixing diagrams in the coastal water (ICES 1995; Bennekom and Otto 1989; Hydes and Edmunds 1992). Data from selected transects reveal the variability due to sampling in different water masses and seasonal variations. However, for the continental coastal water, an increase of nitrate was reported between 1980 and 1990 (QSR 1993; ICES 1995).

The continental coastal water is characterised by high loads of nutrients during winter, caused by increased discharges during this season and by remineralisation, which is not competing with the light limited primary production in this turbid area (Brockmann et al. 1990).

Background data for nutrients in the German Bight during winter are defined as 0.43 - 0.60 µM phosphate, 9 - 17 µM silicate and 0.4 - 1.2 µM nitrite (Laane 1992). No data are given for nitrate due to lack of historical data.

In the German Bight, trend investigations have been performed between 1978 and 1990, using extended station grids during winter (Körner and Weichart 1991). These results did not show any significant trends. However, comparing the data from 1978 (Weichart 1985) with measurements from 1936 (Kalle 1937), a significant phosphate increase was found in the coastal water of the German Bight.

Using the data from Helgoland roads, collected since 1962, several trend analyses have been performed (Radach and Berg 1986; Radach et al. 1990a; Hickel et al. 1993; Gerlach 1990). It was shown that between 1962 and 1984 there was an increase of all nutrients except silicate, which was reported to decrease (Radach et al. 1990a). However, in 1985 and 1986 there were data inconsistencies, due to methodological problems, resulting in low silicate values. Since 1982 a phosphate decrease was observed (Hickel et al. 1993). Nitrate increase as a sudden event in the late seventies (Hickel et al. 1997) was observed from 1980/81 until 1987; a successive decrease occurred after 1991 and an increase again in 1993/94. A delay in nitrate increase of three years in relation to coastal waters is interpreted by remineralisation of particulate material transported over a long distance. The presented silicate concentrations decreased from 1980/81 to 1986 and then increased to absolute maxima. Since these data have not been intercalibrated with independent measurements it must be assumed that inconsistencies have not been corrected in all these publications. It also has to be considered that especially nitrate concentrations are strongly dependent on variable freshwater discharges.

Statistical analyses of the monitoring data from Helgoland for the German Bight indicate that only the phytoplankton data can be assumed to be representative for the surrounding area (Radach et al. 1990 a).

A project to optimise the design of pollutant monitoring in the German Bight revealed the high variability in this area (Heinrich 1994) and the need for measurements coupling different parameter sets and time series with space oriented data. High-frequency measurements in the Elbe estuary have recently shown the significance of events in the nutrient regime (Nies et al. 1999).

In the coastal zone of The Netherlands (< 10 km) phosphate concentrations decreased since 1981, and in 1990 they reached the values of 1976 (Klein and van Buuren 1992). There was no trend for nitrate. In the area from 50 to 70 km offshore, no general trend for any of the nutrients was detectable.

High loads of ammonium are observed especially in the estuaries and coastal waters during winter due to temperature limited nitrification. But higher values can be observed in summer, caused by temporary and locally dominant remineralisation processes.

A special eutrophication aspect for the German Bight is the interaction with the Wadden Sea, which receives nutrients from the large estuaries of the rivers Rhine, Elbe and Weser (Wolff 1988; de Jonge and van Raaphorst 1995) and shows an increase from the 1960s to the 1980s. It is assumed that particulate organic material is seasonally trapped in the tidal flats and that nutrients are released following remineralisation of this material during summer (see chapter 1.4). For this reason, long distance imports of low but permanent loads also have to be considered.

There have been significant shifts of nutrient ratios in the past. At Helgoland, the increase in phosphate concentrations until the mid-1970s caused a decrease of the inorganic N/P ratio (Hickel et al. 1994). Then the ratio again increased due to the recent phosphate elimination and did not decrease thereafter, because there was no respective reduction in nitrogen discharges. A shift of nutrient ratios is assumed to stimulate *Phaeocystis* development, a species which becomes dominant under N-limiting conditions (Riegman 1994). At the same time P and Si depletion was reported to enhance *Phaeocystis* increase as well (Lancelot 1995). The contradictory findings may be explained by the recently discovered very complex life cycle of this species, and the specific sensitivity of the different morphological stages to nutrient limitation. Shifts of N/P ratios as well as frequency of nutrient pulses will generally affect the species composition in the North Sea (Riegman et al. 1992).

Since there are only few measurements of dissolved organic compounds in the coastal water, no information is available on concentration changes of DOM. With respect to POM, it remains difficult to assess any long term evolution in the area of interest, because concentrations of particulate material are controlled by the highly variable turbulence conditions.

For any assessment of trends, it is i.a. necessary to analyse the variability of data. For this reason, means and the respective variabilities have been calculated for nutrient data from the German Bight. Besides inter-annual variability of monitoring data, it was also possible to estimate short-term variability of some seasonal data sets.

Summer data are only available since 1978 as are winter data since 1985 (Fig. 2.3.0.1). The data sets originate from different sources and have been proven to be compatible. Summer monitoring data have been supplemented by data sets from project research cruises.

Sampling was mostly performed in the inner German Bight (Fig. 2.3.0.2). This has to be considered during evaluation of means and local variabilities calculated for areas of 0.5° longitudinal range and of 0.25° latitudinal range.

2.3.1 Mean winter nutrient gradients and their variability

During winter, mean surface temperatures were lowest along the coasts (< 4 °C) and highest in the open water (> 5 °C) (Fig. 2.3.1.1). Variability corresponded to the mean distributions of river plumes and was highest (> 100%) along the East and North Frisian coasts controlled by the river plumes of Rhine and Elbe/Weser. Thus, the annually changing gradients are mainly controlled by the variable fresh water discharges and prevailing wind directions.

Due to the shallowness of the area, near bottom temperatures were similar, caused by the nearly complete mixing during wintertime. Off the shallow North Frisian coast high variability was found at the surface and near the bottom, but off the East Frisian coast the variability near the bottom was lower, indicating partial stratification in this area.

Mean salinity gradients increased from the easternmost coastal stations (< 30) towards the open sea (> 34,7) (Fig. 2.3.1.2). Near bottom salinity showed higher concentrations for the inner stations, indicating some vertical salinity gradients, which are permanent near the estuaries (Frey

1990). Variability of salinity was highest in the area affected by the combined river plumes of the Elbe and the Weser, in particular at the surface near the river mouths (> 10 %). The area controlled by the mostly northward moving river plumes can clearly be identified by the salinity and temperature gradients and by the respective variability as well. Sometimes offshore winds turn the river plume towards north-west, indicated by high variability in this area too.

Due to the nutrient discharges by the rivers, the nutrient gradients followed the freshwater gradients: highest mean concentrations of phosphate occurred in the river plume areas of the Rhine/Ems and Elbe/Weser (> 1.6 µM), as well as silicate (> 20 µM) and nitrate (> 30 µM) (Fig. 2.3.1.3 - 5). The residual current, moving to the north, entrains nutrients from the southern area into the German Bight (see also van Beusekom et al. 2001). Towards the central North Sea these concentrations dropped to less than 0.5 µM phosphate, less than 0.3 µM silicate and less than 5 µM nitrate. However, it has to be considered that the most western stations may be affected by the Dogger Bank production during winter as well (Brockmann and Topcu, 2001).

Variability of phosphate was highest (30 %) off the river plume water due to its changing extension. Silicate and nitrate showed highest variability (> 80 %) at the surface near the mouths of the Elbe and Weser, probably caused by interannually changing discharges. The high variability is caused by the different spreading of the river plume, reflected also by the variability of salinity.

Because nitrification rates are strongly dependent on temperature, this process was reduced in the rivers during winter, resulting in high ammonium concentrations in the combined river plume water of the Elbe/Weser (> 6 µM) (Fig. 2.3.1.6). Also the nitrite pool accumulated, indicated by high nitrite concentrations (> 1.5 µM) in the inner as well as in the extended (> 1 µM) river plume water (Fig. 2.3.1.7).

While the variability of phosphate, silicate and nitrate was highest within the river plumes, variability of ammonium was highest (> 80 %) offshore indicating remobilisation at the frontal system of the continental coastal water. High ammonium variability was also induced by the low values (< 1 µM) in the central North Sea water. For nitrite, highest variability (> 100 %) was found west of the variability maximum of ammonium, caused by its low concentrations. This indicated the succeeding nitrification processes at changing locations of the frontal system surrounding the nutrient-rich coastal water, providing ammonium. Since DIN was dominated by nitrate during winter, the gradients and its variabilities were similar to those for nitrate (Fig. 2.3.1.8).

DIN/phosphate ratios of more than 25 in the continental coastal water indicated the unbalanced nutrient discharges with still high nitrogen loads but significantly reduced phosphate loads (Fig. 2.3.1.9). Variability was highest (> 50 %) in the continental coastal water but extended also to the west into central North Sea water.

DIN/Si ratios remained below 4, whereas ratios > 4 (partly > 6) were found in the southern central German Bight and not in the coastal water (Fig. 2.3.1.10). This indicates some balanced nitrogen loads in comparison to silicate, which is assumed not to be affected anthropogenically.

P/Si ratios were mostly between 0.1 and 0.2 in the open German Bight and dropped below 0.1 in the inner coastal water, which is an additional indication for the reduced phosphate loads (Fig. 2.3.1.11). Variability for these ratios reached values above 100 % in central areas of the German Bight and decreased towards the central North Sea, due to mixing with the deeper open water, balancing the mean concentrations.

Assuming that in rivers with no or minor damming up the discharges of silicate are less anthropogenically affected than the discharges of nitrate and phosphate, the anthropogenic contribution of these nutrients can be calculated, based on the Redfield-ratios of N/Si = 1 and P/Si of 0.06. The corresponding maps show clearly the surplus of nitrogen in relation to silicate reaching in the coastal water more than 10 µM, at some location as in the Elbe river plume more than 20 µM (Fig. 2.3.1.11, 12). In the open German Bight a relative surplus of phosphate in comparison to silicate was observed.

2.3.2 Mixing diagrams during winter

Mixing diagrams including data from all depths reveal for the period between 1985 to 1998 for nitrate and silicate mostly linear correlations without any significant differentiation (Fig. 2.3.2.1, Tab. 2.3.2.1). However, the situation for phosphate was completely different, indicating for the years before 1991 a higher phosphate discharge than for the period 1991 - 1995. This was due to the phosphate decrease in the river discharges. However, for the following period (1996 - 1998) higher phosphate was indicated again. The slopes decreased from 0.31 to 0.08 during the early 1990es and increased during the last 1990s again to 0.21. This was in the range of the yearly slopes, presented by Körner and Weichart (1991).

Tab. 2.3.2.1: Equations and correlation coefficients of correlations between nutrients and between nutrients and salinity during winter in the German Bight

	years, winter	Equation	n	r^2
Sal - NO3+2	1985 - 1997	-9.28 * X + 327.06	1017	0.916479
Sal - NO2	1985 - 1997	-0.15 * X + 6.10	364	0.177484
Sal - NH4	1985 - 1997	-1.14 * X + 40.89	1022	0.591875
Sal - Si	1985 - 1997	-4.31* X + 151.87	967	0.878374
Sal - PO4	1985 - 1990	-0.31* X + 11.45	439	0.913386
Sal - PO4	1991 - 1995	-0.08 * X + 3.51	454	0.471486
Sal - PO4	1996, 1997	-0.21* X + 7.68	128	0.784291
NO3+2 - PO4	1985 - 1990	0.06 * X + 0.62	385	0.807481
NO3+2 - PO4	1991 - 1995	0.01 * X + 0.62	453	0.448449
NO3+2 - PO4	1996, 1998	0.03 * X + 0.51	199	0.616458
Si - PO4	1985 - 1990	0.06 * X + 0.62	385	0.807481
Si - PO4	1991 - 1995	0.02 * X + 0.60	457	0.530601
Si - PO4	1996, 1998	0.03 * X + 0.51	199	0.616458
NO3+2 - Si	1985 - 1998	0.45 * X + 0.14	1038	0.887745
NH4 - NO2	1985 - 1998	0.061* X + 0.74	430	0.069997

The mixing diagrams of ammonium and nitrite did not reveal any cluster which could be assigned to specific time periods (Fig. 2.3.2.2). One reason may be that ammonium discharges by the Elbe have decreased significantly since 1990 (Fig. 2.2.3). However, from the significant correlation as well as from the gradients (Fig. 2.3.1.6) it must be assumed that the rivers are still the dominant sources for ammonium in the German Bight during winter, together with different remobilization events from the sediment. These may have affected interannual modifications, causing, together with advective movements of the river plumes, the observed high variability in central areas of the German Bight (Fig. 2.3.1.6). Nitrite was correlated with ammonium due to nitrification.

The changes of phosphate were also reflected by the correlation diagrams between the different nutrients (Fig. 2.3.2.3). While silicate and nitrate did not show any changes, giving mostly a N/Si-ratio of about 2 (M/M), the relationship between phosphate and the other nutrients changed significantly between 1990 and 1991, resulting in highest slopes for the period 1985 - 1990 and lowest for 1991 - 1995. This means that phosphate showed in relation to salinity as well as to the other nutrients a recent increase following the reduction during the early 90's. The reason for this increase is, besides remobilization from sediment, a higher discharge in 1998/1999 (Fig. 2.2.4) (see also Lenhart and Pätsch 2001).

The mixing diagrams of the nutrient ratios showed again the relative constancy of the DIN/Si-relationship within the salinity gradient, and, furthermore, the decreases of N/P- and Si/P-ratios with increasing salinity (Fig. 2.3.2.4) caused by the different relationships within the mixing endmember.

2.3.3 Time series during winter

Time series are presented for the total data and three different salinity regimes 21 - 30 (estuarine water), 30 - 33 (inner coastal water), 33 - 35.5 (outer coastal water), considering the different effects of discharges and advection (see Fig. 2.3.1.2).

Total German Bight

Mean salinity at the sampled stations in the German Bight was mainly in the range between 30 and 34 (Fig. 2.3.3.1). Due to nearly complete vertical mixing during this season no significant differences between surface and bottom water were observed, but surface water was generally less haline and in 1987 also colder. Nutrient concentrations were slightly higher at the surface, indicating the freshwater source. However, as indicated by the standard deviations, there were no significant differences.

In relation to the interannual changes of salinity, silicate, nitrate and ammonium showed some inverse correlation due to the dominant effects of nutrient-rich freshwater discharges on the nutrient concentrations. At minimum salinities in 1994 maxima of silicate, nitrate and ammonium were observed (Fig. 2.3.3.2). The development of phosphate (Fig. 2.3.3.1) was dominated by the decrease in 1987 from 1.5 µM to 0.75 µM in 1995. These time series were generally confirmed by independent sampling during the Young Fish Survey in the German Bight (Brockmann and Topcu 2001).

Correspondingly, the N/P ratios increased, having a maximum in 1994, as did Si/P ratios, starting in 1990 (Fig. 2.3.3.3). The DIN/Si ratios reached their maximum in 1990, due to low nitrate concentrations, and remained at 2.5for the other sampled period , indicating a surplus of nitrogen discharge in relation to silicate.

Estuarine water (salinity 21 - 30)

The decline of phosphate described before was also observed within the water masses of low salinities between 21 and 30, especially from 1990 (2.4 µM) to 1991 (1.5 µM) (Fig. 2.3.3.4). The lowest mean (1 µM) was observed in 1995. Compared with salinity the variation of silicate and nitrate were reversed, caused by the changing influence of freshwater discharges (Fig. 2.3.3.4, 2.3.3.5). The changes of ammonium concentrations were only partly in agreement with the other nutrients and deviated especially from 1989 to 1991, indicating independent sources and sinks. The ratios of DIN/PO_4 and of Si/PO_4 increased significantly between 1990 and 1994, mainly caused by the reduction of phosphate. The DIN/Si-ratios also increased but only slightly from 2.25 to about 2.5 (Fig. 2.3.3.6).

Inner coastal water (Salinity 30 – 33)

In the inner coastal water at salinities between 30 and 33 there was a period of higher temperatures from 1989 to 1995 (Fig. 2.3.3.7). Salinity was also increasing in 1989, but probably also affected by different sampling from year to year.

Phosphate decreased from 1.5 µM to 0.8 µM in 1995, followed by an increase to 1 µM in 1996. The inverse relationship between salinity and nutrient concentration was much less significant than for the estuarine water. Only a few indications were found for this relationship, such as in 1994, with low salinity and high nitrate concentrations (Fig. 2.3.3.8). For silicate this relation was not evident.

During the sampling period there was a weak decrease of nitrate concentrations from about 40 to 30 µM. Silicate concentrations showed a variability around 15 µM and ammonium around 6 µM. Due to the phosphate decrease DIN/phosphate and Si/phosphate ratios increased significantly, reaching their maxima in 1994 and 1997 (Fig. 2.3.3.9). The ratios of DIN/Si were mostly around 3, indicating a 3fold nitrogen surplus compared to silicate when assuming the Redfield ratio of 1:1 (Redfield et al. 1963).

Outer coastal water (Salinity 33 – 35.5)

For the outer coastal water with salinities between 33 and 35.5 mean salinity was around 34 (Fig. 2.3.3.10). The low salinity in 1993 was accompanied by high phosphate, silicate and nitrate concentrations, indicating a greater influence of riverine discharges during this year (Fig. 2.3.3.10 and 2.3.3.11). There was only a weak decrease in phosphate concentrations from about 0.8 to 0.6 µM. Silicate showed a high variability around 6 µM and nitrate around 12 µM. The silicate and nitrate maxima in 1993 corresponded with a salinity minimum. Ammonium, the means of which were in the range of 1 - 3 µM, showed highest concentrations from 1990 to 1994, as has been observed also for the other salinity regimes.

The DIN/phosphate ratio showed an increasing tendency from about 15 (close to the Redfield ratio of 16) to 25 due to the phosphate decrease (Fig. 2.3.3.12). The increase of the Si/P-ratio started in 1990 and reached a maximum in 1997 with 14 (M/M). This relationship clearly reflected the effect of decreasing phosphate concentrations.. DIN/Si-ratios remained around 2.5 in the outer coastal water, except for a maximum of 10 in 1990.

2.3.4 Mean summer nutrient gradients and their variability

Water temperature was highest (> 18 °C) in the river plume of the Elbe and in the coastal water (Fig. 2.3.4.1), decreasing especially in the bottom water towards the open North Sea. The temperature differences between surface and bottom water indicated the extent of thermal stratification covering the whole deeper area of the German Bight. Variability was mostly between 10 and 20 %, indicating that sampling had been performed during similar seasonal situations.

Mean salinity showed from west to east a decrease from more than 34.7 to less than 31 in the Elbe river plume, both at the surface and near the bottom (Fig. 2.3.4.2). Variability was lowest (< 1 %) in the western part of the German Bight and increased especially at the surface of the inner German Bight (> 3 %), which was controlled by the variable Elbe/Weser river plume affected by freshwater discharges and prevailing weather situations changing from year to year.

At the surface of the central German Bight mean phosphate concentrations were mostly between 0.1 and 0.2 µM (Fig. 2.3.4.3). In the bottom water, concentrations were often between 0.2 and 0.3 µM, caused by remineralisation of sedimented biomass. The primary production was fed by phosphate-rich river discharges (> 0.5 µM) and remobilisation along the shallow coast of the Wadden Sea, increasing the concentrations to more than 0.2 µM. Variability at the surface was especially high in the western and north-eastern part of the German Bight, surpassing 100 %. Lowest variability was found off the East Frisian Islands of less than 50 %. The low variability off the shallow coasts indicates the constancy of phosphate remobilization in these areas. At the bottom, variability was highest off the island of Sylt (> 100 %), probably caused by changing local processes at the shallow banks. Lowest variability of less than 30 % was found in a smaller central area.

Silicate concentrations were < 1 µM at the surface in the outer German Bight, in the central area between 1 and 2 µM, increasing towards the inner German Bight, in the river plumes of the Elbe and Weser, to more than 3 µM (Fig. 2.3.4.4). Variability of more than 90 % was observed in some areas of the inner German Bight as well as at the western stations in the open water. However, the highest variability was again found in an area off Sylt in bottom waters. Near the bottom, silicate concentrations were mostly between 2 and 3 µM, decreasing towards the south and the north-west and increasing to 3 – 5 µM towards central areas along the Elbe valley with 3 - 5 µM. This again indicated the remineralisation of sedimented diatoms in the bottom water. Variability was mostly between 50 and 70 %.

During summer, nitrate concentrations were reduced to less than 0.5 µM at the surface in the western German Bight, indicating possible limitation (Fig. 2.3.4.5). Mean concentrations increased in the coastal water as well as the river plumes of the Elbe and Weser (> 2 µM) and reached off Sylt more than 10 µM. Variability was highest (more than 150 %) in the inner German Bight and was mainly between 100 - 150 % at the surface of the open German Bight.

Near the bottom, nitrate concentrations were between 0.5 and 2 µM in an area spreading from the south-east to the north-west parallel to the deep Elbe valley. South-west of this area, nitrate dropped to less than 0.5 µM, to the north-east it increased to 2 - 5 µM. Variability was again highest in the inner German Bight, but also in the northern part of the sampled area, surpassing 150 %.

Mean ammonium concentrations were at the surface mostly between 1 and 2 µM in the open German Bight, increasing in the continental coastal waters to more than 2 µM, partly to 3 - 4.5 µM in the area off Sylt (Fig. 2.3.4.6). Here and in the inner German Bight variability was highest with more than 100 %. Near the bottom in the outer western part of the German Bight ammonium concentrations were below 2 µM, increasing in the Elbe valley to more than 4 µM and reaching a maximum in the inner German Bight of more than 7 µM. This is a clear indication of remineralisation within the bottom water. Variability was again highest (>100 %) in the inner German Bight and off the island of Sylt.

Nitrite concentrations were at the surface mainly below 0.1 µM and increased in the coastal water to 0.2 - 0.5 µM (Fig. 2.3.4.7). Variability was again highest (>100 %) in the inner German Bight decreasing towards the north-west. Near the bottom, nitrite was again mostly below 0.1 µM and increased towards the shallow coast of Schleswig-Holstein to 0.2 - 0.5 µM as at the surface. Variability was lowest in central areas, then increasing towards the inner and outer German Bight above 60 %.

The generally high variability off Sylt indicated an area of highly variable nutrient concentrations during summertime probably influenced by up- and downwelling as well as remobilization forced by the shallow Amrum banks. It can be assumed that in this area especially ammonium was remobilized and successively nitrified, causing the observed increases of mean ammonium, nitrite, and nitrate concentrations.

The DIN concentrations during summer were mostly controlled by the ammonium concentrations, reaching highest values (> 5 µM) towards the coast and near the bottom, indicating the remineralisation in progress (Fig. 2.3.4.8).

DIN/PO_4 ratios of more than 120 (M/M) were observed in this area off Sylt and at the frontal area of the north-westward propagating Elbe/Weser river plume (Fig. 2.3.4.9). Ratios between 5 and 20 were only observed in distinct western and northern parts of the German Bight. Highest variability (> 200 %) was found in the inner German Bight. Near the bottom, ratios below 20 were detected in the whole western part of the German Bight, increasing in central areas to above 20. In the inner German Bight the variability of more than 100 % was highest. It has to be stressed that N/P ratios were lower along the shallow tidal flats than in the central German Bight, caused by the remobilization of phosphate during this season.

Mean DIN/Si ratios were very patchy with 4 - 6 in the central German Bight increasing to partly more than 10 in the north-eastern and at some locations south-west of the river plume area (Fig. 2.3.4.10). Variability was highest in the inner German Bight, surpassing 300 %. Near the bottom in the western German Bight ratios below 2 were observed, in the central part mostly between 2 and 4, increasing towards the shallow coast. Again, high values were observed both at the surface and near the bottom off Sylt. Variabilities were similar at the surface with more than

100 % in the inner and central German Bight, decreasing towards the open water. This means that especially at the surface the silicate balance was low in comparison to nitrogen, caused by the longer lasting fixation within the shells of diatoms.

PO_4/Si ratios were mainly between 0.1 and 0.2 in the central German Bight and increased especially in frontal areas of the river plume of the Elbe and off Sylt (Fig. 2.3.4.11), indicating local silicate fixation. Variability was highest in the outer river plume area (> 100 %) and off the island of Sylt (> 150 %). This could be attributed to both the variability of phosphate (Fig. 2.3.4.3) and of silicate (Fig. 2.3.4.4) reaching highest values off Sylt as well.

Assuming
(i) Redfield ratios of 1:1 for N:Si and 1:16 for P:Si as natural relationships,
(ii) that silicate was not anthropogenically influenced and can be used as natural background,
it can be concluded that during summer there were only few spots with higher nitrogen loads, indicating anthropogenically increased nitrate, but extended areas with relatively high phosphate concentrations (Fig. 2.3.4.11, 12).

Such areas were detected at the surface between the Elbe and Weser rivermouths and furthermore off Sylt. However, the latter spot cannot have been caused directly by anthropogenic influences, because it is far away from direct discharge sites. Rather, decoupling of remineralisation was the reason for these high ratios, because silicate fixed within the cell walls of diatoms will be remobilised much later than nitrogen.

A surplus of phosphate relative to silicate was detected within the inner coastal water, but also at some north-western stations. For this reason, the high values near the tidal flats can be interpreted as indirectly anthropogenically affected, because phosphorous will be trapped transiently in the Wadden Sea and remobilized during summer.

2.3.5 Mixing diagrams during summer

The mixing diagrams showed no clear relationships between nutrients and salinity during summer (Fig. 2.3.5.1, 2.3.5.2). Apart from nitrate, the nutrients often showed maxima combined with low salinities but for the total data set there was no significant correlation (Tab. 2.3.5.1). This means that discharges did not control the nutrient gradients during summer but rather internal biogeochemical processes with different turnover times of the individual nutrients. Another reason for the missing linear relationships may be the nitrogen depletion and the remobilization of phosphate and silicate during summer.

Even for the separately analysed surface or bottom data there were no significant correlations. In the shallow German Bight the separating effect of the thermohaline stratification was only reflected by the accumulation of low values at the surface and high values near the bottom. That held in particular for silicate.

Due to the dominance of internal conversion processes, a separate analysis of time periods did not reflect the reduction measures for phosphate, as has been seen for the mixing and correlation diagrams during winter (Fig. 2.3.2.1 and 2.3.2.3).

Also for the correlations between the different nutrients no linear relationships were found, apart from certain similarities (Tab. 2.3.5.1) between nitrite and ammonium and between phosphate and silicate (Fig. 2.3.5.3). The relationship between nitrite and ammonium was determined by the nitrification. The correlation between silicate and phosphate, nutrients being simultaneously released to some degree, was also found during summer in this area, reflecting a parallel remobilization in the bottom water and consumption in the mixed layer and resulting in a mean ratio of 18 (M/M). This was close to the assumed Redfield-ratio of 16, reflecting natural conditions.

Correspondingly, the mixing diagrams of the nutrient ratios did not reflect any clear relationship (Fig. 2.3.5.4). There was a high scattering in the salinity range between 31 and 34.

Tab. 2.3.5.1: Equations and correlation coefficients between nutrients and between nutrients and salinity during summer in the German Bight

	years, summer	Equation	n	r^2
Sal - NO3+2, surface	1986 – 1998	-0.47 * X + 16.54	473	0.0914109
Sal - NO3+2, intermediate layers	1986 – 1998	-0.07 * X + 3.74	877	0.0022806
Sal - NO3+2, bottom	1986 – 1998	-0.21* X + 8.91	431	0.0159831
Sal - NO2, surface	1986 – 1998	-0.02 * X + 0.73	382	0.0710329
Sal - NO2, intermediate layers	1986 – 1998	-0.02 * X + 0.73	774	0.0698431
Sal - NO2, bottom	1986 – 1998	-0.02 * X + 0.85	339	0.0899517
Sal - NH4, surface	1986 – 1998	-0.15 * X + 7.08	475	0.0101839
Sal - NH4, intermediate layers	1986 – 1998	-0.26 * X + 11.35	877	0.0175943
Sal - NH4, bottom	1986 – 1998	-0.25 * X + 12.03	432	0.0150538
Sal - Si, surface	1986 – 1998	-0.42 * X + 15.40	474	0.117276
Sal - Si, intermediate layers	1986 – 1998	-0.21 * X + 9.21	875	0.0229572
Sal - Si, bottom	1986 – 1998	-0.27 * X + 12.06	431	0.035019
Sal - PO4, surface	1986 – 1998	-0.04 * X + 1.51	469	0.0750841
Sal - PO4, intermediate layers	1986 – 1998	-0.01 * X + 0.63	875	0.0085579
Sal - PO4, bottom	1986 – 1998	-0.03 * X + 1.21	429	0.0302573
NO3+2 - PO4, surface	1986 – 1998	0.12 * X + 1.54	500	0.0691188
NO3+2 - PO4, intermediate layers	1986 – 1998	0.23 * X + 1.88	951	0.106263
NO3+2 - PO4, bottom	1986 – 1998	0.21 * X + 2.77	482	0.0762387
Si - PO4, surface	1986 – 1998	0.07 * X + 0.07	496	0.318075
Si - PO4, intermediate layers	1986 – 1998	0.05 * X + 0.11	480	0.208532
Si - PO4, bottom	1986 – 1998	0.05 * X + 0.08	949	0.226867
NO3+2 - Si, surface	1986 – 1998	0.12 * X + 1.54	500	0.0691188
NO3+2 - Si, intermediate layers	1986 – 1998	0.23 * X + 1.88	951	0.106263
NO3+2 - Si, bottom	1986 – 1998	0.21 * X + 2.77	482	0.0762387
NH4 - NO2, surface	1986 – 1998	0.02 * X + 0.07	410	0.0691549
NH4 - NO2, intermediate layers	1986 – 1998	0.02 * X + 0.07	850	0.186221
NH4 - NO2, bottom	1986 – 1998	0.02 * X + 0.10	391	0.140443

2.3.6 Time series during summer

Total German Bight

For the total water masses in the German Bight the mean surface temperature was varying around 16 °C, with a maximum of 20 °C in 1994 (Fig. 2.3.6.1). Bottom temperature was around 12 °C, showing a similar variability. In 1987 and 1989 significant differences between surface and bottom temperatures were detected. Salinity reached during 1994 a minimum of 32 and was generally fluctuating around 33, both at the surface and near the bottom. In most cases bottom salinity was higher than at the surface.

Mean phosphate concentrations (about 0.25 µM) showed higher values for the time period after 1992 than before. This was also valid for the bottom water, where the concentrations were often up to 0.1 µM higher than at the surface, caused by nutrient consumption in the mixed layer and remineralisation above the bottom. The same was true for silicate, with a mean at 1.5 µM at the surface and 3 µM near the bottom. In 1992 small maxima occurred at the surface for phosphate, silicate (Fig. 2.3.6.1), nitrate and nitrite (Fig. 2.3.6.2), caused probably by river inflow or advection from other sources which was not reflected by low salinity. High nitrate and nitrite concentrations in 1987 were accompanied by a high variability, mainly originating from sampling near the Elbe estuary, which exhibits steep gradients. Nitrate showed a decreasing tendency (5 to 1 µM), as did nitrite. Ammonium reached a relatively high level (> 3 µM) from 1989 to 1991 and again in 1996 and 1998. For ammonium, bottom values were by more than 1 µM generally higher than at the surface, indicating remineralisation of sedimented biomass. These high ammonium concentrations in the bottom water underline the significance of remineralisation, affecting also phosphate and silicate.

With the exception of 1991 DIN/P ratios remained below 100 (Fig. 2.3.6.3). DIN/Si ratios were generally below 10 and often higher at the surface (Si fixation) than near the bottom. Si/P ratios were mostly below 25 and showed also a maximum in 1991. Therefore, it can be assumed that the high DIN/P values in 1991, which showed a high variability, were caused mostly by low phosphate concentrations (Fig. 2.3.6.1). The cruises in 1991 and 1992 were performed in June, before phosphate was remobilized seasonally. However, in 1992 high phosphate concentrations were detected in June as well.

Estuarine water (Salinity 21 – 30)

For the estuarine water masses with salinities between 21 and 30 only few data sets were collected due to the low freshwater discharges during summer. Temperature was highest (20 °C) in 1994, as was salinity (29.8) (Fig. 2.3.6.4). At the same time, relatively low phosphate and silicate values were observed as well as low concentrations of all nitrogen nutrients, which showed a decreasing tendency (Fig. 2.3.6.4 and 2.3.6.5). DIN/P as well as Si/P showed increasing tendencies (Fig. 2.3.6.6).

Inner coastal water (Salinity 30 – 33)

Within the inner coastal water of salinities between 30 and 33 all nutrients as well as salinity and temperature showed strong fluctuations (Fig. 2.3.6.7 - 9). There were some inverse relationships between salinity and nutrients, indicating effects of river discharges, correlating salinity dilution with nutrient increase. Highest phosphate and silicate concentrations were observed in 1992 at low salinity (31.5). Highest DIN/P ratios were found in 1991, caused by high nitrate and low phosphate values. The latter were also causing high Si/P ratios. The high DIN/Si ratios in 1990 and 1996 were coupled with silicate minima. However, flagellates were dominant at Helgoland roads since 1979 and diatoms in 1996 (Fig. 2.5.1.1) but no significant correlations between nutrients and phytoplankton abundance were detected.

Outer coastal water (Salinity 33 – 35)
In the outer coastal water at salinities between 33 and 35 again no trends were observed (Fig. 2.3.6.10 - 12). Salinity was lowest (33.5) in 1994. Mean phosphate fluctuated between 0.1 and 0.3 µM, silicate between 1 and 3 µM. Nitrate had a maximum in 1992 with 2.5 µM. Nitrate was mostly below ammonium concentrations, caused by remineralisation as dominating process. Ratios of DIN/P and Si/P were highest in 1991, when low phosphate concentrations were observed. Since ratios were calculated from individual values, the means of ratios were higher than those estimated from the means of the individual nutrients.

2.3.7 Short-term variability of nutrients in the German Bight

For the interpretation of time series the representativeness of the data has to be considered. Representativeness in space can be achieved by calculation of means from sampling in a specific area, as has been done for the German Bight or salinity regimes. However, representativeness in time is difficult to gain from short cruises.

Monthly measurements have shown a high seasonal variability in the German Bight (TUVAS, Heinrich 1994). For this reason it is difficult to estimate "typical summer" values from cruises performed in the period from June to August (Fig. 2.3.0.1). Especially during summer cruises, interannual variability (chapter 2.3.6) will be overlaid by seasonal differences, caused by different development stages of the ecosystem.

However, even regular measurements during the same period are affected by patchiness, especially during summertime. This means that short-term variability within a couple of days will also influence the representativeness of seasonal sampling.

During the research project KUSTOS, repeated sampling of an identical station net was carried out four times in summer, spring and winter. The sampling lasted 14 to 20 days each time (see Tab. 2.3.7.1). That means that each grid was covered within about four days. This case study yielded some information about possible short-term variability.

Salinity, shown as surface means, was lowest during spring, due to high freshwater discharges by the Elbe and the Weser (Fig. 2.3.7.1). During this season variability was highest (> 16 %) near the estuaries of this rivers, caused by fluctuating gradients by tidal action and changes of discharge rates and wind induced current, controlling the shape and extent of steep gradients as well. Within the frontal area of river plumes variability was also highest during the other seasons (> 2 %) (Fig. 2.3.7.2). Variability of surface temperature was highest during winter (partly > 500 %) due to low values (< 1 °C).

Suspended matter, or seston, is an important factor in eutrophication because high loads may influence the light regime and can be an indicator for resuspension of surface sediment, which is often linked to the remobilisation of nutrients. On the other hand, suspended material itself contains organic matter, which is a source of nutrients directly (urea, free amino acids) or after remineralisation. By this process oxygen will be consumed, which may contribute to oxygen depletion in the bottom layer as well.

Tab. 2.3.7.1: Time table of repeated measurements in the German Bight during three selected seasons

See Figs. 2.3.7.1 - 13

Summer	- grid 0	23.07. - 25.07.1994
	- grid 1	26.07. - 29.07.1994
	- grid 2	02.08. - 04.08.1994
	- grid 3	07.08. - 11.08.1994
Spring	- grid 0	22.04. - 24.04.1995
	- grid 1	24.04. - 27.04.1995
	- grid 5	02.05. - 04.05.1995
	- grid 8	08.05. - 11.05.1995
Winter	- grid 0	24.02. - 26.02.1996
	- grid 1	27.02.1996 (Heli)
	- grid 2	01.03. - 03.03.1996
	- grid 3	08.03. - 10.03.1996

Heli = Sampling by Helicopter

During all investigated seasons suspended matter concentrations increased at the surface towards the shallow coast of Schleswig-Holstein (Fig. 2.3.7.3). This was caused by resuspension and by phytoplankton accumulation (Fig. 2.3.7.13). Highest values were found during winter (> 20 mg/L). Variability was highest during winter as well (> 100 %). High variability was detected in different areas of the German Bight, which only showed some relationships with the variability of chlorophyll and particulate organic material during summer (Fig. 2.3.7.11 - 13). For that season it can be assumed that seston was mainly influenced by the distribution of phytoplankton biomass.

Similar to suspended matter gradients, turbidity, analysed with a Turner nephelometer, measuring the reflection of light by particles, was highest (defined as NTU = nephelometric turbidity units) near the shallow coastlines (Fig. 2.3.7.4). Clear water gives turbidity values near 0. Variability surpassed 100 % in the central German Bight during winter. It should be noted that the winter of 1996 was exceptionally cold, and an absence of stratification allowed a permanent exchange between surface and bottom water. Generally, the variability of turbidity was in the same range as that of suspended matter. These parameters give an indication of light limitation in the shallow coastal water with its high turbidity (see also chapter 2.1.3). It can be assumed that at suspended matter loads of 50 mg/L the depth of the euphotic zone will be restricted to the upper meter (Fast 1993; Ladwig 1997).

Phosphate was highest during winter due to low primary production (Fig. 2.3.7.5). During spring and summer, the highest values were observed in the river plumes and during summer also in some coastal areas. Corresponding to the surface concentrations, variability was lowest in winter (< 30 %) at high concentrations and highest during the other seasons (60 - 140 %) at low concentrations. This and the large extension of the high variability indicated the difficulty of representative phosphate monitoring during summer. In this connection, relationships to salinity gradients are not helpful due to the absence of any significant correlations between nutrients and salinity during summer (Fig. 2.3.5.1, 2).

Silicate, being an important reference parameter for eutrophication (see also chapter 2.3.1), showed seasonal distributions similar to phosphate and similar variabilities (Fig. 2.3.7.6). In spring high silicate concentrations were still detected in the river plume of the Elbe and the Weser. Along the plume fronts, variability between the successive grid samplings was highest due to tidal action and general advection.

Nitrate means were affected as well by the river plume (Fig. 2.3.7.7). During winter, high concentrations were mainly restricted to the eastern coastline, whereas during spring no decrease was observed as for phosphate and silicate because nitrate concentrations were excessive compared to the other nutrients (Brockmann et al. 1999a). Nitrate variability was highest during summer, mostly due to low nitrate concentrations (<10 µM). However, in spite of the low nutrient concentrations, monitoring during this season would have been most effective because oxygen depletion occurred in the bottom water (see Fig. 2.7.8). Comparing the distribution of high variability during the different seasons, the affected areas were completely different. Therefore, "hot spots" for increased monitoring activity cannot be deduced from these findings.

Apart from the winter season, when nitrification is an important process in the central German Bight, means of nitrite were similar to those of nitrate (Fig. 2.3.7.8), with a similar distribution of variabilities. During winter, ammonium showed a minimum where nitrite had a maximum, indicating the ammonium consumption in the central western German Bight (Fig. 2.3.7.9). High ammonium concentrations in the river plume were enhancing nitrification in these areas. Gradients of variability were completely different compared to those of the other nutrients.

Dissolved organic nitrogen, which becomes the dominant nitrogen fraction during summer (Butler et al. 1979; Brockmann et al. 1999a), reached maxima in the central German Bight (Fig. 2.3.7.10). Again, this important nitrogen fraction, which is usually neglected in monitoring programs, showed a different pattern of variability in comparison to the inorganic nitrogen nutrients. The highest variability was found in the central bight (winter) or at the north-western borders of the sampled grids.

Particulate nitrogen showed maximum winter concentrations along the coast of Schleswig-Holstein where variability was also highest (Fig. 2.3.7.11). During spring and summer the distribution of particulate nitrogen was more patchy. Maxima were observed at the river plume fronts where production often reached maxima at the border between nutrient-rich turbid and less nutrient-rich clear water. In spring and summer, areas of high variability were found in the central German Bight. Similar distributions of means and variability were found for particulate carbon (Fig. 2.3.7.12), indicating the close relationship of these biomass parameters. However, in comparison with chlorophyll fluorescence (Fig. 2.3.7.13) only for winter was a similar distribution detected, caused by a local bloom during calm weather conditions in North Frisia (Brockmann et al. 1999a). During spring, only for some locations around the river plume was a coincidence of maxima found. During summer, fresh organic matter was probably quickly degraded, indicated by only small overlapping of maxima of chlorophyll and particulate carbon. Variability of chlorophyll was highest (> 70 %) during spring in large areas of the German Bight. This underlines the difficulty of representative monitoring of this "key" parameter.

2.3.8. Seasonal phase distribution of N and P

Gradients of total N in the German Bight clearly showed the importance of river discharges by high concentrations in the estuaries (Fig. 2.3.8.1) (Brockmann et al. 1999a). The area between the Jade and Eiderstedt was characterised by high nitrogen loads (> 100 µM, during summer > 50 µM), especially during winter and spring. Highest concentrations were found during spring when the freshwater discharge reached maxima (about 1000 m^3/s), and lowest concentrations were found during summer due to low discharges and ongoing consumption and partial sedimentation.

The seasonal conversion of nutrients was well reflected by the phase distribution. The contribution of DIN (M %) was highest during winter at low phytoplankton activity and still dominant during spring, because of the high discharge rates (Fig. 2.3.8.2). During summer DIN became the minor fraction in the German Bight, due to conversion from particulate and dissolved organic compounds. DON was the dominant phase during summer (Fig. 2.3.8.3), but was also present in winter and spring in the open water. Lower concentrations were found in the tidal flats due to accumulation and decomposition activity, bound to suspended matter. Dominance of PN was only detected during spring and summer at some locations in the tidal flats, probably mostly originating from resuspension in this shallow area and some local net primary production as during winter 1996 (Fig. 2.3.8.4).

Apart from the estuaries also the tidal flats are sources of total P, as indicated by increasing concentrations during all sampled seasons (Fig. 2.3.8.5). During winter, phosphate was by far the dominant fraction (> 75 M %), decreasing during spring and then becoming the minor nutrient due to surplus of nitrogen and dominating only in the inner river plume. In summer phosphate again increased, due to remobilisation in the tidal flats (Dick et al. 1999; van Beusekom et al. 1999) (Fig. 2.3.8.6). The fraction of DOP can be neglected during winter. In spring DOP reached one quarter of TP and became dominant during summer in a central area of the German Bight (Fig. 2.3.8.7). Similar to PN , PP was kept in suspension in the shallow Wadden Sea, causing its dominance at many locations in this shallow area (Fig. 2.3.8.8). During winter PP contributed < 10 % to total P in the central German Bight.

From these findings it is obvious that for budgeting nutrient elements in the coastal water and in the North Sea as well (Brockmann and Kattner 1997) the organic fractions can no longer be neglected in the monitoring programs, otherwise it will be still more difficult to understand eutrophication effects. N and P bound within the fraction of dissolved organic compounds can be utilised directly (urea, free amino acids) or after conversion. Particulate material will contribute to oxygen depletion in the bottom layer. Moreover, many sorption processes will influence the effects of contaminants and, by this, the abundance and composition of sensitive species, interfering with eutrophication effects .

2.4 Increase in primary production and phytoplankton biomass

2.4.1 North Sea 'Greenness'

Aggregated 'Greenness'-data taken as an equivalent for phytoplankton density and compiled from long-term CPR-surveys (Continuous Plankton Recorder) revealed an overall increase in phytoplankton biomass for the offshore areas of the North Sea since the mid 1980s (Reid et al. 1998). Increasing absolute 'Greenness' indices as well as extension of the growth season were observed. Especially, a shift in the summer 'Greenness' index from moderate values in the 1950s to continuous high phytoplankton biomass throughout the growth season since the mid 1980s has occurred (Fig. 2.4.1.1). It is not possible to correlate the CPR data directly with other biological or abiotic parameters (such as nutrient concentrations), because these are not measured in the CPR itself. However, 'Greenness' data from smaller parts of the North Sea generally show consistently higher phytoplankton biomass in the southern compartment including the German Bight, which is directly affected by coastal runoff. This is in keeping with the highest annual primary production levels throughout the entire North Sea, as evidenced by the investigations of the NERC project (see below).

2.4.2 Phytoplankton biomass at Helgoland Roads

Analysis of the German Bight long-term series Helgoland Roads (Hickel 1998) supports the assumptions of phytoplankton biomass increase since the 1980s drawn from the CPR-surveys for offshore areas of the North Sea. The phytoplankton biomass determined by means of cell counts and volumetric determination of the carbon content exhibits a sharp increase in phytoplankton carbon (PPC) concentrations at the end of the 1970s, which was concomitant with rapidly increasing riverine N and P loads to the German Bight. The annual median PPC concentrations increased from an average of 15 µgC/l before 1979 to a mean value of 24 µgC/l in the 1980s. Concentrations still remained at an elevated level until 1996, which marks the end of the data made available (Fig. 2.4.2.1). A more detailed analysis of the dataset kindly provided by W. Hickel and J. v. Beusekom (BAH/AWI) revealed that the steep increase at the end of the 1970s cannot be attributed to an increase in absolute summer PPC maxima (Fig. 2.4.2.2). It is rather due to an increase in the median biomass of flagellates during the growth season (see chapter 2.5.1) and, to a lesser extent, in winter (ibid., p. 607). Hickel (1998) attributed the observed increase in phytoplankton biomass to the nanoflagellate fraction only.

The Helgoland roads permanent station is located relatively close to the coastline and is hence under the variable influence of the Elbe and Weser runoff. This might be the reason for an earlier increase in phytoplankton biomass when compared to the increase in the more offshore regions, as evidenced by the CPR index. This hypothesis is supported by long-term investigations in the Dutch coastal zone at Marsdiep, which also reveal an increase in chlorophyll-a concentrations in the end of the 1970s. A drawback in the long-term data from Helgoland Roads is that there is no normalisation for salinity which would account for the strong variability in freshwater influence. However, considering the large number of data points, the long-term evolution of phytoplankton biomass appearing in the smoothed graphs rather reflects average conditions over a broad range of salinities (29 - 35).

However, the observed change in phytoplankton stocks at Helgoland Roads could not be identified definitely as a result of the eutrophication process because of overriding effects of hydrographic changes in the Helgoland region (Hickel 1998).

Other comparative investigations of meteorological long-term measurements (e.g. air pressure as a non-locally determined parameter) at LV Elbe 1 show no clear evidence for assuming that a noticeable climatic change has taken place during the 23 years of investigation (Radach and Berg 1986). In keeping with this, Radach et al. (1990) assume that only the anthropogenic eutrophication can be responsible for the increase of phytoplankton biomass.

After Hickel et al. (1997, p. 66) it is possible that the observed increase of flagellate biomass at Helgoland Roads is a consequence of the simultaneous change in N/P ratios (ibid., p. 66). Nevertheless, the authors also note that climatological influences on the hydrographic regime can play a role in a modification of phytoplankton succession.

In sum, Hickel et al. (1997, p. 88) concluded that it is not possible to derive a direct effect of eutrophication on the phytoplankton development on the basis of the Helgoland Roads data set, but that there is indirect evidence for eutrophication effects in the area such as:

- the elevated phosphate and nitrogen concentrations in winter at higher salinities (> 33),
- the long-distance transports of particulate organic nutrients required to sustain the observed large plankton blooms in the northern German Bight and Fanø Bight; these substances sink and are remineralized far away from their supposed sources and improve the nutrient supply in the area concerned (Hickel et al. 1997, p. 88),
- the accumulation of organic substances in bottom water which is subjected to elevated oxygen consumption (Hickel et al. 1989),the observation of oxygen deficiency in the bottom water of the south-eastern North Sea since the early 1980s.

2.4.3 Chlorophyll concentrations

Trends

At the permanent station of Norderney, which is situated at the seaward border of the East-Frisian Wadden Sea, chlorophyll concentrations have been monitored fortnightly since 1985. The annual cycles for active and inactive chlorophyll as well as maximum summer chlorophyll concentrations for this period were plotted on the basis of the dataset kindly provided by M. Hanslik (NLÖ). Annual total chlorophyll maxima (~2/3 active + 1/3 inactive) regularly reached concentrations > 50 µg/L. Very high total chlorophyll concentrations with more than 120 µg/L were observed recurrently (Fig. 2.4.3.1). Whereas the annual cycle shows an interannual variability lacking any obvious trend, especially in absolute total chlorophyll maxima, the annual summer maxima in chlorophyll (period July - September) exhibit a significant trend (r = 0.73) towards slightly decreasing concentrations since 1993 (Fig. 2.4.3.2). As reported for the western Wadden Sea, there was no reduced primary production due to decreasing nutrient inputs (De Jonge 1997, Lenhart 1999, 2001).

The AlgFes (Algen Früherkennungssystem) monitoring programme for the northern German Wadden Sea was started in 1990 and includes two offshore stations in the German Bight: west of the island of Sylt and east of the island of Helgoland (sampling stations 10 and 3 in Fig.

2.4.3.3). The data, kindly transmitted by J. Göbel (LANU), were processed in the same way as those of the Norderney monitoring station. The data of the AlgFes stations reveal relatively low but quite regularly occurring annual chlorophyll-a maxima approximately of about 15-20 µg/L (Figs. 2.4.3.4, 2.4.3.5), interrupted by single peak events with up to 265 µg/L during massive blooms. Summer maxima in chlorophyll-a concentrations (period July-September) decrease slightly but significantly (r = 0.61) at the station west of Sylt (Fig. 2.4.3.6). The station east of Helgoland does not show any trend in summer chlorophyll-a concentrations.

Spatial distribution

A series of synoptic studies carried out in the framework of different research projects gives an overview of the geographical distribution of chlorophyll-a in the inner German Bight. Chlorophyll-a summer surface concentrations, averaged from data of 10 surveys conducted during the KUSTOS project in the period July to August 1994 revealed relatively moderate maxima of about 12 µg/L near the coast in the NE corner of the study area (Fig. 2.4.3.7). Lowest concentrations were found in the outermost western parts. This pattern may be a consequence of the nutrient supply originating from the Elbe discharge. After leaving the maximum turbidity zone in the estuary, light conditions ameliorate steadily so that an improved nutrient uptake by phytoplankton is enabled in the transient zone of both sufficient nutrients and light availability. The corresponding distribution pattern of chlorophyll-a concentrations in 5 m depth is generally coherent with that of the surface, but shows a more extended area of higher concentrations in the northern part. For variability see Fig. 2.3.7.13.

Highest chlorophyll-a levels were found in areas exhibiting oxygen deficiency (< 4 mg O_2/L) in the bottom water, as observed in August 1994 (see Fig. 2.7.8). The inverse relationship between surface chlorophyll-a levels and oxygen bottom water concentrations was significant at the 10 % confidence level.

On the basis of the oxygen consumption project ('Sauerstoffzehrung im Bodenwasser der Deutschen Bucht') Hickel et al. (1989) calculated a BOD_{22} of > 4 mgO_2/L, corresponding to a reduction of oxygen concentrations of more than 50 % after 22 days for an in situ chlorophyll-a concentration of 10 - 15 µg/L in the bottom water of the south-eastern German Bight (Fig. 2.4.3.8). This semi-experimental investigation conducted in 1986 showed that even relatively low concentrations of phytoplankton biomass can, under certain meteorological conditions, lead to oxygen deficiency. The spatial distribution and absolute amount of chlorophyll-a concentrations in 1986 was similar to that during the oxygen deficiency event in summer 1994. However, in contrast to 1994, oxygen deficiency was not detected in the region, which may be due to shorter residence times of the bottom water or differences in the total organic load.

The summer chlorophyll-a concentrations in August 1987 and August 1988 as described by von Westernhagen et al. (1989) again exhibit similar spatial distribution patterns as those described above with generally higher concentrations in coastal waters when compared to offshore areas (Fig. 2.4.3.9). In 1987 maximum chlorophyll-a concentrations of > 20 µg/L and a phytoplankton carbon concentration of > 1500 µg C/L were observed in the region from Horns Rev (Denmark) to the coastal waters west of the island of Sylt. This massive and presumably passive accumulation of planktonic biomass could be attributed almost exclusively to dinoflagellates (Fig. 2.4.3.10) especially to *Ceratium furca* (see also Tab. 2.5.2.1). In spite of the high amount of degradable organic matter present in this sea area, oxygen deficiency was not observed during the survey .

The investigations mentioned above illuminate the spatial distribution of phytoplankton biomass in the inner part of the German Bight, generally featuring higher concentrations near the coast and lower levels offshore, often with maxima north of the island of Helgoland and off the Danish west coast. It can thus be supposed that the fertilising effect of continental discharges will exhibit itself at a considerable distance from the estuary due to the remote transport of organic particles (Hickel et al. 1989). Data for offshore areas west of 6° E are widely lacking, hence, remote sensing is introduced to evaluate the situation in the outer parts of the study area. The surface chlorophyll-a distribution in the North Sea can be revealed by 'SeaWifs' (SEA-VIEWING WIDE FIELD-OF-VIEW SENSOR) an optical sensor installed on the SeaStar spacecraft, launched in 1997. The series of images (4 out of 7 available composites for August and September) shown in Figs. 2.4.3.11 a - d represent typical summer situations during the period 1997 - 2000. The images are composites of monthly averaged chlorophyll-a concentrations. Whereas concentrations in the central North Sea and in the outer region of the German Bight were usually below 1 µg/L, high summer chlorophyll-a levels > 10 µg/L regularly prevailed in the inner German Bight.

The comprehensive evaluation of the available chlorophyll-a data in the area reveals that according to the definitions of eutrophication as outlined by the CSTT 1997 (Comprehensive Studies Task Team), at least the inner German Bight has to be considered as an eutrophic area, since summer chlorophyll-a concentrations persistently exceed the threshold value of 10 µg/L.

2.4.4 Primary production

A main question is whether phytoplankton production has changed since anthropogenic eutrophication has started. However, there are no long-term data of primary production available for German territorial waters in the North Sea. During the past three decades a doubling of means of the phytoplankton biomass produced annually was observed along the Dutch coast (Riegman 1994). Until approximately 1975 there was an increase of phytoplankton biomass in the southern North Sea which continued in the Dutch coastal waters until 1983 (Reid et al. 1990).

Because of the lack of data in the study area it is not possible to draw a conclusion with respect to significant long-term changes in the formation of primary organic matter. However, considering the north-eastward direction of the residual coastal current pattern it can be speculated that a similar trend may prevail in the region as in the Dutch coastal water: Long-term data at Marsdiep near the island of Texel revealed that annual primary production has increased from 150 g C/m² in the period 1964 - 1976 to about 300 g C/m² at the end of the 1980s (de Jonge 1990). A direct response in terms of increased primary production and an increase of biomass following periods of high river discharges as observed along the Dutch coast (Schaub and Gieskes 1991) suggest a batch-culture-like behaviour of the coastal community. This is of relevance to the situation off the German coast, since different particulate and dissolved material will be imported from the Dutch coast to this area by the coastal current.

Recent estimations of the primary production in the German Bight calculated by Rick et al. (1998) resulted in an annual production of about 400 g C/m². Fig. 2.4.4.1 gives a compilation of the few existing data for annual primary production, as measured in different parts of the area of interest. There is a large interannual variability lacking any obvious trend. Highest values of more than 600 g C/m²·a were measured at Helgoland Roads in the late 1960s (Hagmeier 1967, 1968, 1969, 1970, pers. comm.). Some of the measurements, however, were made with different methods and have thus to be interpreted with care. Nevertheless, the German Bight is classified as the most productive region of the North Sea in terms of primary production (Joint and Pomroy 1993), with peak values usually occurring in summer in the inner part of the area (Hagmeier, pers. comm.). As an approach, chlorophyll-a concentrations (see above) can be taken as an indicator for primary production, since 65 % of its variation in the coastal North Sea can be explained by changing levels in chlorophyll-a (Joint and Pomroy 1993).

Following the classification of Nixon (1995), which defines an area to be eutrophic if the organic carbon supply exceeds 300 g C/m² y, the German Bight has to be considered explicitly as a eutrophic area.

However, with respect to seasonal eutrophication events leading to undesirable effects such as short term bloom eruption it should be noted that accumulated primary production on an annual basis is a weak indicator.

2.5. Disconnection of food webs

2.5.1 Changes of functional phytoplankton groups

A shift in functional phytoplankton groups changes the food available for higher trophic levels such as herbivorous zooplankton and, through the trophic cascade, for fish and other top-predators. Eutrophication may result in an enhanced growth of smaller phytoplankton species, which in turn favours the development of smaller zooplankton organisms (Fransz et al. 1992) resulting in the proliferation of undesirable forms such as jellyfish. An enhanced growth of flagellates, which profit from excess nitrogen and phosphate supply in relation to silicate, can lead to the formation of less edible species. This may thus cause enhanced sedimentation of degradable material, and, as a consequence, oxygen deficiency in the bottom water, if meteorological conditions are adequate. Roughly, a non-limiting Si and DIN supply at a Si/DIN ratio of > 1 predominantly results in the development of diatoms, which promotes the transfer of organic matter through the trophic food chain to copepods and finally to fish while low ratios lead to reduced zooplankton grazing (Officer and Ryther 1980). This is in keeping with observations in the Bay of Brest and at the south-eastern US-coast (Turner et al. 1998).

The described increase in PPC concentrations observed near Helgoland (Fig. 2.4.2.1.) is combined with a shift in the quantitative dominance of functional phytoplankton groups. A fragmentation of the total PPC concentration into diatoms and flagellates (Fig. 2.5.1.1) reveals a pronounced increase in flagellates carbon concentrations in the period of accelerated nutrient loads at the end of the 1970s. In the early 1960s to the mid 1970s flagellate carbon concentrations did not exceed 10 µg C/L while in the 1980s and 1990s annual median PPC concentrations were significantly higher (15 - 20 µg C/L). These elevated levels were almost equally due to an increase in summer and winter nanoflagellate stocks (Hickel et al. 1997, p. 47). A differentiation of the long-term flagellate development in summer into different salinity ranges revealed that the biomass increase was most pronounced at lower salinities of 29 (Ladwig et al. submitted). In contrast to the flagellate evolution, a more or less fluctuating diatom carbon concentration with single peak values of annual medians up to 20 µg C/L was observed, but without any clear trend.

The oscillation in the annual phytoplankton development exhibits an approximately 3 - year cycle (Radach and Berg 1986, p. 31). The cycle of flagellate biomass appears to be inverse to that of diatoms. This may indicate a conditioning of the flagellate development by the preceding diatom bloom, with low spring blooms resulting in high consecutive flagellate blooms due to higher levels of residual N and P left after a largely Si-limited diatom spring bloom. There were no indications anymore of such a phase-delay of functional phytoplankton blooms after the massive flagellate increase at the end of the 1970s.

It is widely assumed that silicate availability is important in determining the extent and breakdown of the diatom spring bloom in the German Bight. Silicate imports are not directly related to man-made eutrophication, but may be influenced by land use and aquatic engineering in the watershed of the rivers, such as the construction of artificial barriers and reservoirs as well as the extension of arable land and soil acidity (e.g. Conley et al. 2000; Humborg et al. 2000). Reduction in Si discharge by rivers may have significant implications for planktonic foodweb structure in the coastal zone, since diatoms are replaced by species not requiring silicate for growth. Since the beginning of routine silicate measurements near Helgoland in 1966 silicate winter concentrations have decreased in coastal water masses with lower salinities (Radach and Berg 1986). This process, together with changing N and P loading may have affected the composition of functional phytoplankton groups in the German Bight.

From mesocosm experiments with natural phytoplankton communities of Dutch coastal waters it was concluded that P-reduction favours the dominance of diatoms over flagellates. By contrast, at high P-loads diatoms were limited by Si and flagellates become dominant (Smaal and Peeters 1995). The same effect on species composition was observed when N-loads were reduced, while both primary production and biomass (chlorophyll-a) decreased proportionally with decreasing N- and P-loads.

It is thus supposed that the concomitant steep increase in riverine N and P discharges in the 1970s (see chapter 2.3) may have caused an elevation of residual N and P concentrations after the diatom spring bloom up to levels where light limitation for flagellate growth predominates. However, this hypothesis has to be tested in future with further investigations focusing on nutrient turnover, including aspects of the seasonal sedimentation and remineralisation cycles, as well as the role of light limitation. There are indications that large-scale environmental changes related to the great salinity anomaly at the end of the 1970s interact with the increase in nutrient loads, causing a change in the community structures of the North Sea.

The period 1976 - 1981 was characterised by decreasing salinities, which coincides with the observed shift in species composition at Helgoland Roads (Radach and Berg 1986, p. 28). In addition, for a 23-year period of investigation (1962 - 1984), the authors found an increasing trend in water temperature of 2.17 ° C , which could also promote a modification in species composition due to different optimum growth temperatures or changes in biogeochemical processes. It is thus difficult to attribute the observed changes in phytoplankton species composition to one causative factor only.

Similar shifts in phytoplankton species composition to those at Helgoland Roads were also observed at the western Wadden Sea long-term series at Marsdiep (Netherlands), showing a proliferation of non-diatom species in the second half of the 1970s. The duration of the bloom season of non-diatoms increased markedly (Cadée 1992) and flagellate biomass increased as well (Philippart et al. 2000). Apart from these effects, there was a change in species composition towards large forms or colony forming species with higher sinking velocities. These changes coincided with different levels in nutrient ratios (N:P) and therefore imply a strong causal relationship with eutrophication (ibid.).

There are sound indications of an increase of frequency and duration of *Phaeocystis* blooms along the Dutch coast (Cadée and Hegeman 1986, p. 29) and, consequently, of changes in species composition including zooplankton. Changes of phytoplankton composition due to eutrophication have been discussed in several reports, such as the *Phaeocystis* dominance during the post-spring-bloom-season (Zevenboom 1994; Riegman 1994). Data of the Continuous Plankton Recorder (Reid 1978) revealed a change of the community structure in the North Sea phytoplankton during the period from 1958 to 1973, showing a general increase in season length and phytoplankton colour intensity, as well as a decline in diatoms. Consequently an increase in the abundance of microflagellates was supposed.

In spite of obviously decreasing phosphate concentrations in winter (Fig. 2.3.3.1); flagellate biomass in the German Bight is still persisting at a high level (Fig. 2.5.1.1). A possible explanation for this apparent inconsistency may be that phosphorus limitation is only of minor importance for flagellate development in the region and that grazing pressure has declined because of the predominance of less edible forms. It has been hypothesised that eutrophication leads to prey-predator mismatches with observed changes in phytoplankton dynamics which are primarily the result of failure of grazing processes and secondarily a result of nitrification (Smayda, pers. comm.).

Moreover, the change in N/P ratios in coastal waters due to the recent P-reduction is supposed to cause in a larger area of the North Sea a shift towards slow growing algae adapted to low phosphate concentrations, including harmful and toxic species (Skjoldal 1992). It was hypothesised that *Chrysochromulina* type algae are usually living in association with organic marine snow aggregates which presumably are P limited microenvironments.

Neophyta

At the level of species composition, there have been modifications due to the intrusion of allochthonous phytoplankton forms (neophyta). The Indopacific diatom species *Coscinodiscus wailesii*, which was found in European waters in 1976 (Boalch and Harbour 1977) because of its nuisance effects (mucus formation) on fishing, was observed in the German Bight since 1984 (Hesse 1988). This large species is now well established in the region and may share a substantial part of the spring phytoplankton community in the German Bight. In spite of its size it may even grow under conditions of water column stratification and low Si supply during early summer (Hesse et al. 1995a). Besides, there has been increasing evidence for the recent intrusion of other non-indigenous phytoplankton species, among these several toxic forms (*Chattonella* spec., *Fibrocapsa japonica*, *Heterosigma akashiwo*), which contribute to the total number of potentially harmful species in German coastal waters. These species may constitute a permanent risk to aquaculture, fisheries, tourism and the marine biota in case of occasional mass development. The brevetoxin-like toxin of *Fibrocapsa* has been found in high concentrations in the tissue of dead seals at the German West Coast (Siebert unpubl.).

The import of these species has *per se* nothing to do with eutrophication, but the establishment of non-indigenous flagellates can be enhanced by high N- and P-concentrations in relation to silicate.

2.5.2 Bloom events

Increased intensity and frequency of phytoplankton blooms are usually considered to be one symptom of eutrophication. For the purpose of subdividing a region in trophic categories, it is therefore useful to evaluate the geographical occurrence of elevated phytoplankton biomass, indicating bloom situations. However, the term 'phytoplankton bloom' is not well defined. The occurrence and duration of a 'bloom' is not always directly related to enhanced phytoplankton production and may in some cases constitute a merely passive accumulation of cells. There are some agreements to consider phytoplankton occurrence as a bloom when it amounts to an abundance of $> 1*10^6$ cells/L (Livingston 2001, p. 39). However, this designation is somewhat subjective and depends on the natural characteristics of an individual species, its size and production rate. Some of the older literature used water discolouration as a definition of an algal bloom. Another definition considers high numbers of cells concomitant with low species diversity (Fryxell and Villac 1999, p. 420). Biomass thresholds for blooms due to anthropogenic nutrient enrichment have always to be adjusted according to the shape of the natural annual phytoplankton cycle in the specific region. Such exceptional blooms frequently occur in German Bight waters. However, also in earlier times, exceptional phytoplankton blooms were occasionally observed in the region e.g. for *Coscinodiscus concinnus* (Savage and Wimpenny 1936), *Phaeocystis* spp. (Wulff 1935) and the phototrophic ciliate *Mesodinium rubrum* (Lohmann 1908). Red tides of the heterotrophic dinoflagellate *Noctiluca scintillans* in the coastal North Sea were already described by Aurivillius (1878), cited by Ostenfeld (1913).

Zevenboom (1994) gave a compilation of maps based on field observations of plankton blooms in the North Sea. The author sampled distribution maps from several sources and produced an integrated overview for each of the investigated plankton species. Figures 2.5.2.1 - 2.5.2.6 show a cutaway for the German territorial and adjacent waters, because by transboundary transports the German Bight will import nutrients and plankton from southern areas and export this material after partial conversion, which is controlled e.g. by the actual residence time (see tab. 2.1.2.1), to northern coastal waters.

The following species are suggested to be most important for eutrophication effects in the German Bight (after Zevenboom 1994):

Gymnodinium mikimotoi

The fish-killing dinoflagellate *Gymnodinium mikimotoi* (former *Gyrodinium aureolum*) occurred in toxic concentrations in the inner German Bight. Increasing extension of the blooms is supposed to be linked to increased eutrophication (Fig. 2.5.2.1).

A mass bloom of the toxic dinoflagellate *G. mikimotoi* (*G. aureolum*) was observed in 1968 near Helgoland (Hickel et al. 1971). In the same year this species caused fish mortality off the Danish west coast. Concentrations of about 360 Mio. cells/L were observed.

Dinophysis acuminata

The German coast is also affected by blooms of the shellfish-poisoning dinoflagellate *Dinophysis acuminata* and other potentially toxic *Dinophysis* species. *D. acuminata* shows a widespread occurrence in German territorial waters (Fig. 2.5.2.2).

It is important to note that all these observed blooms which do encompass toxic species are suggested to be linked to eutrophication, although the author notes that climatic influences, meteorology and hydrography cannot be dismissed (Zevenboom 1994).

Noctiluca scintillans

In spite of the fact that the heterotrophic dinoflagellate *Noctiluca scintillans* has not to be considered as a phytoplankton organism, distribution and abundance of this form are pertinent indicators for the trophic status of the planktonic food web since the diet of this protist is mainly made up by phytoplankton and it represents a key species in the German Bight.

High abundances of *N. scintillans* are a regular phenomenon in the coastal waters of the German Bight (Fig. 2.5.2.3, Tab. 2.5.2.1). Maximum cell concentrations usually exceed 1000 cells/L (resp. ~204 mg C/m^3) almost every year, peaking preferably in July. During 'red tides', a common phenomenon in the German Bight, surface concentrations may rise up to 70,000 cells/L (Nehring et al. 1995). The development is supposed to be linked to day-length and stimulated by the food availability and the ambient temperature and salinity conditions (Uhlig et al. 1995, p. 29). Correspondingly, abundances are usually much higher in the coastal areas than in the open sea.

Tab. 2.5.2.1: Compilation of exceptional plankton blooms* in the German Bight

SPECIES	YEAR	CONCENTRATION	REFERENCE
Ceratium furca	Aug 1981	> 0.5x10^6 cells/L, resp. max. 3.5 mg PPC/L	Hickel 1983
	Aug 1987	> 0.1x10^6 cells/L	von Westernhagen et al. 1989
	Jul 1993	> 1x10^6 cells/L	Hagmeier et al. 1995
Ceratium fusus	Jul 1982	dense stocks	Hagmeier pers. comm.
	Aug 1984	> 1x10^6 cells/L	Hagmeier 1985
Coscinodiscus concinnus	Apr 1985	max. 0.35x10^6 cells/L	Gerlach 1990
	Mar 1996	twice as usual	Hickel et al. 1997
Coscinodiscus wailesii	May 1984		von Westernhagen et al. 1987
	1991	480 cells/L**	Hagmeier 1992
	1993	650 cells/L	Hagmeier et al. 1995
	1994	2,000 cells/L	Hagmeier et al. 1995
Pseudonitzschia pungens	1992	max. 1.7x10^6 cells/L	Hagmeier 1992
Gymnodinium mikimotoi	Aug 1968	max. 3x10^6 cells/L	Hickel et al. 1971
	Aug 1979	max. 1x10^6 cells/L	Doerffer and Amann 1984
	Jul 1993	max. 124,000 cells/L	Hagmeier et al. 1995
Gymnodinium chlorophorum	Aug 1990	100x10^6 cells/L	Hagmeier 1991
		1x10^6 cells/L	Nehring et al. 1995
Dinophysis norvegica	Aug 1987	> 10,000 cells/L	von Westernhagen et al. 1989
Dinophysis acuminata	Aug 1984	max. 0.16x10^6 cells/L	Nehring et al. 1995
Fibrocapsa japonica	Jul 1997	0.3x10^6 cells/L	Rademaker et al. 1998
Chatonella verruculosa	May 1998	high concentrations	Göbel pers. com.
Prorocentrum triestinum	1993	max. 51,000 cells/L	Hagmeier et al. 1995
	1994	max. 30,000 cells/L	Hagmeier et al. 1995
Phaeocystis globosa	regularly	max. 200x10^6 cells/L	Hanslik et al. 1998, Peperzak 2002
Myrionecta rubra	Jun 1984	0.06 x10^6 cells/L	Hagmeier 1985
	Jul 1994	> 3x10^6 cells/L	Hagmeier et al. 1995
Noctiluca scintillans	regularly	max. 70,000 cells/L	Uhlig and Sahling 1990, Uhlig et al. 1995

*Following an ICES definition (International Council for the Exploration of the Sea) of 1984, exceptional blooms are those plankton blooms which differ from other blooms in terms of their (direct) noticeability or their immediate consequences.

**A cell number of 480 cells/L is adequate to a C concentration of 1.8 mgC/L if a cell diameter of 300 µm (Tomas 1996) and an intracellular C concentration of 54 fg/µm³ is used for calculation (Moal et al. 1987)

Synoptical surveys carried out in the German Bight in summer showed maximum abundance of *N. scintillans* near the East and North Frisian Wadden Sea, often with higher cell concentrations in the bottom water. The start of a 'bloom' in spring is closely connected to relatively small reproductive centres close to the Wadden Sea (Uhlig and Sahling 1990, p. 114; Nehring et al. 1995, p. 200). Spreading of the bloom is then dependent on wind and current patterns. In August, when cell division is interrupted, population centres drift from coastal waters further offshore. At the same time feeding stops and cells accumulate at the sea surface which can lead to the water discolouration mentioned above (Uhlig and Sahling 1990, p. 114). Decomposition of *Noctiluca* 'red tides' may lead to the development of a divers microbial community, involving bacteria, fungi, amoeba and ciliates and to a pronounced oxygen consumption. Mass kills of cockles resulting from oxygen consuming microbial degradation of the bloom have been observed. In addition, some authors attribute ammonium toxicity to the species.

Noctiluca is known to be a non-selective, omnivorous, but very voracious feeder, which may be able to consume up to 10 % of the total organic suspended matter per day during springtime (Hanslik 1987). In turn, it hardly serves as food for other marine organisms. Since organic debris plays a major role in *Noctiluca* feeding, Uhlig and Sahling (1990, p. 115) found no direct correlation between *Noctiluca* occurrence and phyto- or zooplankton abundance or cell densities of other microplankton populations such as ciliates, for example. Therefore, it may be appropriate to consider *Noctiluca* occurrence as an indicator for the eutrophication status in the German Bight. However, trends in cell concentrations of *Noctiluca scintillans* cannot be observed from the Helgoland roads time series 1968 - 1988 (ibid., p. 104).

Besides the contemplation mentioned above, there is further information on plankton blooms in the German territorial waters which were attributed to eutrophication:

Ceratium furca

In August 1981 a bloom of the dinoflagellate *Ceratium furca* led to noticeable discolouration (“red tide”) in parts of the German Bight. Quantitative investigations of Hickel (1983) revealed cell numbers up to over 500,000 cells/L in the upper layer (10 m) (Fig. 2.5.2.4).

In August 1984 the dinoflagellate *Ceratium fusus* reached a density distribution of more than 1 Mio. cells/L (surface water) west of Helgoland (Hagmeier 1985) (Fig. 2.5.2.5). At the same time, an extended red tide of *Noctiluca scintillans* with biomass concentrations of more than 4 mg C/L occurred in the surface waters of the outer German Bight (Hesse et al. 1989 a; Schaumann and Hesse 1990).

Gymnodinium chloroforum

A mass development of the previously unknown dinoflagellate *Gymnodinium chloroforum* occurred in August 1990 (SYNDWAT 1992). Mean concentrations of up to more than 1 Mio. cells/L were detected near Helgoland (Nehring et al. 1995). Hagmeier (1991) observed this species from July to September 1990 in surface samples of the Helgoland Roads time series. It reached maximal concentrations of 100 Mio. cells/L, giving the water an intensive green discolouration and light green foam formation. During calm weather periods mucilage was formed. There are indications that the bloom extended much further seawards than observed.

A compilation of exceptional bloom occurrences in the area of the North Frisian Wadden Sea was performed by Nehring et al. (1995) for the period 1989 to 1992, showing recurrent mass-developments of *Phaeocystis globosa* and *Noctiluca scintillans*, as well as mass occurrence with peak densities of 158,000 cells/L of the toxic dinoflagellate *Dinophysis acuminata* at a density inhomogenity west of Helgoland in August 1984.

Effects of eutrophication resulting in phytoplankton increase and extraordinary blooms in the German Bight have been compiled by Gerlach (1990). Besides the blooms already mentioned above, the author draws attention to the elevated levels (more than 200 µg C/L) of phytoplankton carbon in the summer of the years 1982, 1983 and 1986 near Helgoland roads. In 1985 exceptionally high spring concentrations (more than 300 µg C/L) of phytoplankton occurred, which were due to a mass development of the centric diatom *Coscinodiscus concinnus*. A massive development of this species off the East Frisian Islands in 1996 accompanied by lipid formation is supposed to have led to oxygen deficiency and subsequent formation of 'Black Spots' at the sediment surface of the adjacent Wadden Sea (Hickel et al. 1997, p. 22).

A compilation of exceptional events in the region under study is presented in table 2.5.2.1 However, due to reduced observation density, data for the second half of the 1990s is sparse.

2.5.3 *Phaeocystis*

The prymnesiophyte *Phaeocystis* spp. shows an increase in concentration, frequency and duration of blooms in German coastal zones over the last 5 to 15 years which the author (Zevenboom 1994) attributed to increased eutrophication without further information on the statistical background of this relation (Fig. 2.5.2.6).

Hanslik et al. (1998) analysed the occurrence of *Phaeocystis* blooms on the basis of the Norderney long-term data set covering the period from 1982 - 1991. Although a statistically significant trend in bloom density of *Phaeocystis* could not be demonstrated, the densest *Phaeocystis* blooms, with > 40 - 50·10³ colonies/L, were observed in the last years of the investigated period (ibid., p. 17). However, the authors stressed that there are different theories for an explanation of this observation, some being related to eutrophication, some not (Hanslik et al. 1998, p. 19). Therefore a clear statement about a cause-effect relationship cannot yet be made. Within this context, it should be considered that the life-cycle of *Phaeocystis*, which strongly affects the bloom dynamics, appears to be very complicated. From recent findings there are indications of sexual reproduction and gamete formation. Moreover, the importance of CO_2 limitation on cell growth in large colonies was recently elucidated by Peperzak (2002, pp. 182-189).

2.6 Increased sedimentation of biomass

Increased sedimentation of pelagic organic material was found to enhance the benthos production in such an amount that population shifts occurred (Josefson et al. 1993). However, in shallow areas it is not possible to measure sedimentation rates properly. For certain parts of the Wadden Sea, an increase in particulate organic matter input has been claimed since the early 1950s (see above). In the East Frisian area of the Wadden Sea, there was evidence for a decrease in the sedimentation of fine-grained material as a result of dyking and land reclamation (Delafontaine and Flemming 1999). Therefore, in this area the tidal flats may act no longer as a sink for the imported particulate organic material which will now either be decomposed in the water column or leave this area by tidal advection, advected by the coastal current to north-eastern tidal flats.

2.7 Increased oxygen consumption and oxygen depletion in bottom water and in sediment layers

In connection with long distance imports and transient stratification, the German Bight is considered to be an area sensitive to eutrophication (Nauta et al. 1992).

In North Sea waters the oxygen saturation concentration varies naturally between 7.5 and 12.0 mg O_2/L (Buchwald 1990, p. 332). Oxygen saturation concentration depends on temperature and salinity and is a non-conservative parameter of a water body (Grasshoff 1976, p. 60). Under balanced conditions oxygen supply into the water column is ensured by water turbulence. In the German Bight, oxygen deficiency is mainly a result of oxygen consumption during the microbial degradation of organic matter. Periods of calm weather and stratification of the water column promote this deficiency by hindering the oxygen transport into the bottom layer.

Longer periods with values below 2 mg O_2/L are found to be critical for the bottom fauna (Rosenberg 1980) and will lead to death of most animals unable to escape (Christensen 1998, p. 76). For most higher marine organisms 3.0 mg O_2/L is the minimum oxygen concentration (Azarovitz et al. 1979). Many fish and benthic invertebrates are affected at values below 4.0 mg O_2/L already, and will attempt to escape (Christensen 1998, p. 76). Even a reduced oxygen content below 60 % saturation may exert stress on the fish fauna. For example, some marine fishes leave areas with oxygen contents from 2.5 – 4.6 mg O_2/L (Niermann and Bauerfeind 1990, p. 72).

At the same time, stratification may also favour the formation of algal blooms by reducing the critical depth. This holds especially true for those parts of the German Bight where a steady input of nutrients occurs through the influx of less denser river water into the upper layer of thermally stratified areas in spring and summer. Typical for this constellation was the situation in the summer of 1981:

Due to a longer period of calm weather, large phytoplankton blooms were formed in the stratified inner area of the German Bight, succeeded by oxygen deficiency, an event which had not previously been detected (NSTF 1993). The oxygen content in the bottom water layer was reduced to less than 4 mg O_2/L in wide areas of the German Bight and the waters off Jutland. At some stations the oxygen content was reduced below 2 mg O_2/L (Fig. 2.7.1). In the following two years the previous large-scale oxygen deficiencies were observed again. Dethlefsen and von Westernhagen investigated an area of 16,500 km² in the German Bight and the waters off Jutland in 1982. 60 % of the area showed reduced oxygen content (< 4 ml O_2/L) and a considerable part even values below 2 ml O_2/L (Dethlefsen and von Westernhagen 1982, p. 182). Due to the transboundary transport of nutrients and organic matter within the coastal current (Otto et al. 1990), Danish coastal waters are affected by material imported from the German Bight which in turn receives material from southern areas.

Mortalities of fish and benthic invertebrates were observed during the years 1981 - 1983 as revealed by dredges, underwater photography and television. Figure 2.7.2 shows the areas of fish mortalities occurring in September 1981. They were mainly located west of Horns Rev and coincided approximately with those parts where the oxygen content fell below 4 mg O_2/L (Fig. 2.7.3). A similar situation was observed in August 1982. However, the detected fish mortalities were located closer to the coast of Jutland (Fig. 2.7.4), again in those areas with an oxygen content below 4 mg O_2/L (Fig. 2.7.3). On the 1983 survey (Fig. 2.7.5) the lowest oxygen concentration up to the present was observed. It was followed by a marked reduction of species and individual numbers of the macrobenthos in the areas particularly affected (Fig. 2.7.6) (Hickel et al. 1989, p. 57).

In 1989, oxygen deficiency was observed again from September until October. The affected area west of Horns Rev (Fig. 2.7.7) was smaller compared with the large-scale oxygen deficiencies in 1981-1983 but its consequences were similar: Oxygen values dropped below 2 mg O_2/L, fish disappeared and the benthic fauna partly suffered heavy losses (Niermann 1990, p. 1).

Investigations during the KUSTOS project from July to August 1994 revealed low oxygen concentrations in the inner German Bight west of the North Frisian Wadden Sea (Brockmann, pers. comm.).

Example of a development of oxygen depletion

Oxygen concentrations (mg/L) in the bottom water of the German Bight were analysed during 4 succeeding grid samplings in July/August 1994 (Fig. 2.7.8). Lowest oxygen was found along the Elbe valley as reported before (Brockmann and Eberlein 1986). From the successive grid investigations it can be seen that oxygen depletion increased during the time of investigation. The minimum dropped from < 4 mg/L to < 2 mg/L. Also the area with oxygen concentrations below 7 mg/L increased. This is the first reported case study where the development of oxygen depletion in the German Bight can be followed.

The vertical east-west transects during the last grid sampling show that the oxygen depletion was not restricted to the sediment surface, but affected the whole bottom water near the front between vertically mixed and stratified water (Fig. 2.7.9).

This area was not covered by the spreading river plume water (Brockmann et al. 1999 a, b) but bordered this area (e.g. Fig. 2.3.7.7). It can be assumed that in spite of relatively low nutrient concentrations the net primary production was enhanced by permanent nutrient provision to the stratified area above the Elbe valley. By these factors (nutrient supply and inhibition of phytoplankton sedimentation by stratification) ongoing primary production and successive sedimentation of phytoplankton, cut off from the nutrient source by the spreading water masses, would cause oxygen depletion by remineralisation in the stagnant bottom water. Thermal stratification enables the formation of the oxygen depletion in the enclosed bottom water, as is evident from the vertical profiles of temperature and oxygen concentration (Fig. 2.7.9). Additionally, accumulation of particulate material along the front may enhance this severe eutrophication process.

Stratified areas adjacent to the river plume would generally trap biomass, causing significant eutrophication effects. This is, to some degree, a natural process and it has to be further elaborated to which extent this process is influenced anthropogenically. Stratification, accumulation and long residence times cannot be modified anthropogenically, only the load of nutrients in the river plume. Since low nutrient concentrations can also cause this effect during long lasting stratification and trapping of particulate material at low exchange rates of water masses, transboundary transports have to be considered as well as remobilisation of nutrients which are seasonally trapped by onshore transports of particulate matter (Postma 1984).

2.8 Eutrophication induced changes in benthic biomass and species composition

2.8.1 Macrophytes

The investigations of Reinke in 1889 already elaborated that macroalgal distribution in the open German Bight (excluding the Wadden Sea) is mostly restricted to the rocky island of Helgoland (cited by Lüning and Asmus 1990, in: Lozán et al. 1990 eds.). For this reason possible eutrophication effects on macroalgae are only locally of quantitative importance in this region. However, changes in macroalgae development at Helgoland which are due to eutrophication may be used as an indicator for the trophic situation in a wide area of the German Bight, since this site is under variable influences of different water masses and hence has an integrating character with respect to space and time.

Kornmann and Sahling (1994) compared recent data on the macroalgal flora of Helgoland with investigations of the last century carried out by Kuckuck (1894, 1897; cited in: Kornmann and Sahling , p. 365) and assumed an increase of macroalgae biomass at this site. Especially green algae such as *Ulva* spec. were affected. Kornmann and Sahling found thalli of *Ulva* with up to 5 m in length (ibid., p. 367), whereas this could not be observed by Kuckuck in the past century. Although these historic investigations were at the most semi-quantitative, Kornmann and Sahling attributed this increased abundance of macroalgae, esp. of green algae at Helgoland, in comparison to the 1890's to eutrophication in conjunction with the possible effect of climatic variances and changes in settling substrates due to human activities. Also, the massive occurrence of green algae in the German Wadden Sea is considered to be a consequence of increased nutrient loads (Siebert and Reise 1997, p. 51).

In the Dutch coastal zone, there is evidence for changes in the sublittoral vegetation. Off the island of Texel seagrass beds vanished since 1932 due to increased turbidity (van den Hoek et al. 1979, In: Gerlach 1990, p. 103) which is probably a result of enhanced phytoplankton growth.

2.8.2 Macrozoobenthos

2.8.2.1 Available data

One of the first studies on the quantitative distribution of zoobenthic communities in the German Bight was carried out by Hagmeier (1925) in 1923/1924. Later, comprehensive macrozoobenthos studies were carried out repeatedly in 1966 (Stripp 1969), 1975 (Salzwedel et al. 1985), 1984 (Büsselberg 1984; Rachor unpubl.), 1995 (Thatje and Gerdes 1997) and 2000 (Rachor pers. comm.). Since 1969, almost yearly macrozoobenthos surveys were implemented at 4 stations in the German Bight, covering sandy, muddy sand and muddy substrates and a reference station further offshore west of 'Weiße Bank' (6° E 55° N). Another long-term series for macrozoobenthos was established at a site off the island of Norderney in 1978 (Kröncke et al. 1997; 1998). Furthermore, a number of other benthic studies exist e.g. for the eastern part of the German Bight (i.a. Dörjes 1968; Ziegelmeier 1978).

2.8.2.2 Eutrophication effects

In general, the response of macrozoobenthos development to increasing input of organic matter is not linear, but follows a bell-shaped curve, which was described by the model of Pearson and Rosenberg (1978, p. 234): In a first stage of moderate eutrophication macrozoobenthic biomass increases due to enhanced organic matter. If the organic load continues to increase, harmful effects such as oxygen deficiency and related stress factors may occur, and macrozoobenthic biomass will consequently decrease.

In this context it is notable that, according to recent findings, also high carbon dioxide concentrations in the water (hypercapnia), which may already occur under moderately hypoxic conditions, as well as acidification may affect benthic organisms by impairing the immune system (cited in Hesse 2000).

Increased biomass

Macrozoobenthos data may be more suitable for evaluating long-term eutrophication effects compared to those of short-lived phytoplankton species, because of the macrozoobenthos' more stationary way of life and the potential to integrate chronic stress and enrichment situations. For the same reason, however, pronounced difficulties arise in differentiating between eutrophication induced effects and other (anthropogenic) impacts, especially from bottom trawl fishery, but also from contaminants, changes in sediment morphology, as well as meteorological variability.

Despite the high variability in zoobenthic stocks, an overall increase in sublittoral macrozoobenthos biomass can be assumed for different areas of the German Bight from the existing data base. Rachor (1990) compared investigations carried out in the German Bight in 1923/1924, 1966, 1975 and 1984 (see chapter 2.8.2.1) and found an overall increase in macrozoobenthic biomass of more than 50 % compared to the situation in 1923/1924. Maximum increases of up to 400 % were observed on sandy bottom habitats in the 1970s (ibid., p. 211) (Fig. 2.8.2.2.1). The author attributed these changes largely to eutrophication.

The results for the German Bight are in accordance with findings of Kröncke (1995, p. 75) and others. Kröncke investigated several areas of the North Sea and reviewed the relevant literature on long-term changes in the benthos. The author reported an overall increase in macrozoobenthic biomass by a factor of 2.5 - 4 in coastal regions and in the open sea compared to previous investigations. Taking into account the adverse effects of intensified fishing activities on benthic stocks (repeated bottom trawls in the same area 3 - 4 times per year in large parts of the North Sea), there is evidence that the increase is related to the eutrophication process (ibid., p. 76), and would even have been larger without the impact of fishery. More recent findings emphasise the coincident effect of eutrophication and changing meteorological conditions (see below).

Changed species composition

Beukema and Cadée (1997, p. 1424) underlined the impact of ambient abiotic factors on the benthos development, e.g. current patterns and sediment structure, in the tidal areas of the western Dutch Wadden Sea. It is evident that benthic populations can only react adequately to an enhanced food supply when abiotic factors allow it, for instance when the sediment transport is not too strong for successful settlement.

Especially the variability in the benthic species composition can be very sensitive to changing environmental factors unrelated to eutrophication. In a comparative approach on the mesoscale, it might be appropriate to evaluate the eutrophication status by means of a species diversity index, with eutrophic areas being characterised by a few, opportunistic species of small-sized individuals, but in high abundance. In contrast, regions which are not subject to eutrophication have a broad spectrum of larger species with lower numbers of individuals. As Margalef (1997) pointed out, the diversity index should be applied carefully with respect to pollution.

Macrozoobenthic communities of the German Bight did not exhibit pronounced shifts in species composition since the beginning of quantitative investigations in 1923, but numerical dominance changed within the period 1923 - 1984 in favour of adaptive, short-lived and highly productive species (Rachor 1990, p. 214).

Hypoxia & Macrozoobenthos

Oxygen deficiency in the bottom water is considered to be one of the most negative effects of eutrophication on macrozoobenthic communities. Hypoxia has been observed recurrently (see above) at several locations in the German Bight (Rachor and Albrecht 1983; Brockmann and Eberlein 1986; von Westernhagen et al. 1989; Niermann 1990; Brockmann pers. comm.). Sometimes it was combined with mass mortalities of benthic fauna. Although the macrozoobenthos community can be decimated by hypoxia, resettlement and recovery can take place within a few years in the open German Bight system. After the 1981 - 1983 hypoxia events, the benthic communities in the sandy oxygen deficiency areas of the northern German Bight recovered well in the following years (1984-1986), showing similar diversity, biomass, species, and individual numbers as before (Niermann et al. 1990).

2.8.2.3 Indicator species

In analogy to the biological system of freshwater quality, the occurrence or lack of certain species may be considered as an assessment criterion of the eutrophication status. Among these indicator species Echinodermata are of predominant importance. *Echinocardium cordatum* benefits from enhanced food supply due to sinking phytoplankton cells. Kröncke (1992) described a 7- 15-fold biomass increase of this species due to better food conditions at the southern Dogger Bank over the period 1950/54 to 1985/87. In contrast, a decrease of *E. cordatum* was observed on muddy sediments over the period 1966 - 1984 (Rachor 1990). These sediments are generally characterised by high sedimentation rates of fine organic matter and exhibit elevated oxygen consumption rates. The observed decrease of *E. cordatum* stocks is certainly due to the fact that Echinodermata are hyper-sensitive to reduced oxygen concentrations at the bottom (Rachor 1982, p. 133; 1998, p. 299). The population dynamics of the indicator species *E. cordatum* thus fits well with the Pearson and Rosenberg (1978) model described above (chapter 2.8.2.2).

Other indicator species under discussion are e.g. the polychaetes *Capitella capitata*, *Scoloplos armiger* and *Heteromastus filiformis*. These species, in turn, are generally tolerant to hypoxic conditions and are thus among the few species that survived during serious oxygen deficiency events (Diaz and Rosenberg 1995, p. 262). Heip (1995, p. 123) compiled results on *C. capitata* population dynamics which show that even for this rather well studied species generalisations with respect to the eutrophication problem should be made cautiously: even in the absence of competitors, *Capitella* populations naturally oscillate between low and high densities. Furthermore, important information on food requirements is missing (ibid. 1995, p. 123) for many other species suspected to profit from eutrophication such as *Amphiura filiformis*, *Polydora* spp. or *Corophium volutator*.

Rachor (1982) points out that especially the value of using increases in opportunistic species as an indicator for a high eutrophication status is questionable. Rather, for specific stress situations, such as hypoxia in the bottom water, decreases in sensitive indicator species can be identified for the inner German Bight (ibid., p.128). Especially the infaunal spoon worm *Echiurus echiurus* and the infaunal bivalve *Nucula nitidosa* are considered as indicator species because of their low tolerance to hypoxia (ibid.). The O_2-sensitive species *Echiurus echiurus* disappeared in muddy habitats of the German Bight from 1971 until 1973 along with recurrent low oxygen conditions (ibid. 1992, p. 134). The decline of *Nucula nitidosa* can also be regarded as an indicator for low oxygen concentrations, especially for a slowly increasing deterioration of oxygen supply in muddy substrates. Densities of this long-lived small bivalve decreased significantly in the German Bight in the period from 1976 to 1985 (Rachor 1990, p. 211), but have recovered since then (Fig. 2.8.2.3.1).

Considering the importance of physical factors for the ecosystem of the German Bight (TUVAS 1993; PRISMA 1994), it should be kept in mind that oxygen deficiency and, consequently, its impact on the zoobenthos community is under stochastic control of largely independent weather events. It is thus widely assumed that at least a 30-year data set is a precondition for differentiating between natural variability and man-made changes in macrozoobenthos. Although the suitability of species as indicators for eutrophication effects is limited, they may be a valuable additional parameter within a more consistent set of information.

2.8.2.4 North Atlantic Oscillation

Changes in the zoobenthos community off the island of Norderney during the past decade were attributed to winter anomalies of the SST (SEA SURFACE TEMPERATURE), respectively to fluctuations of the NAO index (NORTH ATLANTIC OSCILLATION). Elevated temperatures are supposed to affect the biota in conjunction with the eutrophication process in a synergistic way (Kröncke et al. 1997, p. 123, 1998):

Due to mild winters, the natural mortality of the bottom fauna will decrease, hence representing better start conditions for reproduction of most benthic species. In combination with enhanced food supply, mild winter temperatures will thus lead to an increase in abundance and biomass of the macrozoobenthos communities in shallow regions, such as off Norderney

(Kröncke et al. 1997). In this context, it is of note that the observed increasing benthic biomass trends in the central North Sea, Dogger Bank, could not be attributed to changes in SST in the period before 1989 (Kröncke 1992, p. 157). In this case, the interconnection between NAO and macrozoobenthos abundance, or biomass development, respectively, has still to be verified.

2.8.2.5 Regional differentiation

Regional differentiation of macrozoobenthos communities with respect to eutrophication sensitivity can roughly be achieved by means of the sediment distribution. In general, mud areas in the German Bight are impoverished in species numbers. These areas are characterised by the *Nucula nitidosa* association (former *Abra alba* association). Macrozoobenthos communities in muddy habitats (for sediment distribution see Fig. 3.1.1) are explicitly affected by a surplus of organic matter deriving from eutrophication, because these sites are accumulation biotopes in general, being naturally characterised by a high amount of organic matter and reduced diffusion processes. Additional input of organic matter through the eutrophication process thus leads to hypoxia much faster than on sandy substrates. In keeping with this, the long-lasting eutrophication process has led to a decreasing trend in macrozoobenthos biomass on mud habitats in the German Bight (Rachor, pers. comm.). In the transition zone to muddy sand substrates, species exhibiting the Pearson-Rosenberg type response to eutrophication, such as *Echinocardium cordatum* are occurring in high abundance because of a better food supply. At times of severe oxygen deficiency, however, mass mortality of *E. cordatum* was also observed in the German Bight. The adjacent area is characterised by the *Amphiura* – association, which still shows elevated biomass levels.

Thus, as an overall regional differentiation for the German Bight, muddy habitats (very fine substrates in Fig. 3.1.1) are considered to be the most problematic areas with respect to eutrophication, while muddy sand habitats are considered to represent potential problem areas because of their lower sensitivity to eutrophication (Rachor, pers. comm.). As a consequence, there are nearly no non-problem areas in the German Bight with respect to macrozoobenthos.

3. Evaluation of eutrophication in the German Bight

3.1 Subdivision of territorial waters

The following water masses can be found in the German Bight area (including the territorial waters):

(i) **R**iver **P**lume **W**ater (RPW) of the Elbe and the Weser, which are mixed close to the outer coastline, defined by the islands. At high freshwater discharge this plume water can spread further northward than 56^0 (Brockmann and Eberlein 1986; Becker et al. 1999; Brockmann et al. 1999 a, b). Its extension is also controlled by wind forces, which often cause a spreading along the North Frisian coast. This water is characterised by low salinity (< 32) and high nutrient loads, higher temperatures during summer and lower temperatures during winter,

(ii) **W**adden **S**ea **W**ater (WSW), which is frequently exchanged between the coastal area and the tidal flats, with a composition strongly influenced by the specific processes in the Wadden Sea. The physical characteristics are similar to the river plume water with more extreme temperatures. Biogeochemical qualities change seasonally. Both water masses often carry high loads of suspended matter due to tidal action and turbulence in the shallow area (Dick et al. 1999),

(iii) **C**ontinental **C**oastal **W**ater (CCW), which is transported by the residual current moving from the south-west to the north through the Bight under normal weather conditions (dominant westerly winds). The inflow of these water masses into the south-western German Bight can be detected by salinities above 34 and relatively clear water. Generally, the CCW is nutrient-rich during winter (Brockmann et al. 1990; Brockmann and Topcu 2001),

(iv) **C**entral **N**orth **S**ea **W**ater (CNSW), which reaches the outermost areas of the German Bight area. It is characterised by low nutrient concentrations, caused in part by the continuous net primary production in the Dogger Bank area (Brockmann and Wegner 1995; Brockmann and Topcu 2001), and salinities above 35. Between the CCW and CNSW, water from the Atlantic passing the Channel (> 35) and coastal water originating from the southern British coast is mixed in. The latter can contribute higher loads of suspended matter, including particulate phosphorus.

(v) During periods of summer stratification, CCW and CNSW are separated into the warmer, nutrient-poorer mixed layer and colder, nutrient-richer bottom water. The RPW is then mixing mainly with the mixed layer water, while strengthening the stratification through haline gradients and covering the bottom water during its spreading.

In summary, the water masses passing the German Bight area can be grouped into (a) CCW which is fed by the water from the German Bight, including RPW and WSW, and (b) CNSW with its different sources.

It has been shown by several investigations that the listed differentiation very well characterises physical, chemical and biological gradients (Becker et al. 1983; Brockmann and Eberlein 1986; Gerdes and Hesse 1993; Martens and Brockmann 1993; Brockmann et al. 1999a).

The extension of water masses is variable due to fluctuating weather conditions, including seasonal changes. However, the distribution is controlled to some degree by the morphology of the German Bight.

A regional differentiation of the German Bight area, including an assessment concerning the OSPAR criteria for eutrophication (see chapter 3.4), based on the morphology and the distribution of sediments (Salzwedel et al. 1985; Figge 1981), and the described gradients for nutrients and eutrophication effects (oxygen depletion and bloom occurrence) results in the following subareas:

(i) The shallow Wadden Sea (< 10 m) with the estuaries of the rivers Ems, Weser, Elbe and Eider. Due to the estuarine circulation and asymmetric tidal currents suspended matter is trapped and partly accumulated. This, together with high loads of organic material from the passing river plumes and from autochthonous production stimulated by high nutrient loads, triggers decomposition processes (de Jonge and Postma 1974; Hesse et al. 1992);

(ii) The deep Elbe Valley (> 30 m) is characterised up to a longitude of 06^0 E by sediments finer than "very fine sand" including muddy and clay areas (Fig. 3.1.1);

(iii) Towards the coast the wedge of the beginning Elbe Valley is surrounded by sediments of fine to coarse sand, forming valley flanks between 06^0 E and 55.3^0 N. The northern flank covers water depths down to 40 m, the southern only to 30 m;

(iv) The adjacent coastal areas are characterised by fine sand;

(v) In the central part of the German Bight area, with water depths between 40 and 60 m, the sediment surface is formed by fine and very fine sand.

At the transition areas between these sediment regimes different frontal systems have been observed frequently (Krause et al. 1986) (Fig. 3.1.2). The surplus of nitrate in comparison to silicate during winter covers the areas (i - v) which is another indicator for nutrient enrichment (Fig. 3.1.3). The thermal and river plume fronts (westerly, outer front) occur at the border of the Elbe valley and its flanks. Upwelling fronts are observed at the inner end of the valley caused by hydrodynamics forcing an inflow of deep water through the narrowing channel or by easterly winds.

The variable extension of stratified water masses has been analysed statistically by Frey (1990)and shown to reflect also to some degree the morphology of the area. Conditions and genesis of stratification were investigated by Budéus (1989), who also calculated the distribution of the Simpson-Hunter stratification parameter in the area. Only the Wadden Sea and a narrow band of the adjacent coastal water is permanently unstratified. Temporary or even permanent density gradients are registered in all other areas including the Elbe estuary.

Oxygen depletion has been detected in the areas (ii) and (iii), the inner Elbe Valley and its flanks. However, it has to be considered that the area of investigation was often restricted to the inner German Bight and that there is only very little information of the region west of Helgoland (see Figs. 2.7.1 – 5). Enhanced primary production and passive as well as active biomass accumulation were observed at the river plume fronts (Hesse et al. 1989a), which are characterised by thermohaline stratification and a steady supply of new nutrients from the plume. Generally there is only little information on plankton distribution patterns in the German Bight, but the area of the Elbe plume often shows a different species abundance than the south-westerly and the central northern area (Krause et al. 1986; Hesse et al. 1989a, b; Martens and Brockmann 1993; Rick et al. 1998).

It can be concluded that the Elbe valley and its flanks work like a sedimentation trap for the particulate biomass advected into or produced in the vicinity, enhanced by nutrients provided by the river plume, the Wadden Sea and upwelling processes. This biomass accumulation is caused by the morphology and the hydrodynamics and reflected by the sediment structure and the observed oxygen depletion in the bottom water (see chapter 2.7).

Combining the different aspects of regional and water mass differentiation the following subdivision is suggested:

(i) Wadden Sea, characterised by trapping of SPM, dominant decomposition mode, permanent interaction with the sediment and the adjacent coastal water and strong continental influences;

(ii) Inner German Bight (06^0 E, 55.3^0 N), controlled by the Elbe/Weser river plumes, characterised by frontal systems and the sediment trapping Elbe valley and its productive flanks. It receives water from the coastal current, river discharges, from the Wadden Sea and upwelling of central North Sea Water. This area as well as the transient outer area are temporarily stratified causing an uncoupling of processes between mixed and bottom layer.

These two areas are suggested to be "problem areas" concerning its eutrophication;

(iii) Transient area (east of 05^0 E) between German Bight and central North Sea, which is affected by imports from different sources, such as CCW, Channel, British CW and CNSW, suggested as a "potential problem area";

(iv) The outer German Bight area (west of 05^0 E) which is mainly influenced by CNSW, suggested as a "non problem area".

3.2 Time series (> 5 years)

The time series data sets have been compiled in table 3.2.1. However, many of these data sets have significant gaps, some are not directly available because data have not yet been stored electronically or have not been transferred to the DOD or other data centres. Some data sets cannot be combined due to insufficient quality. Some time series are not consistent due to change of methods or had to be reduced significantly due to restricted funding. Important parameters, such as silicate as reference value for a less anthropogenically influenced nutrient, were involved only lately. On the other hand, accumulations and compilations of data sets have been performed by NOWESP (Laane et al. 1996), ICES (1999) and are in progress by MAR_GIS (Schlüter et al. 2002), indicating the ongoing need for data sets.

Information about data centres and time series have been compiled by the SYCON project (Damm et al. 2001). The need for more consistent information has been outlined by SYCON as well (Becker et al. 2001).

Tab. 3.2.1: Time series of physical, chemical and biological data for the German Bight area

Area	Period	Resolution	Parameters	reference	remarks
Estuary of the River Elbe	since 1978	Physical parameters daily, nutrients weekly to biweekly	Q (freshwater discharge), Cl (chlorinity) T (temperature), oxygen, pH, inorg. nutrients (PO_4^-, NH_4^+, NO_2^-, NO_3^-, Si), TP, DOC, TOC, SPM	ARGE Elbe	Profiles from Geesthacht to Scharhörn for the months February, May, June, July, August and November Some changes of methods in 1999
Estuary of the river Weser	since 1979	Physical parameters continuously, nutrients biweekly	Q, T, oxygen, pH, inorg. Nutrients (PO_4^-, NH_4^+, NO_3^-), TP, DOC	ARGE Weser	Profiles in the estuary near the coastline (km 80 to – 4),
Estuary of the river Ems	since 1972	monthly	Inorg. Nutrients (PO_4^-, NO_3^-, NH_4^+), TP, TN	NLWA, Hildesheim	TN from 1980 on and PO_4^- from 1981 on
River Rhine – German part (Station Bimmen)	since 1974	physical parameters continuously, nutrients biweekly	Q, T, oxygen, pH, inorg. Nutrients (PO_4^-, NH_4^+, NO_3^-, Si), TP, DOC	IKSR	
Rivers of Schleswig-Holstein (Eider, Miele, Arlau and Bongsiel)	since 1974		Q, inorg. Nutrients	LANU Schleswig-Holstein, Kiel	
Estuaries of Elbe, Weser, Ems, Lake Ijssel, Nordzeekanal, Nieuwe Waterweg, Haringvliet, Schelde	1977 – 1998	Daily	Daily loads of NH_4^+, NO_3^-, PO_4^-, Si, TN, TP	Lenhart and Pätsch 2001	Nutrient loads based on measurements and modelling data;
German Bight: permanent station Helgoland Roads	since 1962	every workday	T, S, inorg. Nutrients (PO_4^-, NH_4^+, NO_2^-, NO_3^-, Si), phytoplankton quantitatively; occasionally microzooplankton; mesozooplankton (> 150 µm) and macrozooplankton (> 500 µm) every second workday since 1974, occasionally megazooplankton since 1989	AWI/BAH	temperature and salinity daily since 1873; Silicate data 1984-1986 questionable
inner German Bight	since 1980	Monthly	T, S, inorg. diss. Nutrients, phytoplankton composition and biomass	AWI/BAH	transects from Helgoland to Cuxhaven, 6 stations, sometimes several depths
inner German Bight	since 1980	6 times per year	T, S, inorg. diss. Nutrients, phytoplankton composition and biomass	AWI/BAH	transects to the estuaries of Eider and Weser, 6 stations, sometimes several depths, (Weser transect now substituted by transect to station P 8 NW of Helgoland)
German Wadden Sea near Sylt: Lister Ley	since 1981	weekly until 2 times per week	T, S, pH, inorg. diss. nutrients, PON, POC, chl-a, zooplankton samples 150	AWI/BAH	inorganic nutrient analysis performed on unfiltered samples; sampling not al-

Area	Period	Resolution	Parameters	reference	remarks
			µm, since 1993: phytoplankton composition and biomass		ways continuous; bulk of zooplankton samples not yet analysed
German Wadden Sea near Sylt: ferry harbour of List	since 1987	Weekly	net phytoplankton, qualitatively	AWI/BAH	sometimes photo or video documentation
German Wadden Sea near Sylt		1 time per year	grain size, macrozoobenthos quantitatively in the sublittoral	AWI/BAH	focus: trend analysis
German Wadden Sea of Schleswig-Holstein, coastal water of the German Bight	since 1989	fortnightly from April until October	T, S, pH, inorg. diss. nutrients, chl-a, dominant phytoplankton abundance (cells/l)	LANU Schleswig-Holstein (AlgFes program)	in total 13 stations, since 1992 15 stations, surface stations only
German Wadden Sea of Schleswig-Holstein		1 to 2 times per year	T, S, grain size, ash content; pH and salinity in the interstitial water, macrozoobenthos quantitatively	LANU Schleswig-Holstein	focus: trend analysis
North Frisian Coast	since 1999	6 times per year	S, T, nutrients, TN, TP, O_2, SPM Si since 1989	LANU	BMLP - Programme
German Wadden Sea near Norderney	since 1982	since 1985 weekly	T, S, pH, inorg. diss. nutrients, microplankton, TN, TP, Si since 2000	NLÖ, research station Norderney	from 1985 until 1994 surface samples at 2 stations, since 1994 1 station only; a main focus of this program is to get more information about Phaeocystis blooms
German Bight: seaward border of the East Frisian Wadden Sea from the Ems estuary to the Elbe estuary	since 1988	fortnightly	T, S, pH, inorg. Diss. nutrients, microplankton, TN, TP, Si since 2000	NLÖ, research station Norderney	since 1997 reduction of the parameter set to plankton analysis only and sample period only from August to October
East Frisian Wadden Sea of Lower Saxony	since 1989	monthly in the period from April/May until August/ September	Benthic macroalgae semi-quantitatively	NLÖ, research station Norderney	data assessment is done from aboard of a plane; observers classify green macroalgae occurrence into 5 categories
East Frisian Wadden Sea near Norderney	since 1976	4 times per year	Temperature in water, sediment and air, salinity and pH of interstitial water, thickness of the oxidation horizont, water content, grain size, organic carbon, $CaCO_3$, macrozoobenthos quantitatively in the eulittoral	NLÖ, research station Norderney	physical/chemical sediment analysis only in the upper 5 cm layer, grain size analysis only 2 times per year
North Sea	 1984-2000	1 to 2 times per year	T, S, oxygen, fish and fish eggs Nutrients, S,T	BFA-Fi Brockmann and Topcu 2001	focus: biological effects of pollutants

Area	Period	Resolution	Parameters	reference	remarks
	1955-1993 - 1999	monthly different	S, T, nutrients,SPM, Chl a., zoopl. Meta data	Radach et al. 1996 Damm et al. 2001	NOWESP SYKON
Monitoring data German Bight (Überwachung des Meeres)	since 1980		T, S, oxygen, inorg. nutrients (NH_4^+, NO_2^-+ NO_3^-, PO_4^-, Si), TP, TN	DHI/BSH	
German Wadden Sea of Schleswig-Holstein		3 times per year	Spatial distribution of macroalgae	NPA	focus: influence on species composition
Estuaries in the German Wadden Sea		1 time per year	grain size, ash content, macrozoobenthos quantitatively in the sublittoral	BfG	focus: trend analysis and spatial distribution
German Wadden Sea near Büsum	since 1991	weekly from April to September, 1 to 2 times per month from September to March	T, S, Secchi depth, inorg. diss. Nutrients, seston, chl-a, phytoplankton quantitatively; underwater PAR (1992-94); oxygen content, bacteria and zooplankton quantitatively (1995-96); primary production (1992-98)	FTZ	the parameter set was reduced in 1998
Different projects with measurements in the German Bight: ZISCH, PRISMA, TUVAS, KUSTOS, TRANS-WATT, SYNDWAT	Single projects less than 5 years, but in some cases nearly the same stations of different projects together for a longer period		T, S, oxygen, inorg. nutrients, diss. and part. organic parameters, seston, phytoplankton, zooplankton and bacteria	ZMK, BSH	ZISCH (1984-1989); PRISMA (1990-1993); TUVAS (1990-1992); KUSTOS (1994-1997): TRANSWATT (1994-1997); SYNDWAT (1989-1991 and 1992-1993 for data preparation)

3.3 Need for more information

Monitoring data should be supplemented as a rule by salinity and temperature measurements and data of particulate compounds by SPM-measurements. Tidal and/or current conditions should be documented for assessing the SPM data. Intercalibration exercises should be performed more frequently and should be improved also to include biological parameters, e.g. by taxonomic workshops. Sampling and measurements should be harmonised at least in similar environments, such as estuaries: the sampling location in relation to the mean salinity gradient, depth, and sampling time in relation to tidal stage or current speed (Becker et al. 2001). For the interpretation of monitoring data, process data are also needed. Some of the necessary information is listed below:

- The Wadden Sea as a special case due to complex hydrodynamic conditions, its high variability and accumulation of organic material has been described extensively concerning eutrophication within a specific project “Wadden Sea Specific Eutrophication

Criteria" (van Beusekom et al. 2001), but systematic investigations of the exchange processes between sediment and water column are still missing.

- The trapping of particulate matter by estuarine-type circulation (driven by density gradients) and tidal asymmetry in estuaries, tidal flats and the coastal zone has to be investigated systematically by measurements and modelling to understand the accumulation and fate of organic material in this dynamic areas.
- Time series of primary production and the limiting role of nutrients for phytoplankton growth are needed for the interpretation of the dynamics (conversion rates) of standing stocks of phytoplankton, measured as cells, chlorophyll or remote sensing indices.
- The grazing potential of zooplankton is most important for the development of eutrophication effects (Wassmann 1990 a, b; 1998; Wassmann 2001) but is up to now only insufficiently investigated and not included in monitoring programs (Fock et al. 2001).
- Regionally better differentiated flushing rates are needed for the comparison of different subareas and the estimation of the reaction potential of specific tidal basins, including estuaries.
- The analyses of seasonal denitrification rates in the different areas of the German Bight are needed to improve nitrogen budgets (Beddig et al 1997). Measurements by Lohse et al. (1993) indicate high regional and seasonal variability.
- Regular observations on the annual oxygen cycle in the German Bight are needed to detect the development of oxygen depletion in the bottom water and to improve the understanding of this complex process, which is controlled by physical forcing, nutrient discharges, phytoplankton composition, grazing potential and conversion rates.
- Retrospective analysis of remote sensing data relevant to eutrophication could be helpful for the interpretation of observed eutrophication effects. Remote sensing data should be used operationally in combination with ground truth measurements.
- Further development of models with a higher spatial and temporal resolution for the German Bight is essential for a better understanding of eutrophication processes and its management. Operational hydrodynamic models are already being applied, but these must be validated more carefully by measurements to allow the development of realistic ecosystem models on this basis.

3.4 Assessment of the German Bight

3.4.1 General assessment criteria

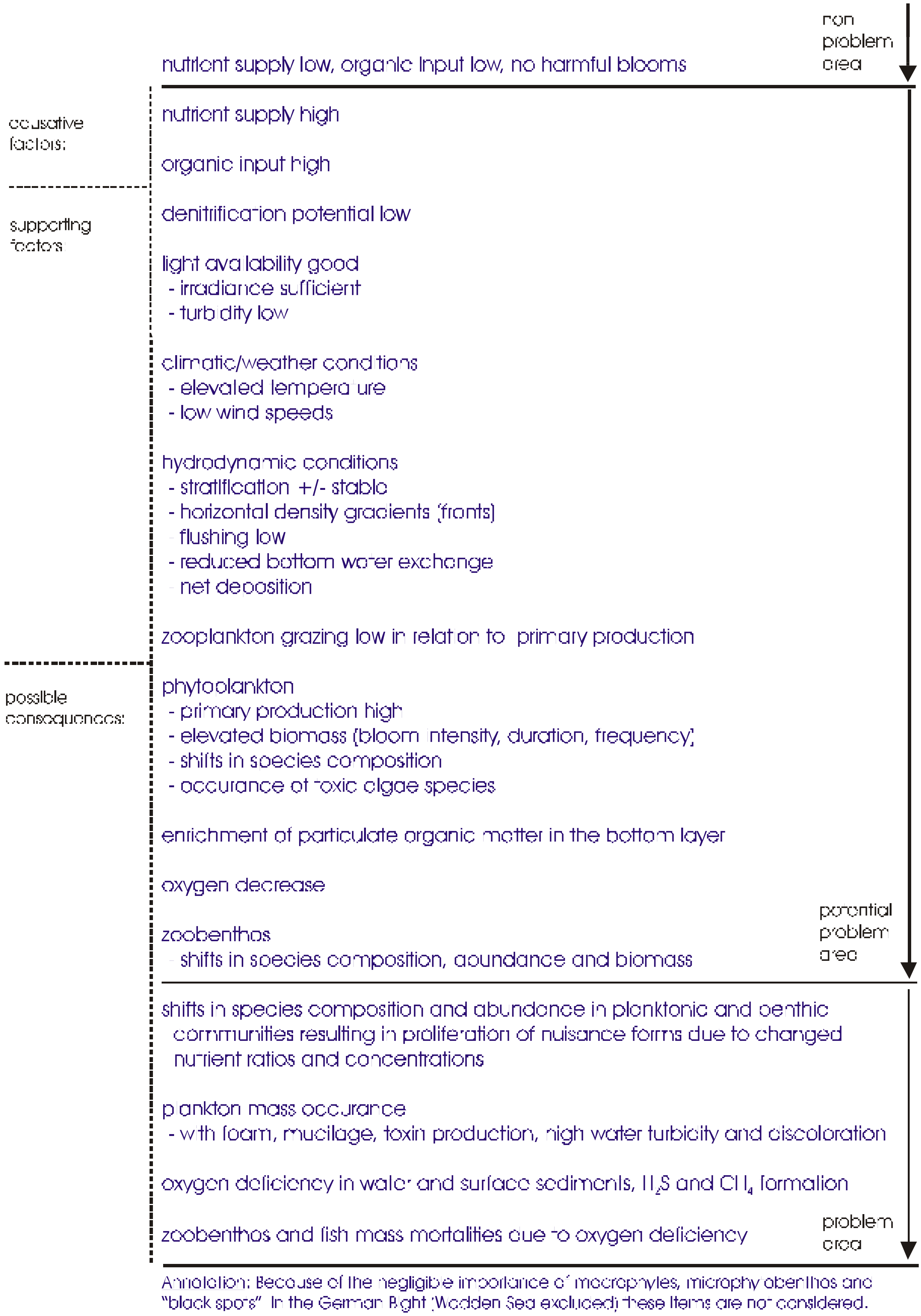
"Undesirable effects resulting from anthropogenic enrichment by nutrients" (OSPAR Strategy to Combat Eutrophication) have to be identified for any assessment aimed at combatting eutrophication. For this reason, natural background concentrations have to be defined in a harmonised way, as well as the whole assessment procedure and observed eutrophication effects have to be analysed for their anthropogenic origin.

In a first approach, several background concentration levels have been compiled by OSPAR (2001). Those for the German Bight have been included in the following compilation (chapter 3.4.2). In the regionally and seasonally variable processes of eutrophication many natural processes are involved besides anthropogenic influences, causing a high complexity and variability.

To subdivide marine waters into eutrophication categories, appropriate assessment parameters are required. OSPAR listed a number of parameters which are supposed to be linked to anthropogenic eutrophication, missing however, were the inputs and discharges of organic material. The single items in this “checklist for an holistic assessment” (EUT(2) 97/2/Info.1-E) have to be proven relevant as eutrophication assessment criteria for the extended German Bight on the basis of the present state of scientific knowledge and available data.

In keeping with the OSPAR Comprehensive Procedure, table 3.4.1 gives an overview of the basic parameters which have mostly been applied to the area of investigation until now and assigns them to one of the three OSPAR categories for eutrophication areas.

Tab. 3.4.1: Classification of eutrophic areas in German territorial waters by means of qualitative assessment parameters



non problem area

nutrient supply low, organic input low, no harmful blooms

causative factors:

nutrient supply high

organic input high

supporting factors

denitrification potential low

light availability good
- irradiance sufficient
- turbidity low

climatic/weather conditions
- elevated temperature
- low wind speeds

hydrodynamic conditions
- stratification +/- stable
- horizontal density gradients (fronts)
- flushing low
- reduced bottom water exchange
- net deposition

zooplankton grazing low in relation to primary production

possible consequences:

phytoplankton
- primary production high
- elevated biomass (bloom intensity, duration, frequency)
- shifts in species composition
- occurance of toxic algae species

enrichment of particulate organic matter in the bottom layer

oxygen decrease

zoobenthos
- shifts in species composition, abundance and biomass

potential problem area

shifts in species composition and abundance in planktonic and benthic communities resulting in proliferation of nuisance forms due to changed nutrient ratios and concentrations

plankton mass occurance
- with foam, mucilage, toxin production, high water turbidity and discoloration

oxygen deficiency in water and surface sediments, H_2S and CH_4 formation

zoobenthos and fish mass mortalities due to oxygen deficiency

problem area

Annotation: Because of the negligible importance of macrophytes, microphytobenthos and "black spots" in the German Bight (Wadden Sea excluded) these items are not considered.

Non Problem Area

In a Non-Problem-Area concentrations of inorganic nutrients are low and the amount of organic material deriving both from autochthonous and allochthonous sources is so small that its stepwise decomposition in the food web would not favour any of the naturally abundant trophic communities.

Apart from more or less stochastic, meteorological factors, such as light availability, the nutrient supply forms the substantial basis for phytoplankton growth and can hence be considered as the first decisive parameter for a differentiation of marine waters into Non-Problem-Areas or Potential Problem Areas. In poorly nutrient supplied waters a facultative enrichment of organic material due to primary production cannot set in. OSPAR suggested the following values for winter stocks of enhanced nutrient concentrations (EUT 1/2/4, EUT (1) 97/3/3): > 0.8 µM phosphate / > 15 µM nitrogen, which may allow for a phytoplankton standing stock of about 1 mg PPC/L, if the Redfield ratio is taken as a basis. Complete respiration of this amount of organic carbon would require about 2.3 mg O_2/L (Parsons et al. 1977, p. 136), which is roughly equivalent to a 30 % consumption of saturating O_2-concentration in the German Bight in summer.

However, winter nutrient concentrations are not well suited for a classification of nutrient availability for primary production during the growth season. Rather, nutrient fluxes into the area of interest during spring and summer have to be addressed, taking into account the residence time of the water body and the remobilization of nutrient minerals from previously formed organic matter and from the sediments.

Potential Problem Areas

If "elevated" anthropogenic nutrient supply is high enough to sustain phytoplankton blooms at a level beyond that mentioned above, deteriorating eutrophication effects may be a consequence. The marine area should thus be classified as Potential Problem Area regardless of the manifestation of any undesirable events. It is important to notice that the nutrient enrichment, high primary production or a high biomass alone do not represent undesirable effects a priori.

If primary production can be transferred to higher trophic levels through the food chain there will generally be no uncoupling of the pelagic food chain. The prerequisite is that the grazing potential, or more precisely the organic carbon demand of the mesozoan secondary and tertiary producers, can get along with this supply. Otherwise there will be a gradual accumulation of primary organic material, resulting in exceptional blooms which are followed by potential deterioration of the ecosystem and/or water quality. For a sound evaluation of eutrophication sensitivity, it is thus necessary to relate primary production to the capacity of organic carbon demand in the food chain.

Problem Areas

Eutrophication Problem Areas are considered here to be exclusively those zones where undesirable effects were manifested in a symptomatic sense in the past, but only if these effects could be traced back to the eutrophication process (anthropogenically elevated nutrient supply) with a high plausibility. In the German Bight this holds mainly true for seasonal oxygen deficiency and phytoplankton mass occurrences although these phenomena, which are highly dependent on hydrodynamic and biological conditions, were not observed regularly. Even though increased formation or input of organic matter is not necessarily the only reason for oxygen reduction in the water, it is a key condition for rapid oxygen consumption in the bottom layer or in the surface sediments.

The microbial degradation of particulate organic matter is the most important oxygen consuming process. In the German Bight 80 % of the total oxygen consumption takes place in the subthermocline water body within a short time (Hickel et al. 1989, p. 69). Phytoplankton blooms were found to constitute the main source for this organic material (Hickel et al. 1989, p. 67).

Mass mortalities of zoobenthos and fish in the region were directly linked to the phenomenon of oxygen deficiency and caused a further escalation of oxygen consumption. A reduction of total species numbers by 30 - 50 % was observed in the German Bight in August 1983 under starving oxygen conditions when compared to the following years with sufficient oxygen supply (Hickel et al. 1989, p. 69). For a more detailed causal analysis it is important to discriminate mortalities within the benthic fauna due to oxygen deficiency from other possible causes with a high probability, since observed changes in zoobenthic communities or fish stocks cannot necessarily be attributed to oxygen deficiencies alone.

With respect to phytoplankton mass occurrences it is important to distinguish between blooms, which derive from active growth processes, and passive accumulation of cells, which often takes place at horizontal density gradients or fronts subjected to convergence. But even without bloom formation or mass accumulations, effects like foam, mucilage and discolouration may also appear. This will lead to a deterioration of the aesthetic water quality which can seriously affect tourism.

Mass developments of toxic algae constitute another undesirable effect of eutrophication. Additionally, there is increasing evidence that shellfish poisoning may occur even without any bloom formation. The reasons for attributing toxin formation to eutrophication are threefold:
Firstly, a shift in nutrient ratios towards increased nitrogen levels will favour flagellate blooms, a group with a high number of toxic forms. Secondly, in many species, toxin production is enhanced at low phosphate concentrations compared to nitrogen. Thirdly, under anoxic conditions, dinoflagellates, which are able to form resting cysts, have a competitive advantage over other forms. A substantial number of toxic dinoflagellates have the capacity of forming resting cysts. Among those, seven occur in the German Bight.

A further criterion for the classification as eutrophication Problem Area is a shift in species composition and abundance which, more generally spoken, results in a proliferation of nuisance (and/or non-edible) forms due to nutrient induced changes in the plankton communities. The same applies to the zooplankton and to the benthos.

3.4.2 Reference background concentrations

The OSPAR assessment procedures are based on comparisons with reference background concentrations. These concentrations have been up to now mainly summarised for the nutrients (Tab. 3.4.2.1). Even for the nutrients, many simplifications and plausible assumptions have been made due to the lack of scientifically sound historical data. It has been discussed by Billen et al.

(1999) that already at the end of the 19th century industrial impacts and increased populations in some cities caused significant discharges and possible eutrophication effects.

In the following table, besides the values given in the OSPAR tables (OSPAR 2001) some other data also have been given for supplementation and comparison.

Tab. 3.4.2.1: Eutrophication - background concentrations of nutrients

Background concentrations/locations	TN µM	DIN µM	NO_3 **NH_4** µM	TP µM	PO_4 µM
River fresh water S: < 1					
estimated (ARGE Elbe 1990,1991)	142		71	1.63	0.32
1911-14 (Brandt 1927)			**7**		
Ems (van Raaphorst et al. 2000)	45			1.8	
Rhine (van Raaphorst et al. 2000)	91		71	2.3	
European Rivers (Laane 1992)	20-71		4-14 **2-14**	0.7-4.5	0.1-2
Dutch Wadden Sea S: 18 – 33.5					
ca.1930 (van Raaphorst et al. 2000)	13			0.8	0.3
1950 (Postma in Krey 1956)		4		1.6	0.6
Coastal water S: 30 - 34.5					
assumed (OSPAR 2001)		19 (NL)			0.65
German Bight					
Laane 1992					0.5
1911-14 (Brandt 1927)			7		
1936 (Kalle 1937)				0.8	0.6
Offshore winter S: > 34.5					
assumed (OSPAR 2001)		10			0.60
1987* (Brockmann and Kattner 1997)	18	15		0.7	0.65
means 1984-2000 (Brockmann and Topcu 2001)		10	9		0.7
1950 (Krey 1956)				0.9	0.7
Off shore summer S: > 34.5					
1986** (Brockmann and Kattner 1997)	18	6.5		0.5	

Mainly annual means are formulated. Means have also been taken from the individual references, neglecting spatial and short-time variability to simplify the table.

* For the offshore spatial references, areas were chosen between the coastal water area and the Dogger Bank area, where continuous primary production reduces the nutrients throughout the year significantly (Brockmann and Kattner 1997; Brockmann and Topcu 2001).

** For summertime, reference values were taken from the open North Sea as well, but only total N and P should be used, due to the fast conversion rates between the different nutrient phases. Vertically integrated values were taken because of the stratification effect on production and decomposition separating the different regimes. The reference areas were chosen between the coastal water area and the Dogger Bank area with continuous primary production (Brockmann and Kattner 1997).

Due to the individual characters of rivers, draining different soil structures, N/P/Si relationships can be very different and need more evaluation.

Quantitative reference concentrations for the biota have not been compiled so far, except for chlorophyll a, for which background concentrations of 2 - 4 µg/L have been defined for offshore areas (OSPAR 2001). For the Netherlands 4 µg/L in the offshore region and 10 µg/L in the coastal water have been identified.

3.4.3 Eutrophication assessment of the German Bight

3.4.3.1 Detailed assessment

A short assessment has been published recently (Ladwig et al. 2001). The assessment in Tab. 3.4.3.1 follows the OSPAR classification schema: Problem Area, Potential Problem Area and Non Problem Area (PA, PPA, NPA). It differentiates mainly between coastal water with salinities < 34.5, covering most of the German Bight in the range between 30 and 34.5, and offshore water with salinities > 34.5. Since nutrient gradients (due to dilution) and eutrophication effects mostly decrease towards the open sea, offshore water usually gets a higher ranking. Concentrations of the recent status and reference concentrations, when possible, are given for direct comparison.

The assessment follows the OSPAR proposal of the Comprehensive Procedure, so that concentrations exceeding the background values by 50 % are evaluated as "problem". Due to the variability, some parameters overlap the PPA and NPA categories and are therefore scored as PPA in this table.

The table is mainly based

(i) on OSPAR criteria, and data for the period 1990 - 1997 outlined in Tables B (OSPAR 2001). "Bx" is refers to these tables,
(ii) on data sets compiled in this report (Brockmann et al. 2002).

Reference concentrations referred to as natural background values have been taken mainly from OSPAR (2001), supplemented by some other references.

Table 3.4.3.1: Detailed eutrophication assessment of the German Bight, related to OSPAR criteria

Assessment criteria/Assessment	Reference	**PA**	**PPA**	**NPA**
I.1 Riverine TN, TP loads related to Q (kt/y) (B 0, B 1, B 2)	N: 77 P: 2	N: 219 P: 9		
I.2 Winter DIN, DIP concentrations (µM) (B 3, B 4)				
Salinity 30-34.5	N: 19 P: 0.65	N: 10-50 P: 0.65-1.6		
Salinity > 34.5	N: 10 P: 0.6		N: 5-15 P: 0.5-0.7	
I.3 Nutrient ratios (M/M) (B 6)				
I.3.1 N/P				
Salinity < 34.5	16		15-40	
Salinity > 34.5	16			7-15
I.3.2 N/Si				
Salinity < 34.5	2		2-6	
Salinity > 34.5	2		1-4	
I.3.3 P/Si				
Salinity < 34.5	0.12		0.07-0.4	
Salinity > 34.5	0.12			0.1-0.2
II.1 Chl.a concentrations, growing season (µg/L) (B7)				
II.1.1 Means: Concentr., S: < 34.5	10 (NL)		4 - 23	
> 34.5	2 – 4	5 - 28		
Duration, Extension	ni			
II.2.1 Maxima: Concentr., Frequency (different seasons)	ni ni	high		
II.2 Nuisance and/or toxic species (B 8, B 9, B 10)				
II.2.1 Duration	Ni	ni		
II.2.2 Extension		80 %		
II.2.3 Frequency	Ni	ni		
II.3 Macrophyte and macroalgae changes (B 11)		Not relevant		
III.1 Degree of oxygen depletion (B 12)				
(mg/L)	8 mg/L	2 mg/L		
(% saturation)	100 %	< 25 %		

III.2 Kills in zoobenthos and fish (B 12)		observed		
III.2.1Benthic community structure (B 11)				
Changes in biomass		observed		
Changes in abundance		observed		
Changes in species composition		observed		
III.2.2 Ecosystem structure (structural changes)		observed		
III.3 Organic matter during summer				
(B 13) Salinity < 34.5 / > 34.5				
TN	ni/15	15-190/	/6-34	
DON	ni/	6-36/	/2-15	
PN	ni/			
TP	ni/0.5	0.8-6/0.6-2.5		
DOP	ni	0.2-4.3/0.3-1.3		
PP	ni			
DOC	ni			
POC	ni			
IV Algal toxins occurrence (B 12)		observed		

Bx = tables in OSPAR 2001   ni = no information

Comments on Tab. 3.4.3.1

Category I: Degree of nutrient enrichment

I.1: River discharges have been calculated as means for 1990 - 1997 (Table B 1 and B 2, OSPAR 2001). The mean freshwater discharge (Tab. B 0) during this period has also been used for the calculation of background discharges. Since the recent discharges significantly surpass the calculated background discharges, the receiving area was assessed as PA.

I.2: Nutrient concentrations in the coastal area (salinity < 34.5) are compared with background concentrations from the Netherlands (Tab. 3.4.2.1), since for the German coastal water no reference value has been defined yet. Mean concentrations (1985 - 1998) were 10 - 50 µM DIN, with an increasing variability towards the coast (Fig. 2.3.1.8). The concentration range was 8 - 173 µM DIN (1990- 1997) (Tab. B 3 in OSPAR 2001). Phosphate means were between 0.65 and 1.6 µM (Fig. 2.3.1.3) with a variability mostly below 50 %, and a range of 0.5 - 2.8 µM (Tab. B 4 in OSPAR 2001). Hence, DIN exceeded 28 µM and DIP 1 µM within the inner German Bight, values which were above the elevated concentrations (background concentration + 50 %).

I.3: Nutrient ratios partly exceed the reference values defined by OSPAR (2001). This holds for DIN/DIP with means (1985 – 1998) of 15 - 40 (M/M) in the coastal water and 7 - 15 offshore

(Fig. 2.3.1.9), and for N/Si with means of 2 - 6 in the coastal water and 1 - 4 offshore (Fig. 2.3.1.10). Therefore, an assessment as PPA follows. However, the differences between nitrate and silicate showed an increasing gradient towards the coast. Within the regionally defined PA a significant surplus of nitrate was mainly found (Fig. 3.1.3). Mean P/Si ratios in the coastal water were 0.07 and 0.4 (M/M), caused by a surplus of phosphate in relation to the reference values (Fig. 2.3.1.10) at some locations. Offshore these ratios were mainly between 0.1 and 0.2 and thus close to the reference ratio.

Category II: Direct effects of nutrient enrichment

II.1: Chlorophyll a data used from 1990-1997, were measured at Helgoland roads (OSPAR 2001). Summer maxima remained high: means in the coastal water (salinity < 34.5) ranged from 7 - 20 µg/L, nearshore (salinity < 30); 20 - 40 µg/L during spring; 5 - 10 µg/L and 7 - 30 µg/L, respectively, during summer (Tab. B7, OSPAR 2001). Maxima of about 20 µg/L were observed in the inner German Bight until 1998 (Fig. 2.4.3.1, 4, 5, 6). The same range is indicated by phytoplankton carbon estimations (800 µg C/L) at Helgoland (Fig. 2.4.2.2).

II.2: Nuisance and/or toxic species were frequently detected (Tab. 2.5.2.1; see also Tab. B 8 – 10, OSPAR 2001), covering up to 80% of the PA (Fig. 2.5.3.2, 3, 6). Various harmful and toxic species have been detected (see chapter 2.5). Besides *Phaeocystis spec.*, the harmful species generally changed interannually. However, monitoring since 1990 covers only the coastal area (Fig. 2.4.3.3).

II.3: Macrophytes including macroalgae were only monitored on the rocky base of Helgoland. An increase of *Ulva* was detected.

Category III: Indirect effects of nutrient enrichment

III.1: Oxygen depletion has been observed by chance during several years, because no regular monitoring is conducted. Concentrations in the bottom water of stratified areas dropped below 2 mg O_2/L in the years 1981, 1982, 1983, 1989 and 1994 (Fig. 2.7.1 – 5, 2.7.7 – 9).

III.2: Changes and kills in zoobenthos and fish kills have been observed in the German Bight and in the adjacent northern area (Fig. 2.7.4, 6). During 1981 - 1983 benthic mass mortalities were observed in connection with oxygen depletion (Tab. B 12, OSPAR 2001). An increase of macrozoobenthos biomass has been detected for sandy sediments (Fig. 2.8.2.2.1, 2.8.2.3.1).
Changes in the ecosystem structure have been observed for the phytoplankton. Especially the increase of flagellates since the late 1970s (Fig. 2.5.1.1) is discussed in context with eutrophication (Hickel et al. 1997).

III.3: So far, organic matter has not been a constituent part of monitoring programs. There are only some research data available for the German Bight, which show variabilities in the range of inorganic nutrients (see Fig. 2.3.7.10 – 12). Since DON and DOP as well as PN and PP replace the dissolved inorganic nutrients during the growth season, not only inorganic nutrients should be monitored, but at least TN and TP which should be compared to the corresponding background concentrations.

Category IV: Other possible effects of nutrient enrichment

IV: Algal toxins were detected as DSP (diarrhetic shellfish poisoning) in mussels from the Wadden Sea in several years (Tab. B 12, OSPAR 2001).

3.4.3.2 Integrated assessment

For an integrated assessment the ranking within the different categories will be combined in Tab. 3.4.3.2 in an holistic evaluation. For each assessment parameter of the different categories in Table 3.4.3.1 it was indicated whether its measured concentration or observed effect relates to a PA, PPA or NPA as defined in the OSPAR Strategy to Combat Eutrophication (OSPAR agreement number 1998 – 18, Annex 1, § 1 (d – f)). The results of this step are summarised in Tab. 3.4.3.2 and explained (cited from OSPAR 2002) below:

a. Areas showing an increased degree of nutrient enrichment accompanied by direct and/or indirect/ other possible effects are regarded as **'problem areas'**,
b. Areas may show direct effects and/or indirect or other possible effects when there is no evident increase in nutrient enrichment, e.g. as a result of transboundary transport of (toxic) algae and/or organic matter arising from adjacent/remote areas. These areas could be classified as **'problem areas'**,
c. Areas with an increased degree of nutrient enrichment, but without showing direct, indirect/ other possible effects, are classified initially as **'potential problem areas'**,
d. Areas without nutrient enrichment and related (in)direct/ other possible effects are considered to be **'non-problem areas'**.

The left part of the following table contains the assessment of the German Bight (GB), the right part the general assessment by OSPAR (2001). For this general assessment, the German Bight was subdivided into coastal water with a salinity < 34.5 and offshore water, salinity > 34.5.

Table 3.4.3.2: Integrated assessment of the categorised parameters

Factor/Classification	GB < 34.5	GB > 34.5	PA	PA	PPA	NPA
Category I Nutrient Enrichment	+		+	-	+	-
Category II Direct Effects		+	+ and/or	+ and/or	-	-
Category III + IV Indirect Effects Other poss. Effects	+	+	+	+	-	-

Since for both areas nutrient enrichment and/or eutrophication effects have been found, the German Bight is assessed as Problem Area. A geographical differentiation related to the location of effects and nutrient enrichment results in the classification of the main part of the German

Bight as Problem Area, whereas the outer part is classified as NPA with a transitional area in between classified as PPA (Fig. 3.1.0, 3.1.1, 3).

However, the German Bight has to be addressed as a sensitive area for eutrophication effects for natural reasons due to

- the close vicinity of still nutrient-rich river plumes where the phytoplankton is light-inhibited by the high natural turbidity, as well as to thermohaline stabilised mixed layers, which promote the formation of phytoplankton blooms,
- long residence times allowing extended phytoplankton growth,
- even longer residence times of bottom water in thermohaline stratified areas, allowing continuous trapping and degradation of sedimented organic material coupled with oxygen depletion, and
- trapping of suspended material in estuaries and tidal flats by estuarine - type circulation, (driven by lateral density gradients) and asymmetric tides providing a permanent import of particulate organic material also from the sea.

3.5 Coupling between the German Bight and the Wadden Sea

Parallel to the German Bight project, another project was implemented, focussing on eutrophication criteria in the Wadden Sea (van Beusekom et al. 2001). For this reason the Wadden Sea processes have been mostly excluded in this German Bight report.

Due to the tidal exchange of water masses between the Wadden Sea and the German Bight the nutrients and suspended matter are also frequently exchanged, linking the different biogeochemical processes including eutrophication. For this reason, the results from the Wadden Sea study will be briefly reported, and some aspects, mostly concerning the German part of the Wadden Sea, are discussed here.

The nutrient regime in the Wadden Sea is also directly affected by the river discharges, which pass this area through the estuaries but also spread into the tidal flats by the prevailing western wind forces. As already outlined (see chapter 2.3.8), the Wadden Sea also receives particulate material from the coastal water, being trapped in the tidal flats. Consequently, nutrients are released seasonally following remineralisation and remobilisation. Especially phosphate is trapped for a longer period by precipitation in the sediments. Phosphate bound in iron-complexes is released mostly during summer (van Beusekom et al. 1999, 2001; Dick et al. 1999; Pohlmann et. al 1999). On the other hand, phosphate precipitated as apatite (Ca-P mineral), which is the major component of phosphorus in Wadden Sea sediments, is fixed over geological time scales (van Beusekom et al. 2001, p. 19).

Caused by the imports of organic matter (about 100 g C/m^2 y), heterotrophic processes dominate in the Wadden Sea (van Beusekom et al. 1999; Hesse et al. 1995 b). The N remineralisation products ammonium and nitrite reach highest values at the end of the annual primary production season, during autumn (van Beusekom et al. 2001, p. 36). These data were used to estimate the eutrophication status of the Wadden Sea. In the area south of the Elbe mouth direct or indirect correlations between nitrogen inputs from the Rhine and Meuse have been found (ibid., p. 27 and 29). For the northern area such correlations were only found for nitrate concentrations during March in the German Bight (ibid., p. 35).

The primary production is at present mostly nitrogen limited in the Wadden Sea, because DIN/PO_4 ratios usually drop below 5 (M/M) during summer (Brockmann et al. 1999 a). However, the production has increased during the last decades (van Beusekom et al. 2001, p. 52). The duration of *Phaeocystis* blooms in the Marsdiep has surpassed the "historic" reference level in the 1980s (ibid., p. 38 ff.) and showed some correlations with discharges of ammonium (% of TN) and silicate.

It is assumed that for the northern Wadden Sea, not the Elbe river input, but nutrient input from the west into the German Bight determines primary production in the German Bight and consequently the organic matter import to the northern Wadden Sea (ibid. p. 45). However, since the Elbe/Weser river plume often passes the tidal flats along the coastline of Schleswig-Holstein (Brockmann and Eberlein 1986) it must be assumed that these direct imports are significant as well.

Different effects of eutrophication have been observed in all biota, but without any clear dose-response relationship, probably due to the high variability, both in space and time, in this area at the border between the sea with events like high floods and the land with highly variable weather (cold winter, hot summer) affecting especially the eulittoral, superimposed by tidal cycles.

Natural background concentrations during autumn have been defined as about 3 µM ammonium + nitrite in the western Dutch Wadden Sea. This value has been transferred to the whole Wadden Sea where it is surpassed by more than 50%, suggesting that the Wadden Sea is a "Problem Area" concerning eutrophication.

It is assumed that for the Wadden Sea the agreed reduction of nutrient discharges by 50 % is not sufficient to improve the status to a "Potential Problem Area". In addition to the reduction of river discharges, also the atmospheric inputs, assumed to be especially effective in the shallow tidal flats, have to be reduced as well.

4. Conclusions

4.1 Deficits and probable solutions

Even long time monitoring data from key locations in the German Bight area have some shortcomings due to modification of methods and/or insufficient estimation of their representativeness (see chapter 3.2 and 3.3). Monitoring data from different locations have not sufficiently been intercalibrated for most parameters. The calibration has to be improved. Intercomparison should involve international exercises and quality assurance. Monitoring especially during summer in the German Bight has to be intensified, since oxygen depletion in the bottom layer and other eutrophication effects were up to now mostly detected by chance.

More information is needed on (i) memory effects of sediments (long-term leaching), (ii) effects of low but permanent imports (transboundary transports), and (iii) effects of phasing of stratification, discharge rates, successive remobilisation of nutrients, and of interaction of nutrient supply and other production limiting factors such as trace metals and vitamins in order to detect eutrophication effects more clearly.

There is nearly no information on effects of algal toxins on the food web or of other telemediator effects, which are needed to assess the role of zooplankton grazing more precisely. In this context, it is also necessary to understand better the bottom up control (limiting role of nutrients and light) and the top down control of phytoplankton growth (grazing capacity of zooplankton).

Besides intensified and intercalibrated monitoring of concentrations and fluxes, processes have to be studied which are involved in eutrophication scenarios as outlined in chapter 1.3. In this context, it is important to note that at present no a priori statements about which nutrient is limiting phytoplankton in oligotrophic - eutrophic or coastal - non coastal environments can be made (Peeters and Peperzak 1990). In the German Wadden Sea, there may be a seasonal switch from phosphorus and/or simultaneous silica limitation in spring to nitrogen limitation in summer (Hesse et al. 1995b). High N/P ratios in estuarine waters promote phosphorus limitation in near-coastal areas, while nitrogen limitation is likely to dominate in the central North Sea (Riegman et al. 1990) but may also occur in areas like the North Frisian Wadden Sea (Brockmann, pers. comm.).The *Phaeocystis* bloom along the Dutch coast seems to be phosphorus limited (Veldhuis et al. 1987), while Lancelot et al. (1986) found it to be nitrogen limited along the Belgian coast. The conclusion is that the partial reduction of only one of the macro-nutrient species will not ultimately solve the eutrophication problem.

If only P discharges are reduced, the export of surplus N from the wider estuaries to the often N-limited coastal water will only shift the eutrophication problems offshore (Conley 2000), as well as enhancing toxic blooms. It has been shown by several measures to reduce nutrient loading that it is possible to achieve significant recoveries (Cloern 2001).

Further research projects aiming specifically at the eutrophication problem have been established in the context of the Water Framework Directive of the European Community. Especially the suggested connection between eutrophication and the occurrence of harmful blooms should be studied by combined methods:

(i) screening of gradients and estimation of process data in the open water to achieve a complex data base as reference,

(ii) enclosing defined water masses in mesocosms with different fertilisation, stratification and turbulence, to get quantitative rates and interactions at near natural but controlled conditions and

(iii) series of coupled (to i and ii) laboratory cultures to estimate specific turnover rate constants, competition processes and food web interactions at different levels.

Generally, for any assessment it has to be considered that the high winter nutrient concentrations used for definitions of problem areas are only indicative, because eutrophication effects like the occurrence of harmful blooms, oxygen depletion etc. during summer are not directly linked to winter discharges and concentrations of nutrients at that time.

4.2 Transferability to other areas

The assessment criteria are generally formulated and are deduced from common OSPAR criteria and can therefore directly be applied to other areas, especially to similar areas with trapping functions which are connected to areas with high production or supply of organic material, such as fjords or deep holes close to a coastal area.

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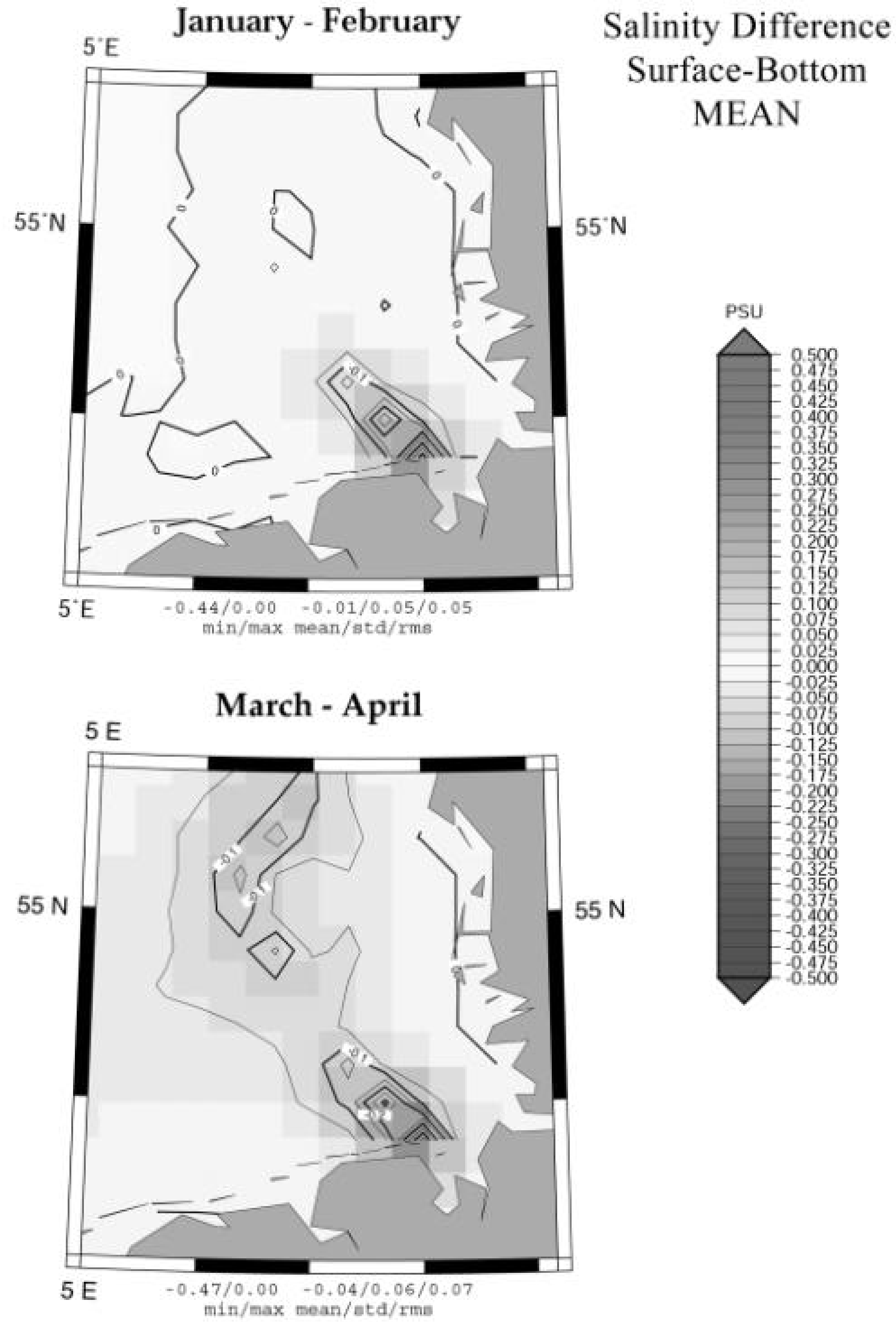

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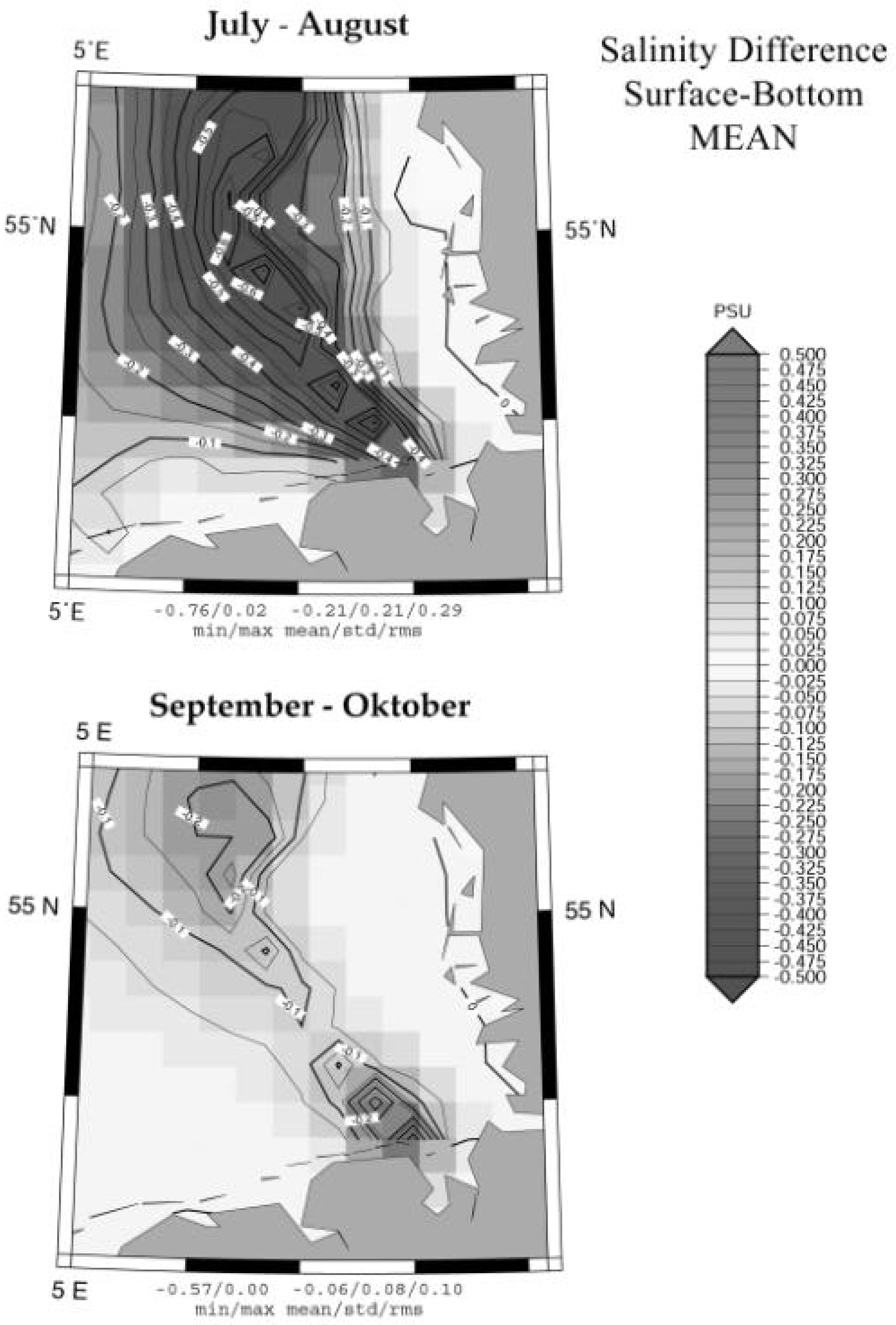

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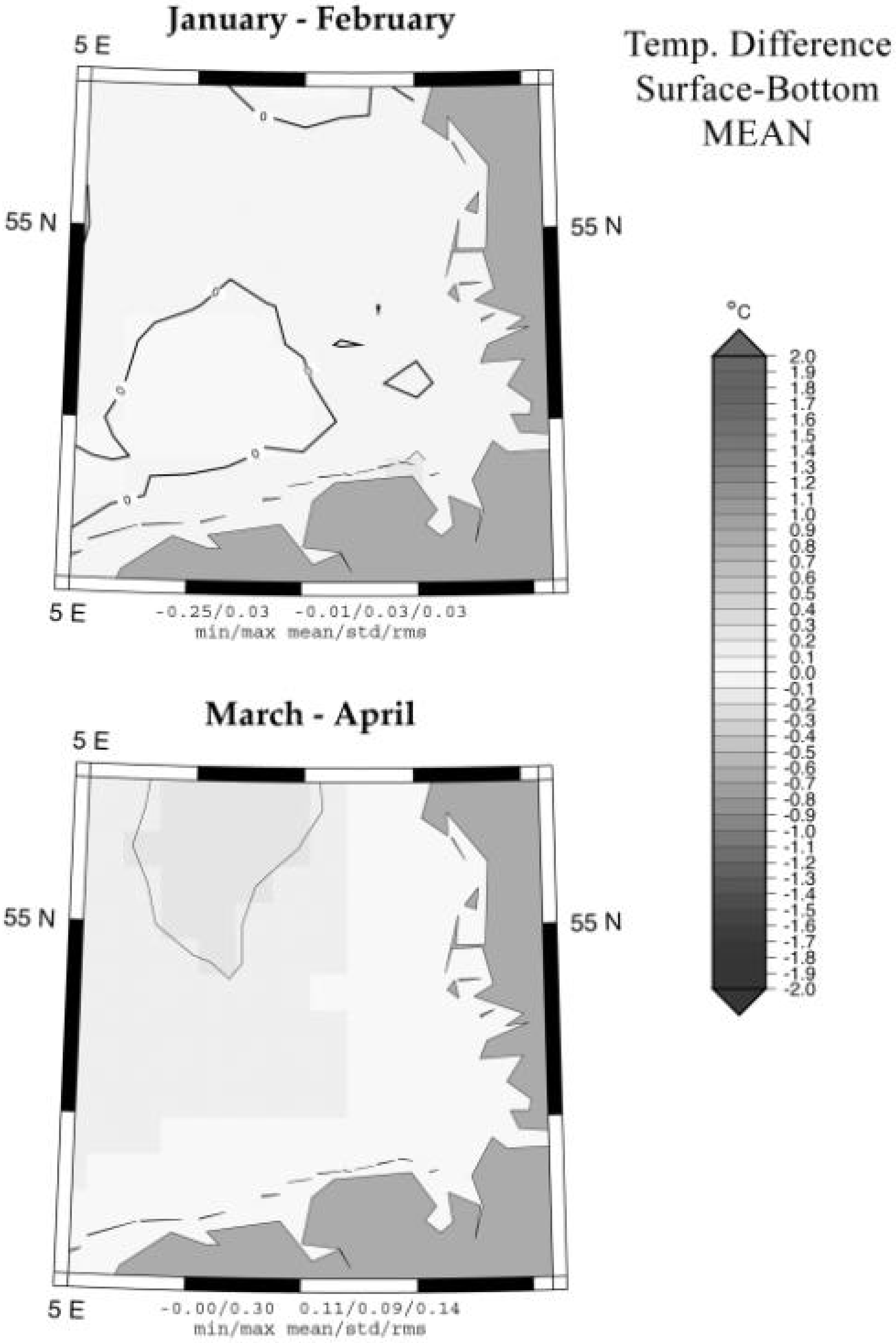

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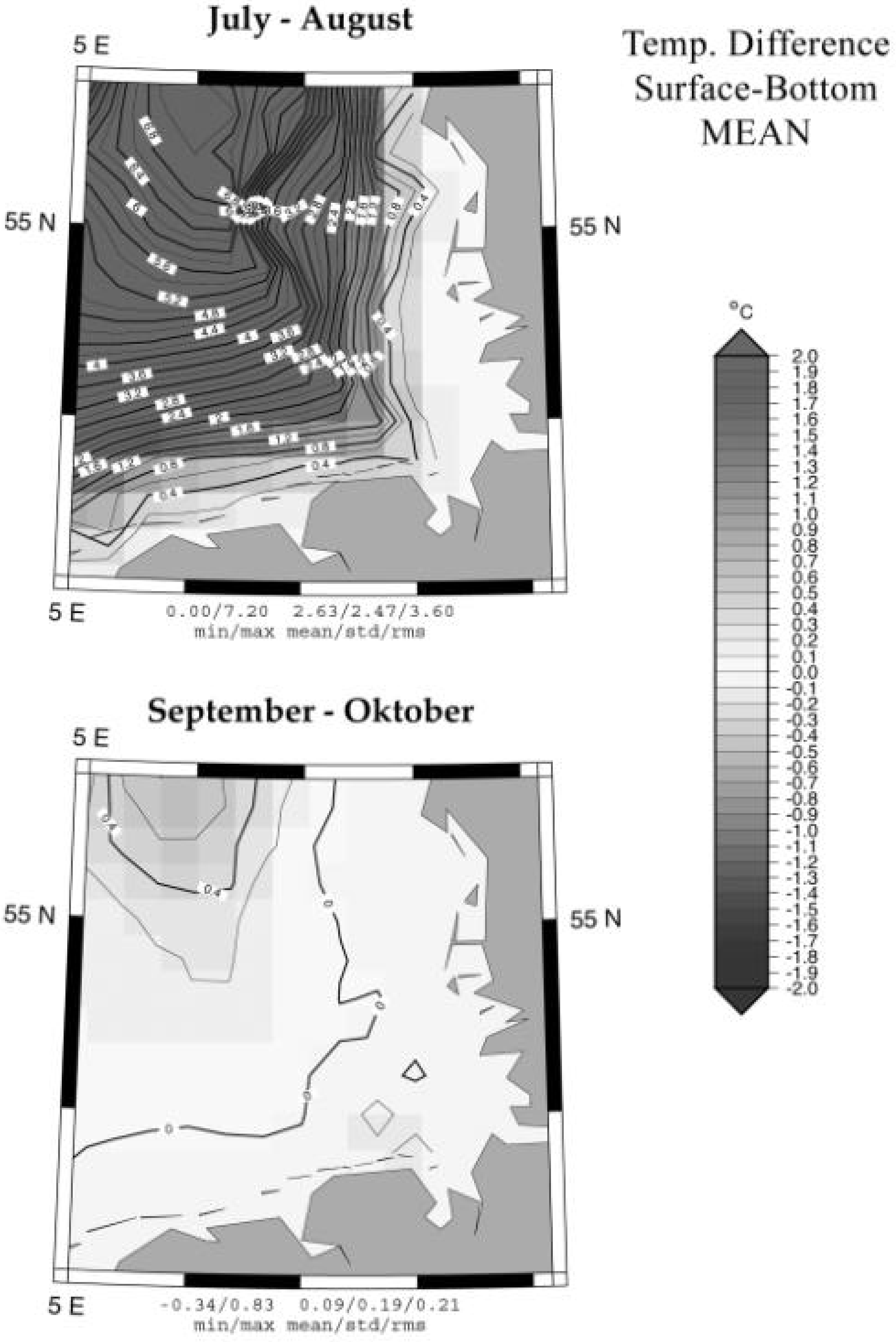

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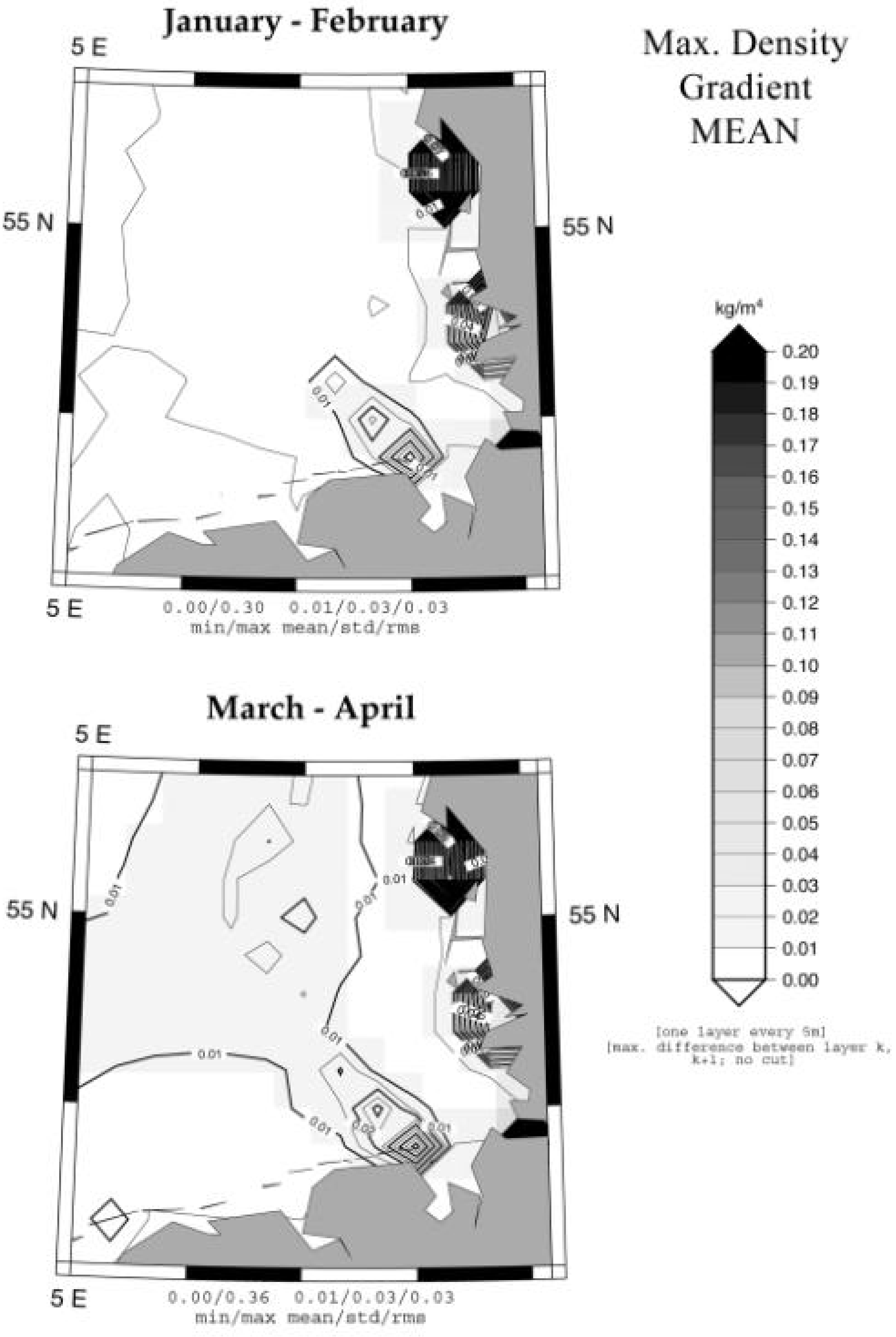

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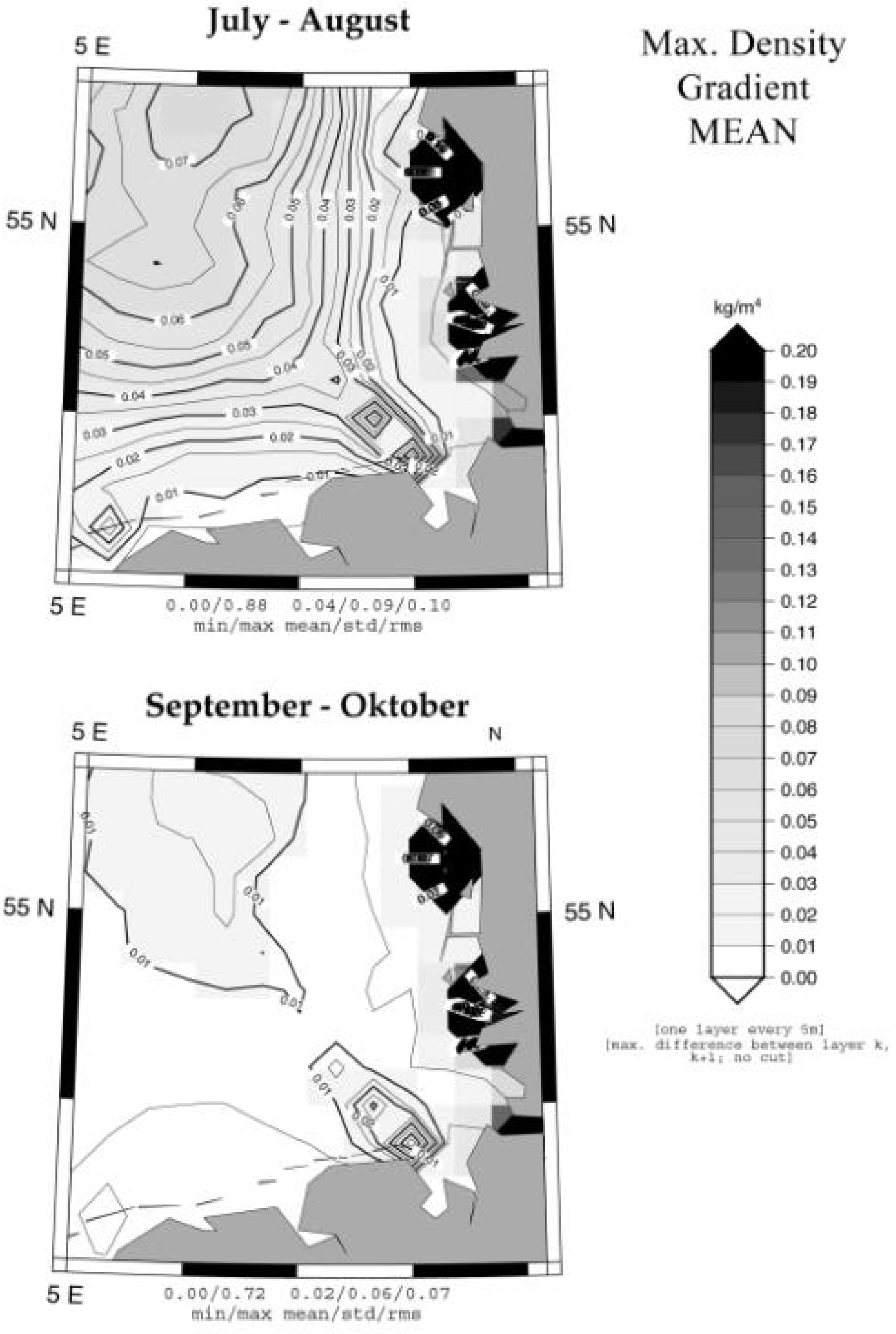

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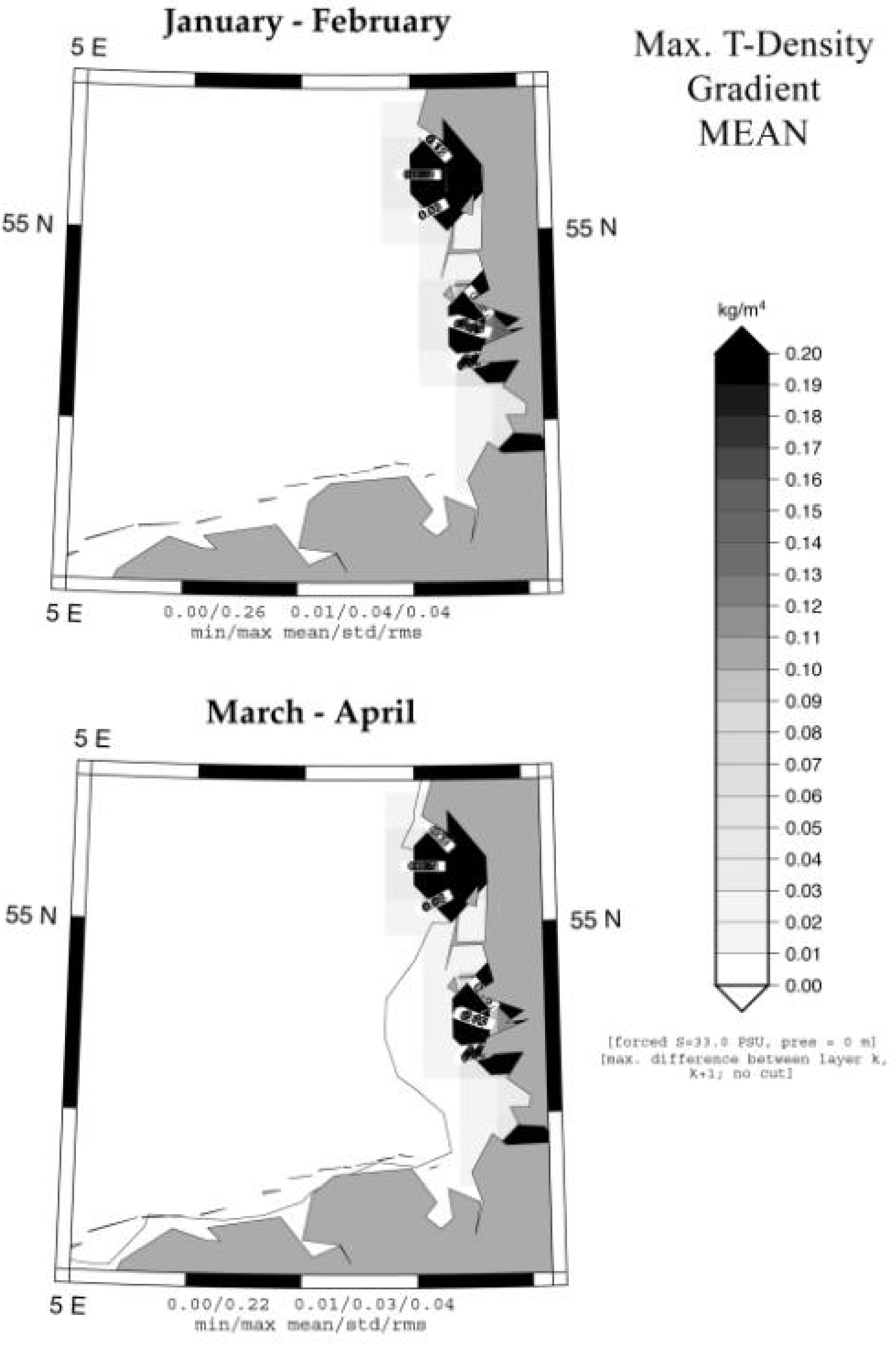

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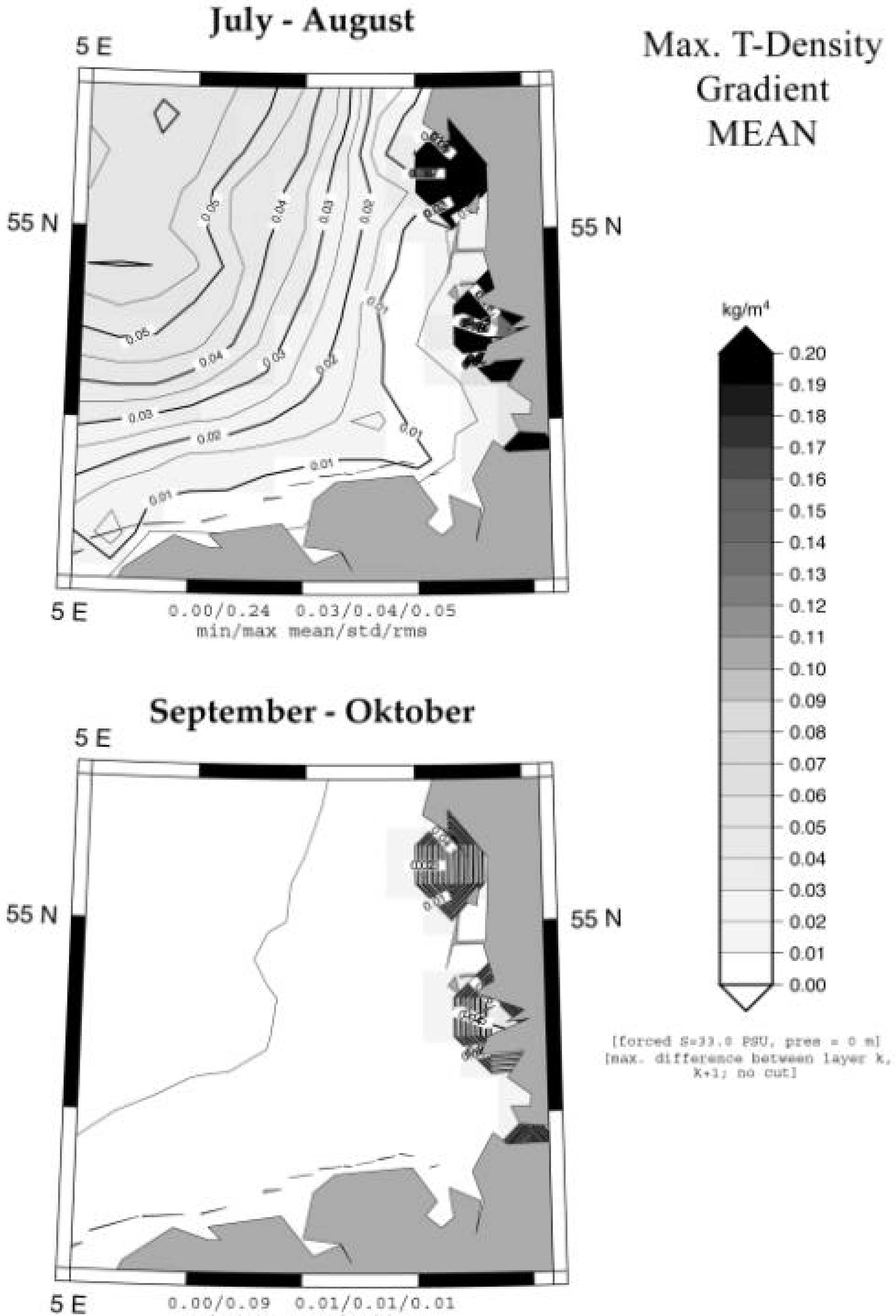

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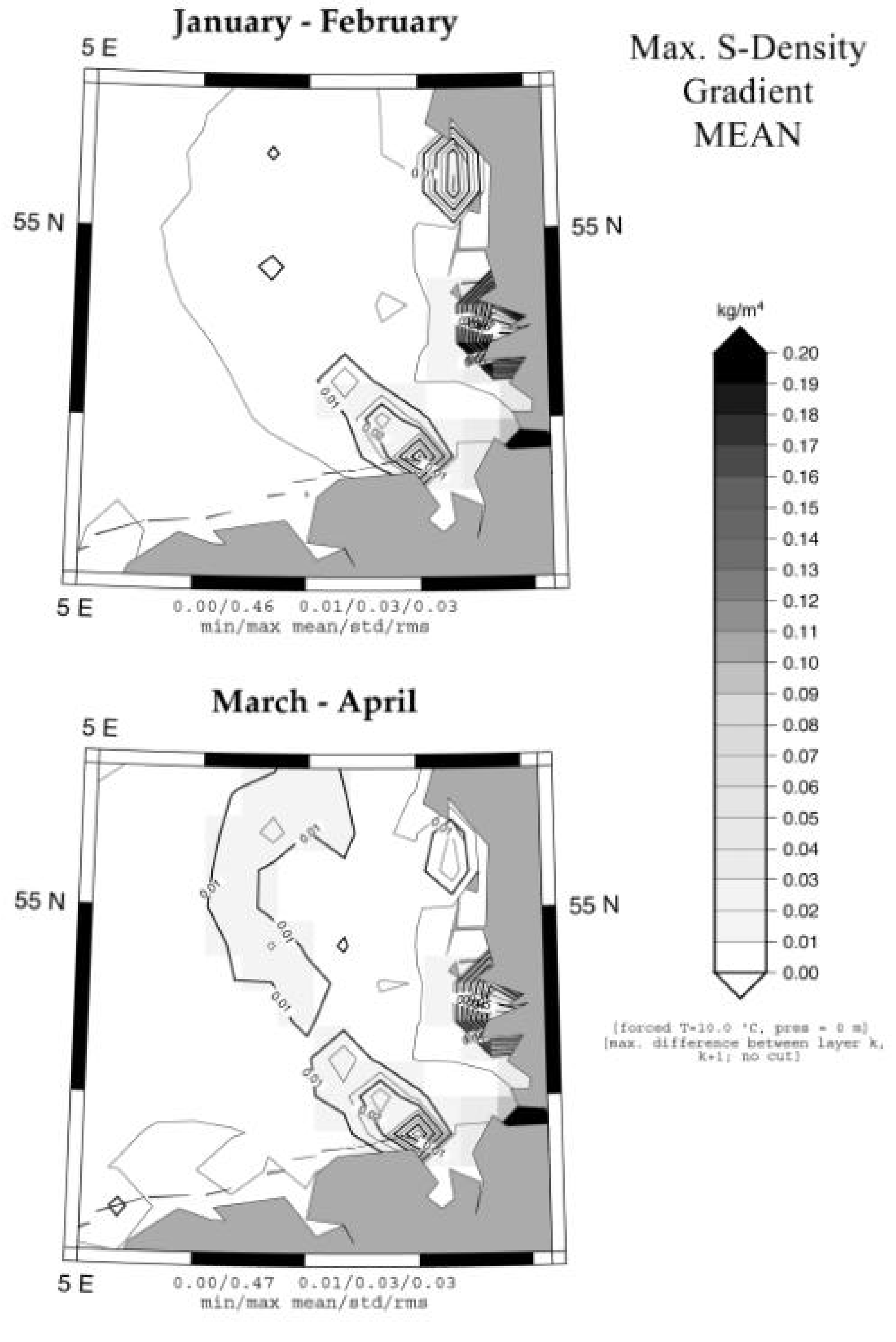

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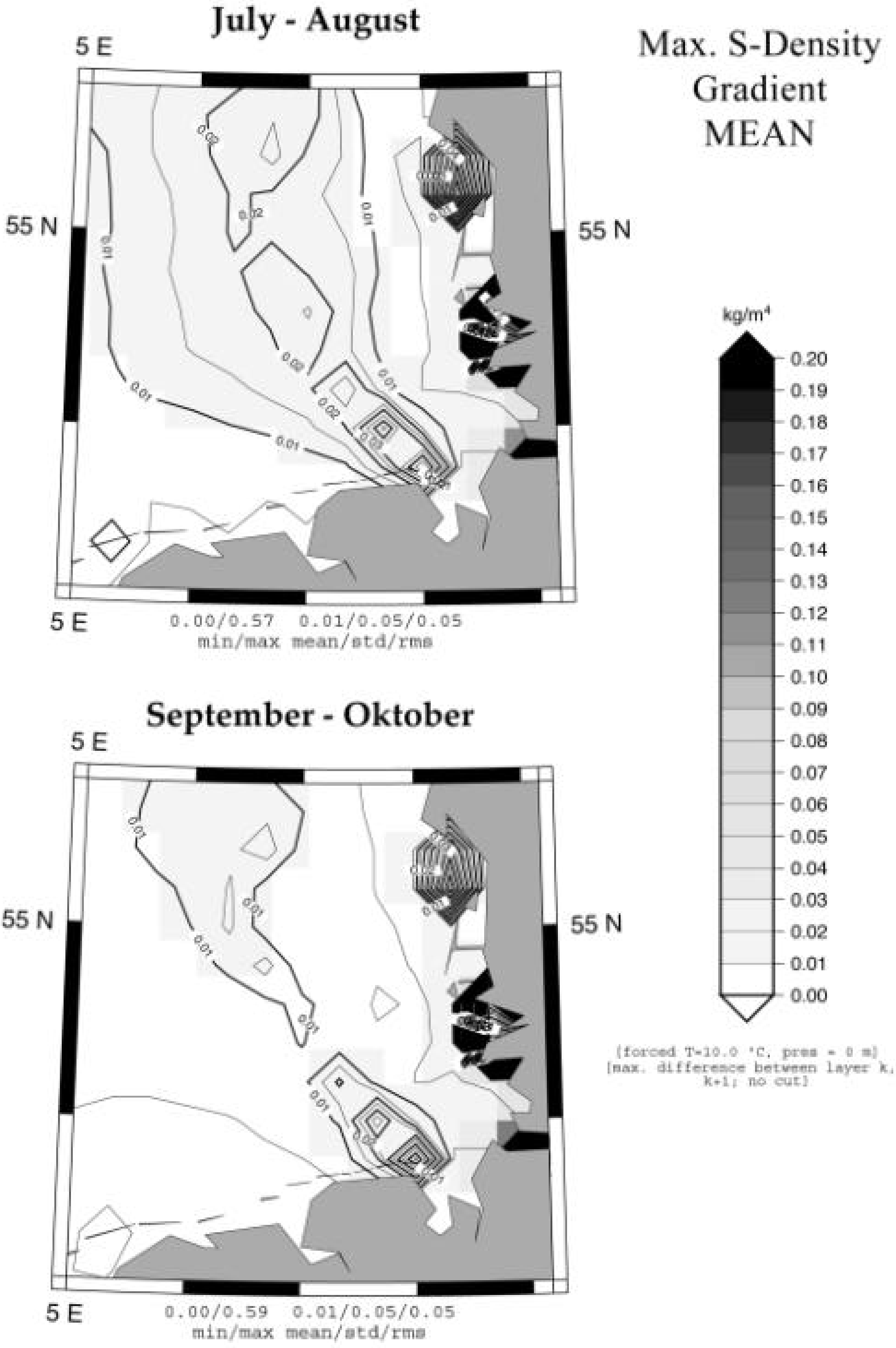

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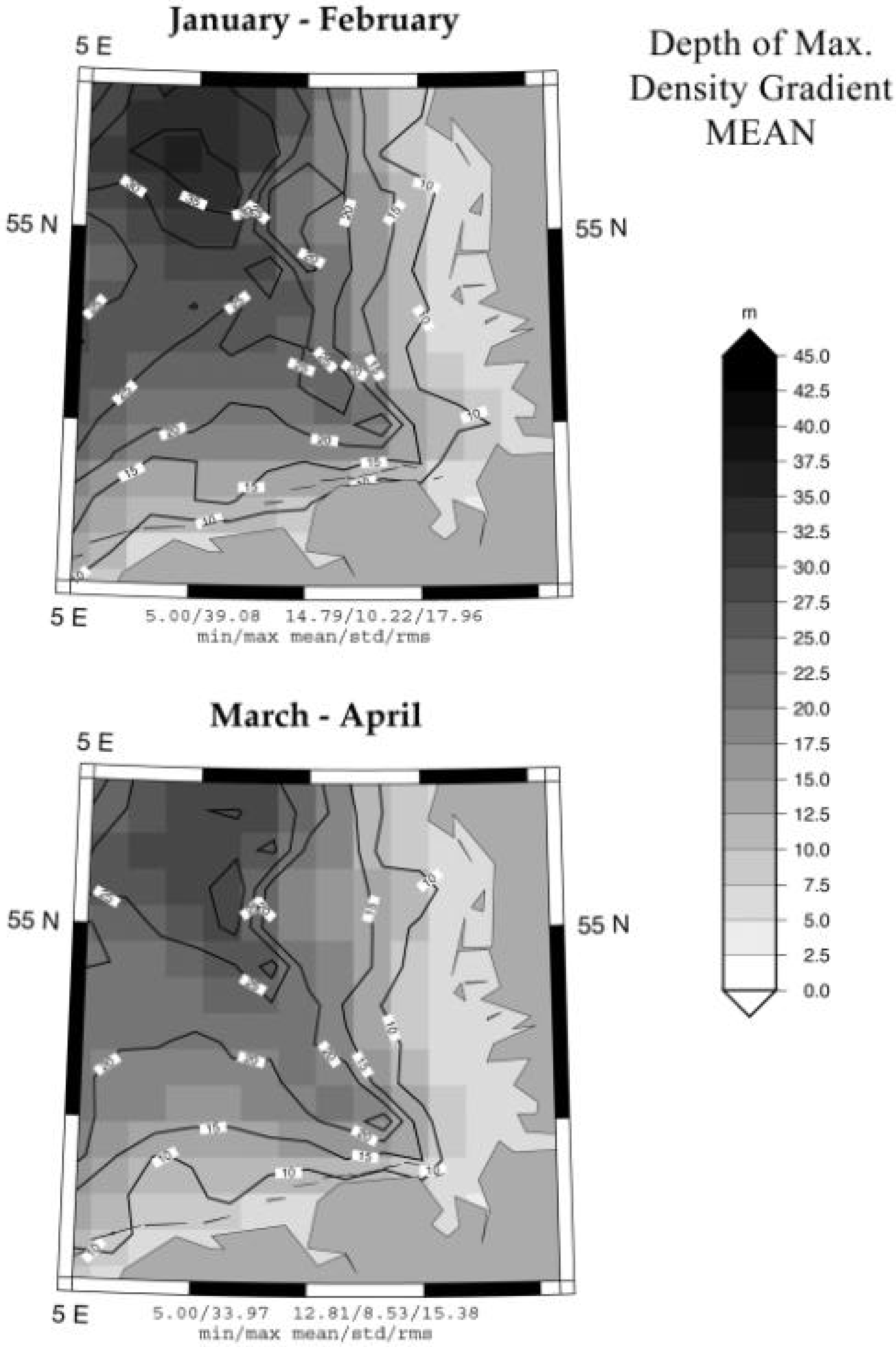

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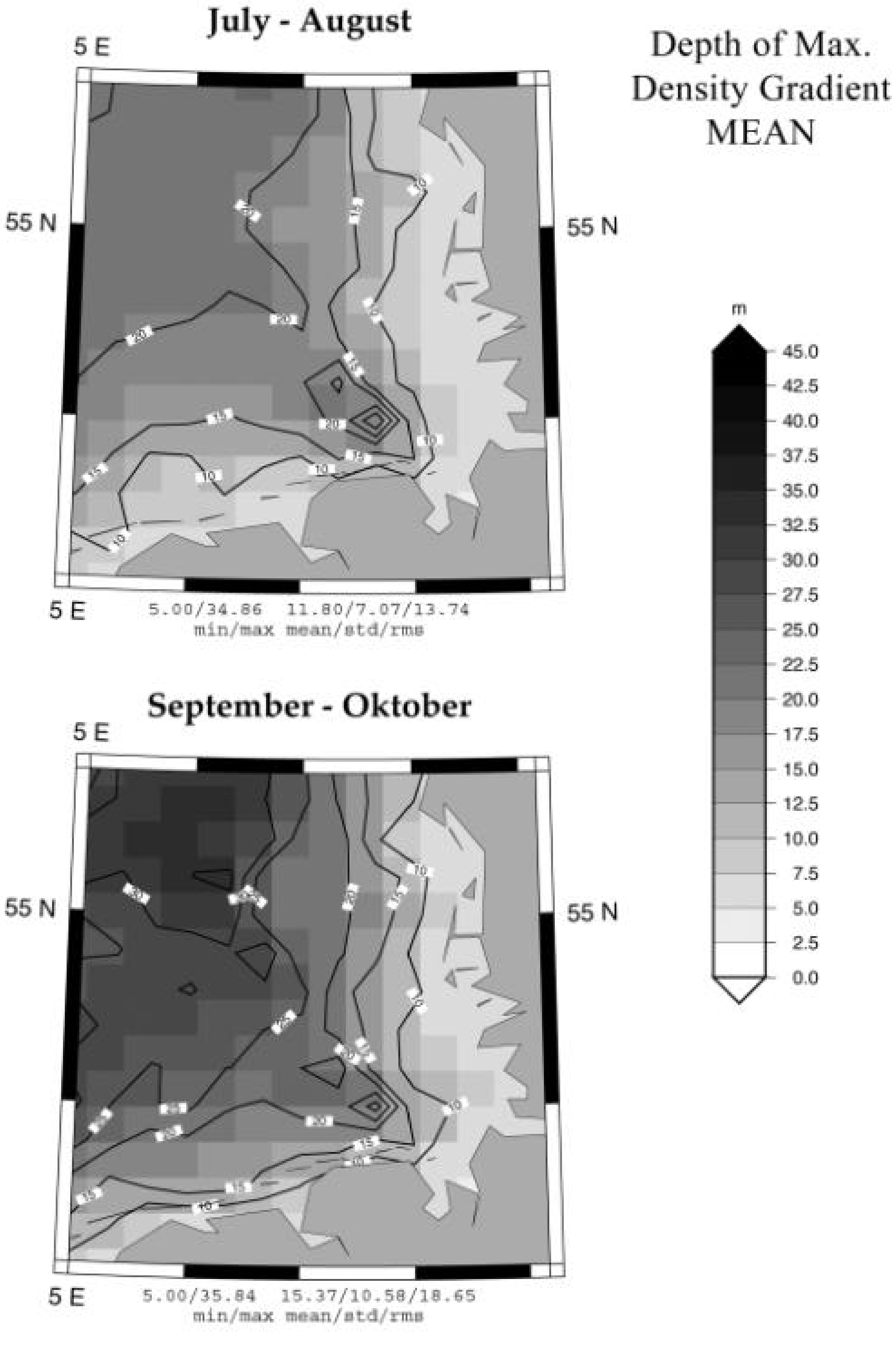

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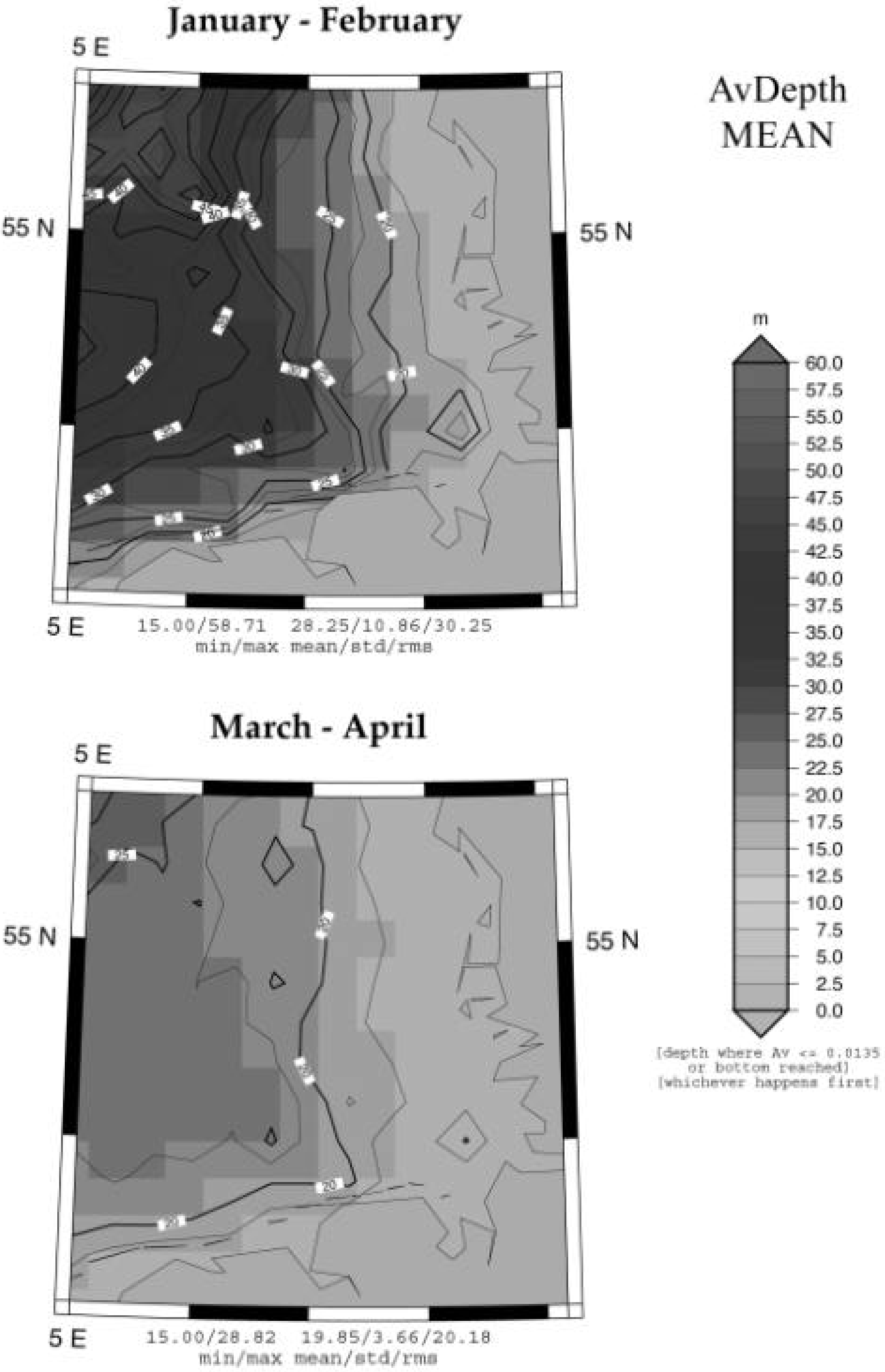

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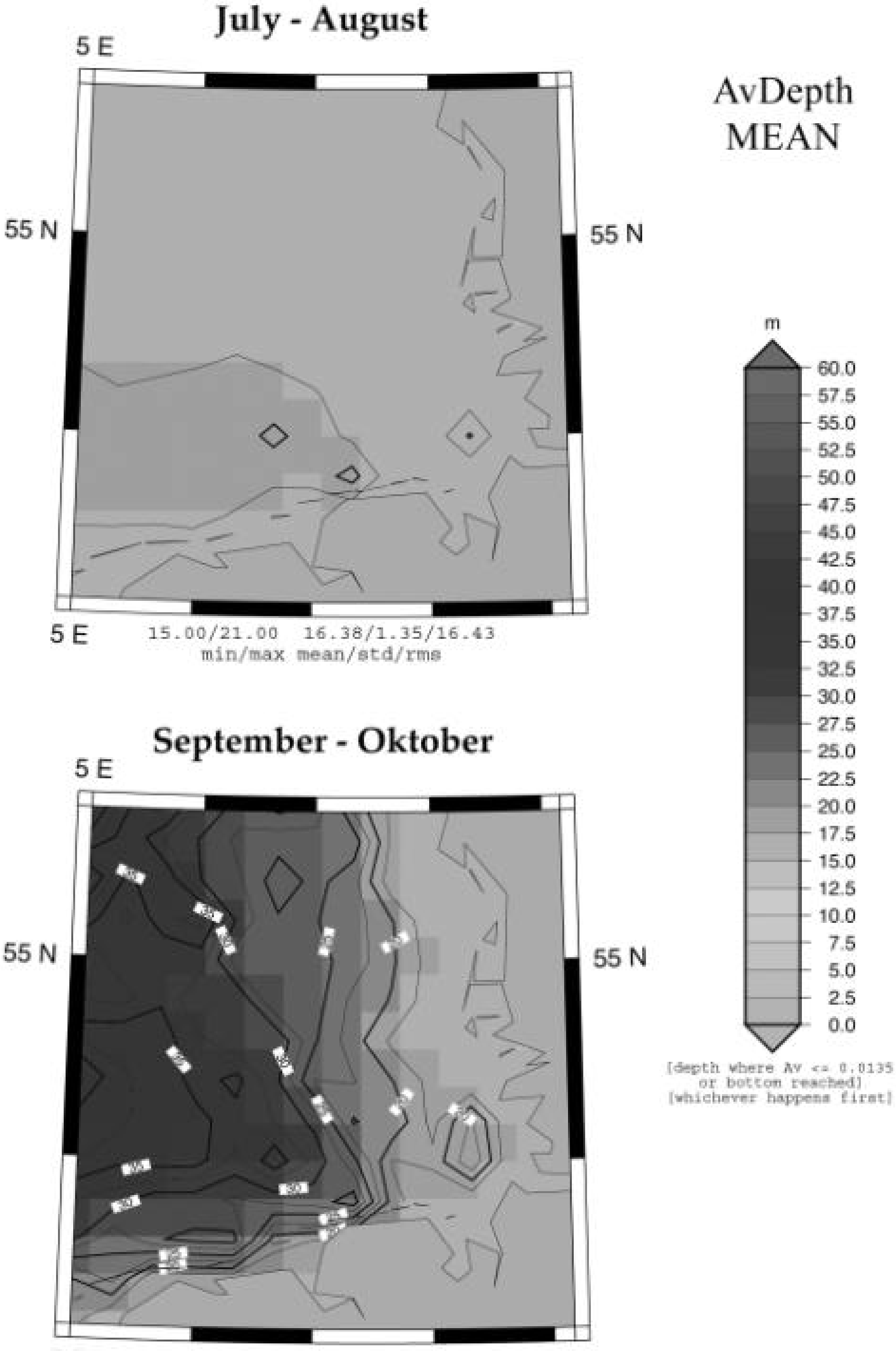

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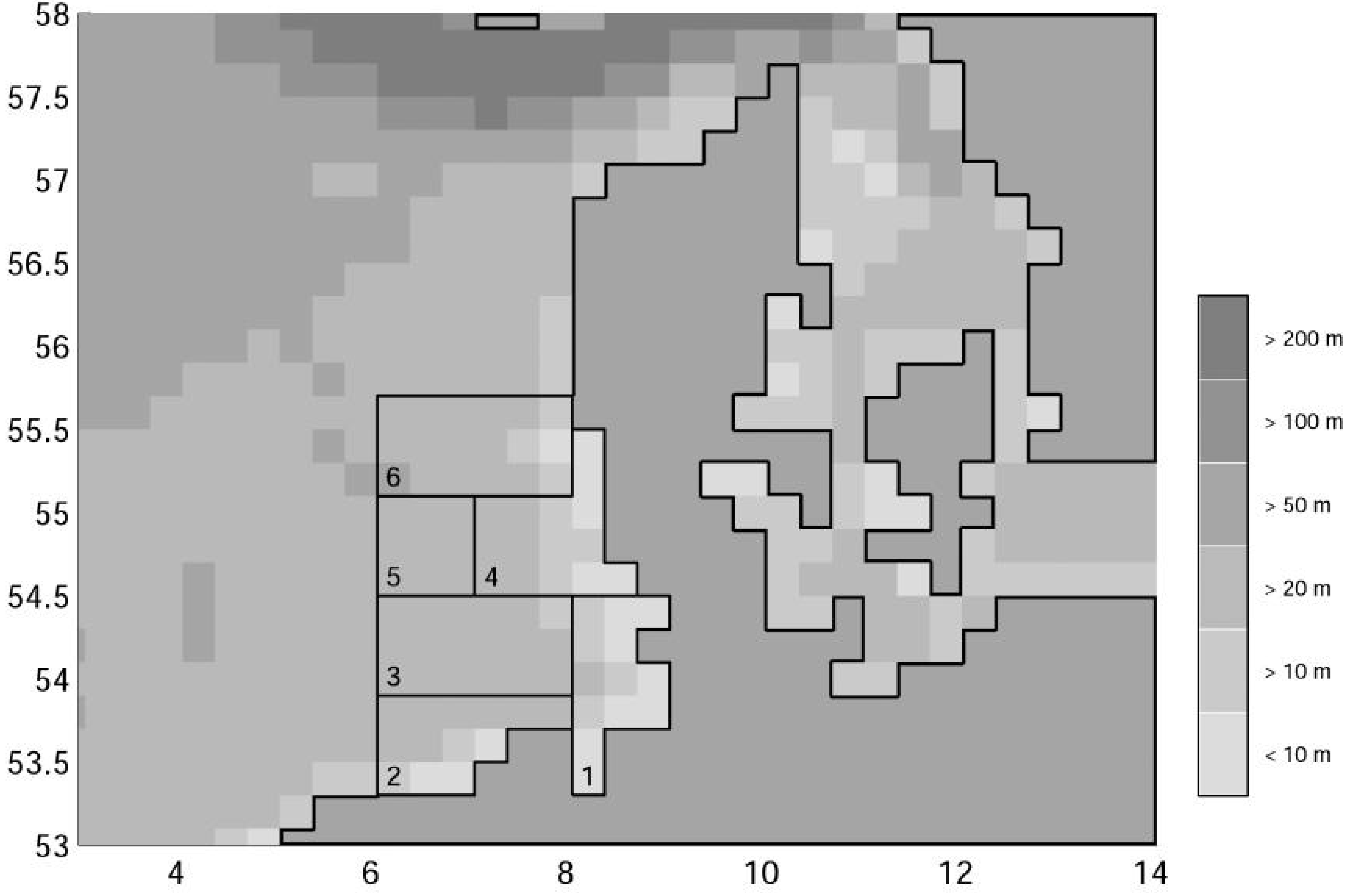


Fig. 2.1.2.1: Division of boxes in the ERSEM-model

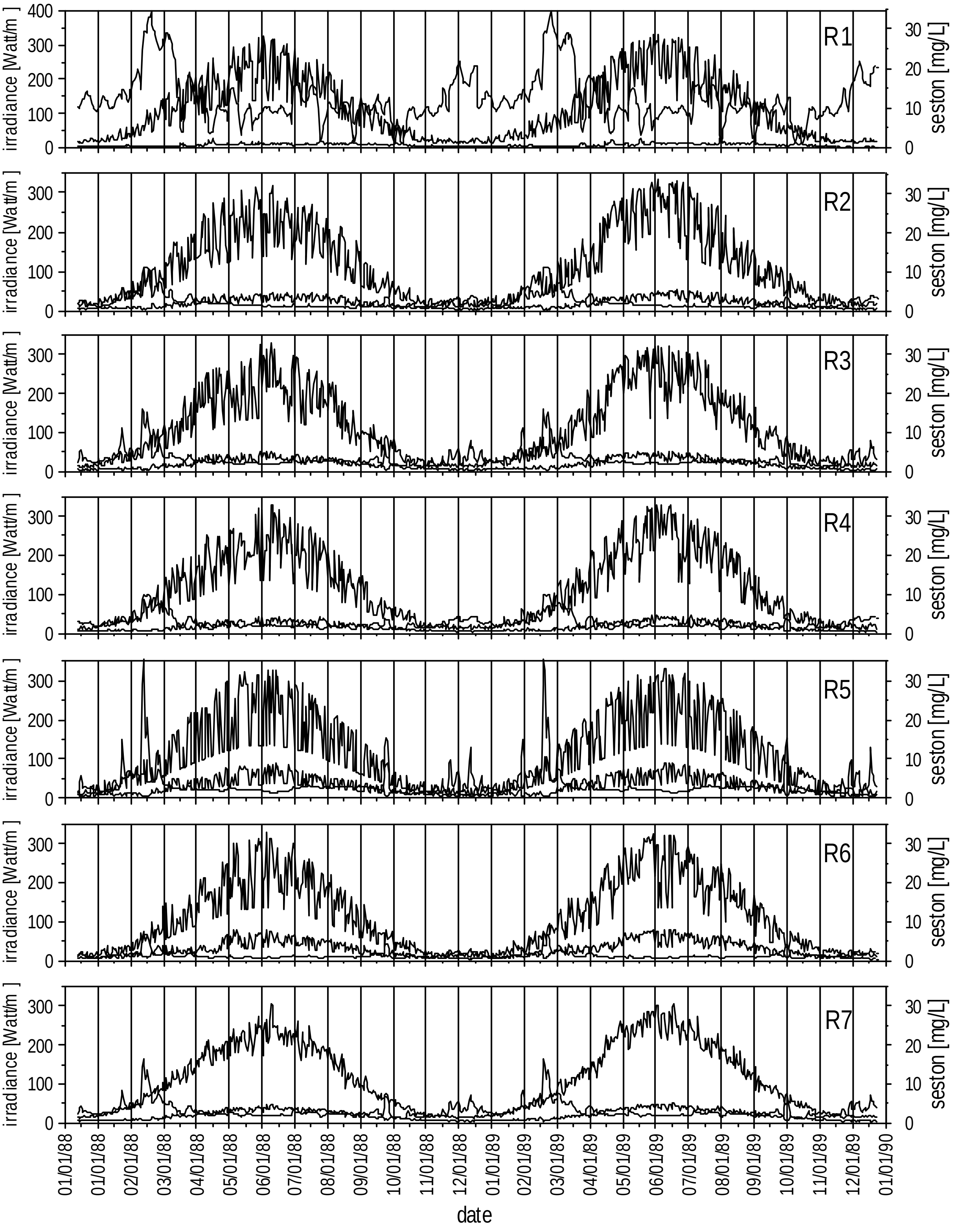


Fig. 2.1.3.1: Daily values (ERSEM) for irradiance [Watt/m²], underwater irradiance at 5m depth [Watt/m²] and seston [mg/L].
The German Bight is partitioned into 6 regions R1- R6; R7 identifies the whole German Bight.

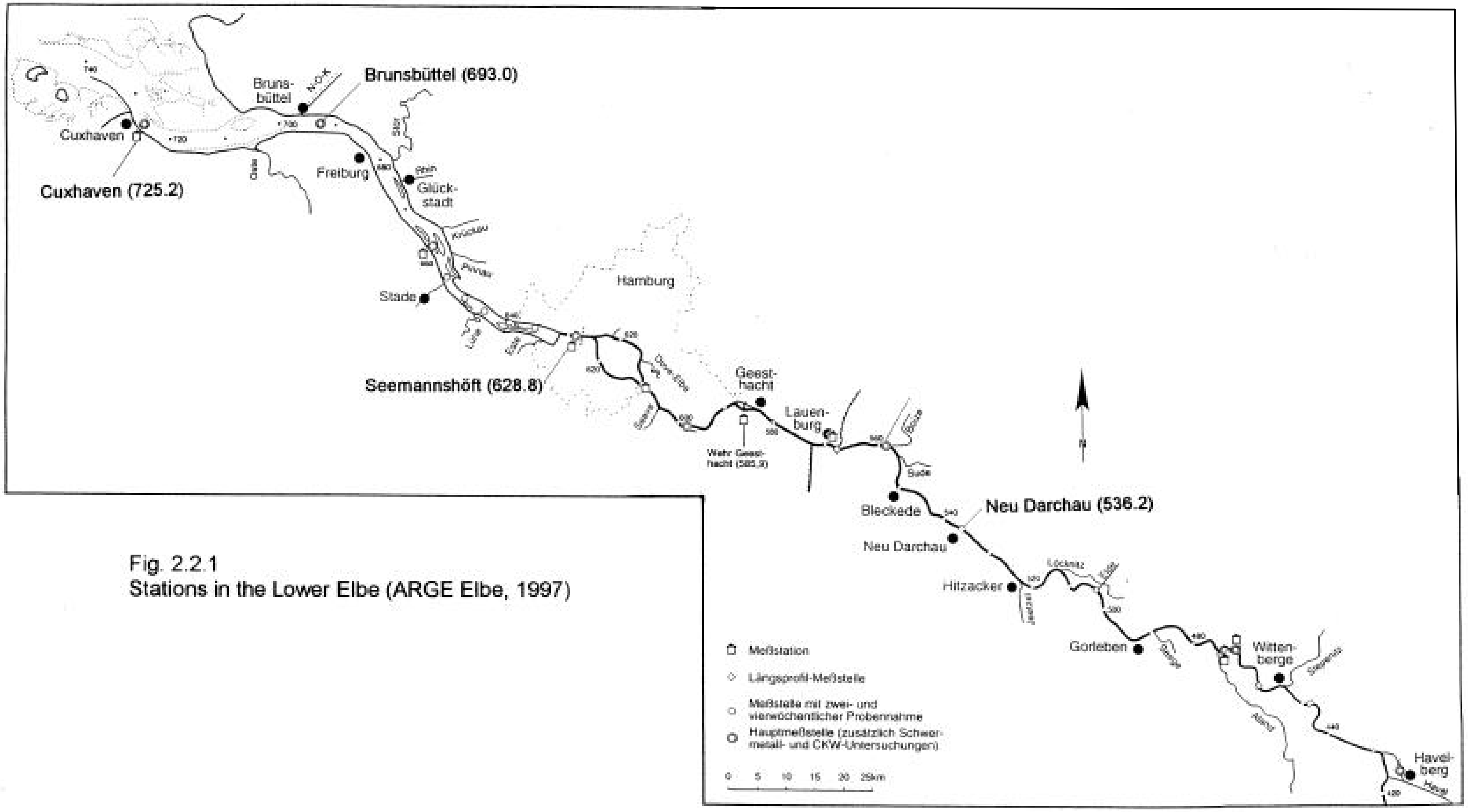


Fig. 2.2.1
Stations in the Lower Elbe (ARGE Elbe, 1997)

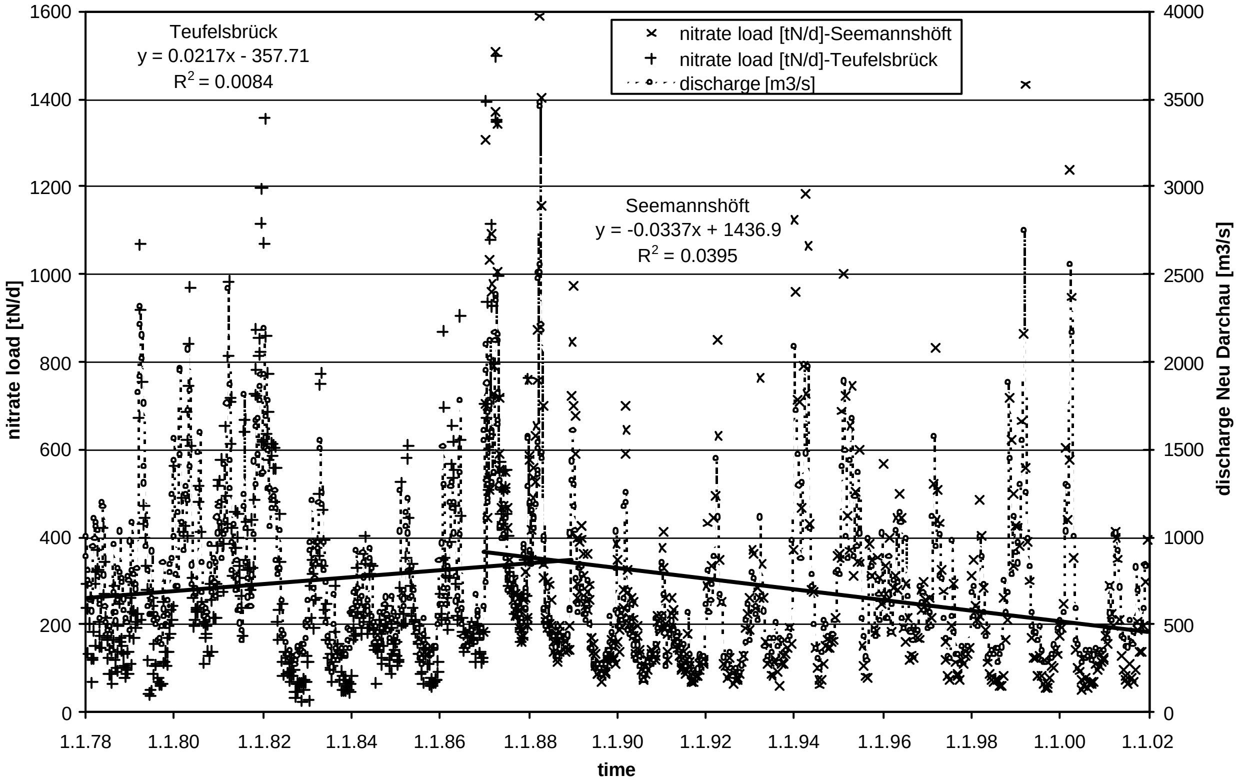


Fig. 2.2.2: Nitrate loads in the Elbe estuary at Teufelsbrück/Seemannshöft (km 628.8) and fresh water discharge at Neu Darchau (km 536.2) between 1978 and 2001 (ARGE Elbe)

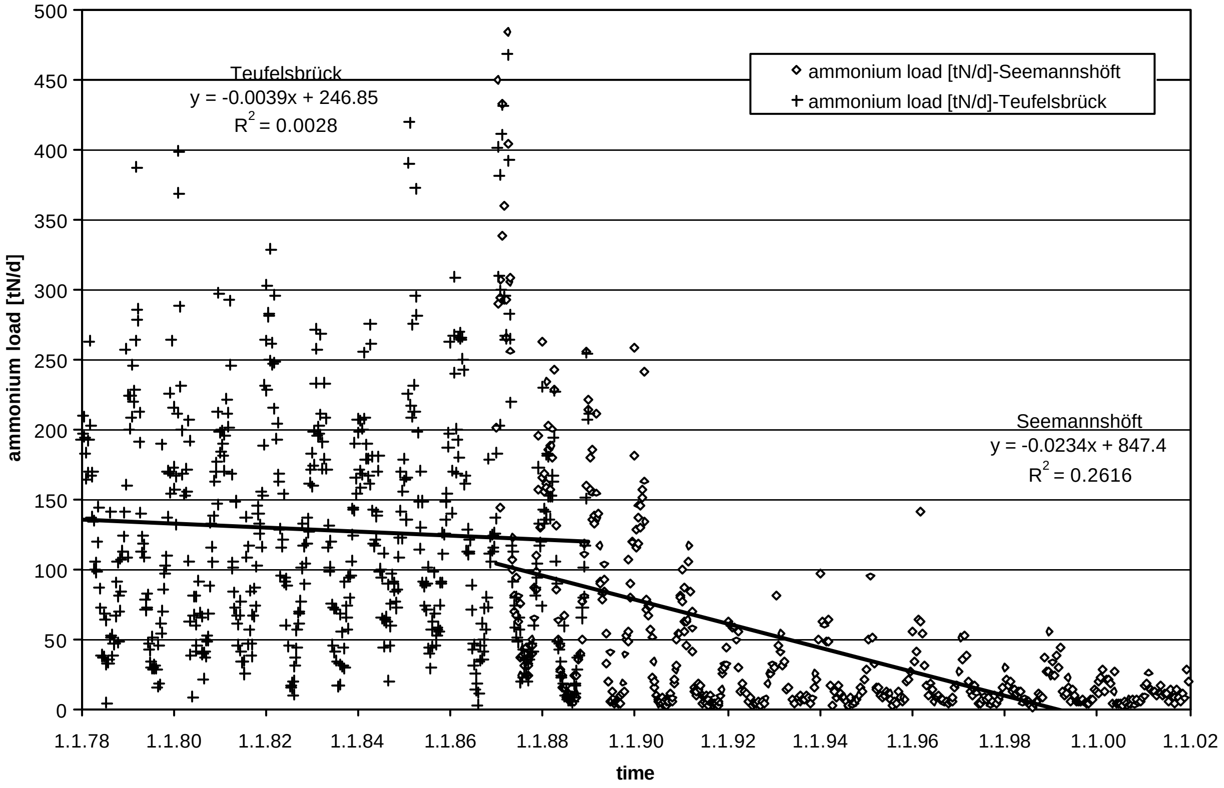


Fig. 2.2.3: Ammonium loads in the Elbe estuary at Teufelsbrück/Seemannshöft (km 628.8) from 1978 to 2001 (ARGE Elbe)

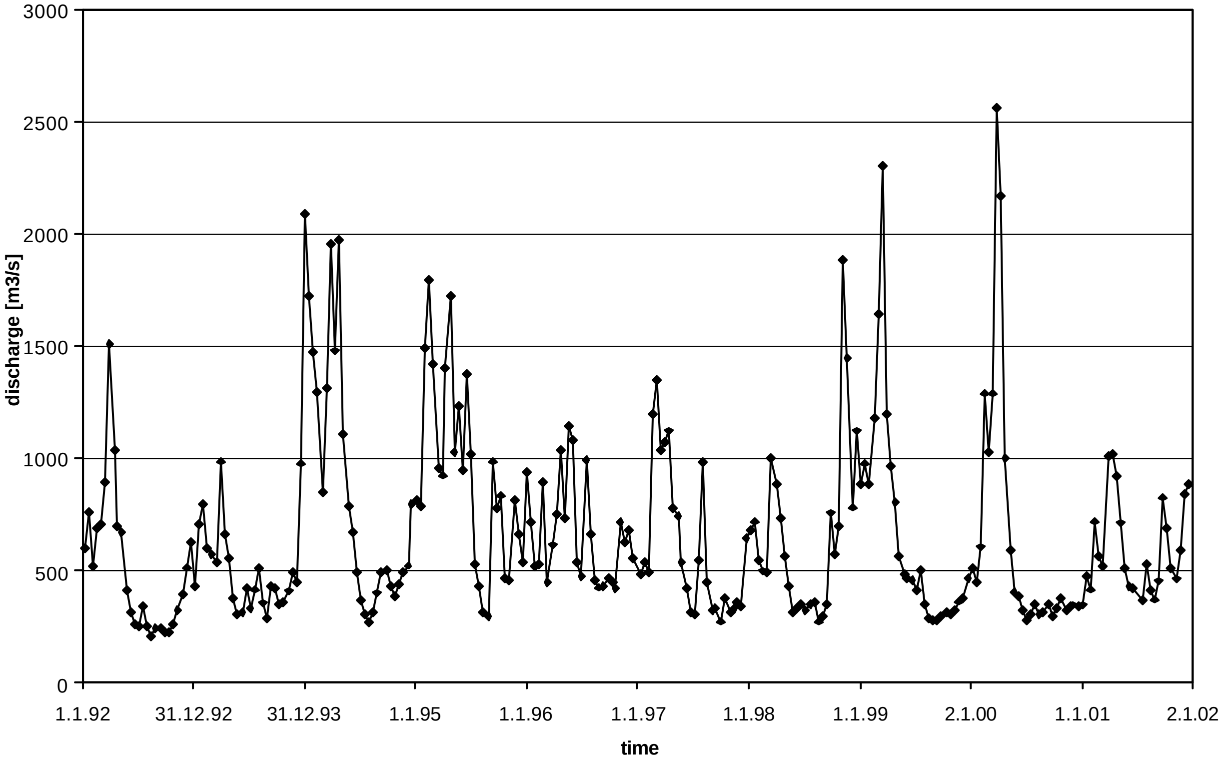


Fig. 2.2.4: Fresh water discharge rates (m³/s) at Neu Darchau (tide free gauge station, km 536.2) from 1992 to 2000 (ARGE Elbe)

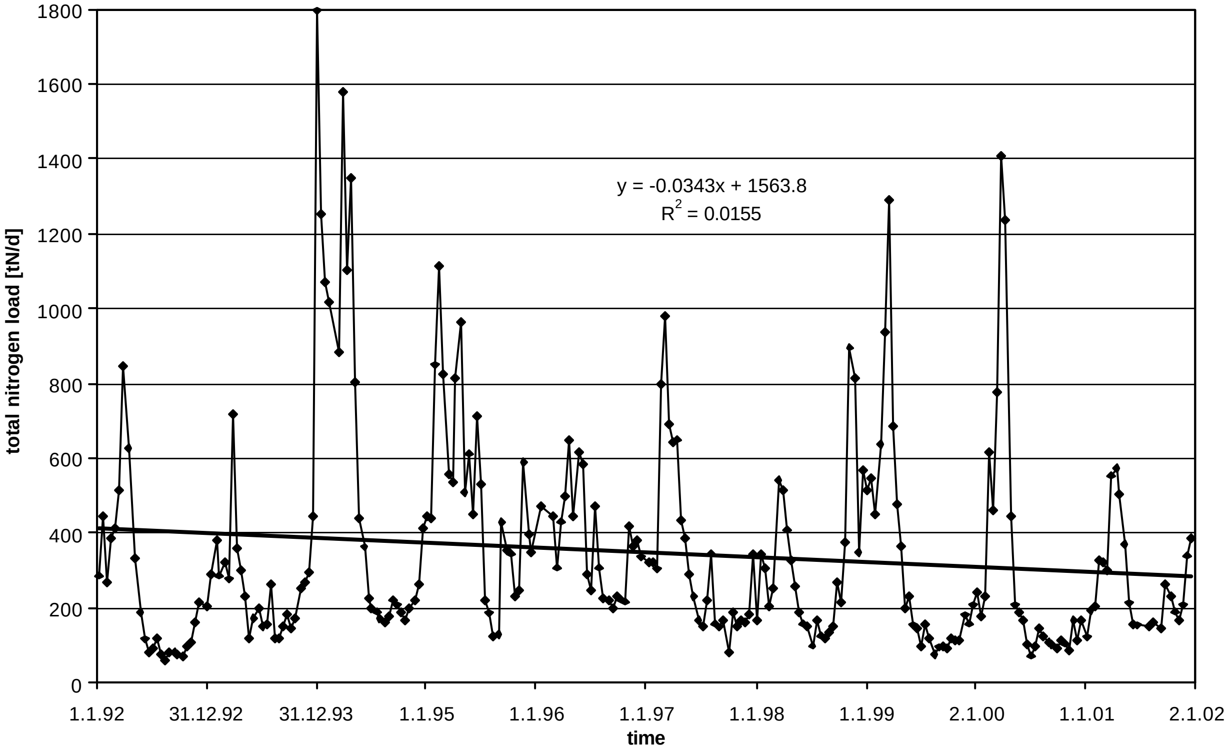


Fig. 2.2.5: Total nitrogen load (t N/d) at Cuxhaven (km 725.2) from 1992 to 2001 (ARGE Elbe)

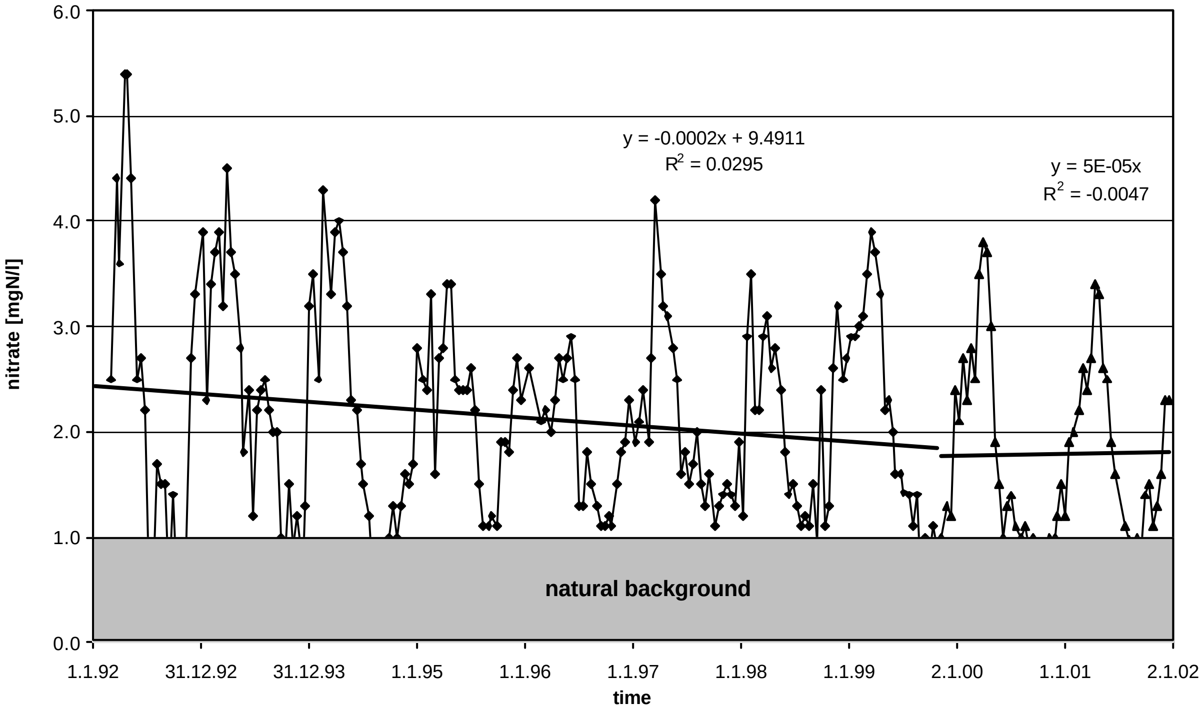


Fig. 2.2.6: Nitrate concentrations (mg N/L) at Cuxhaven (km 725.2) from 1992 to 2001

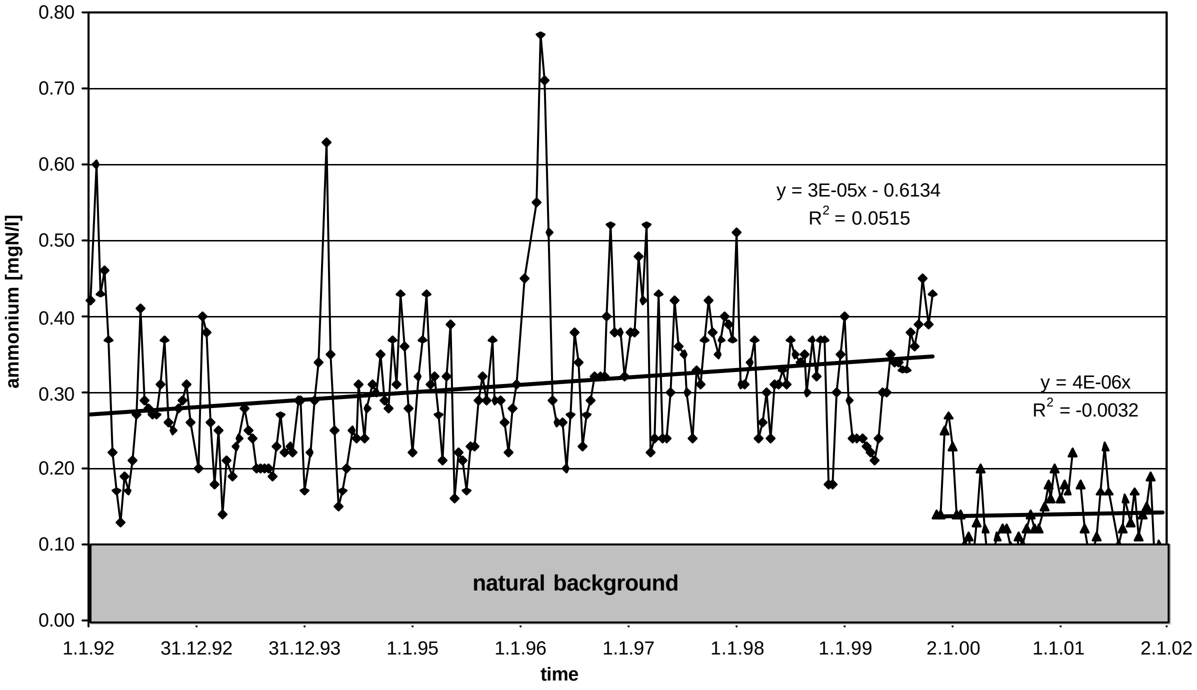


Fig. 2.2.7: Ammonium concentrations (mg N/L) at Cuxhaven (km 725.2) from 1992 to 2001 (ARGE Elbe)

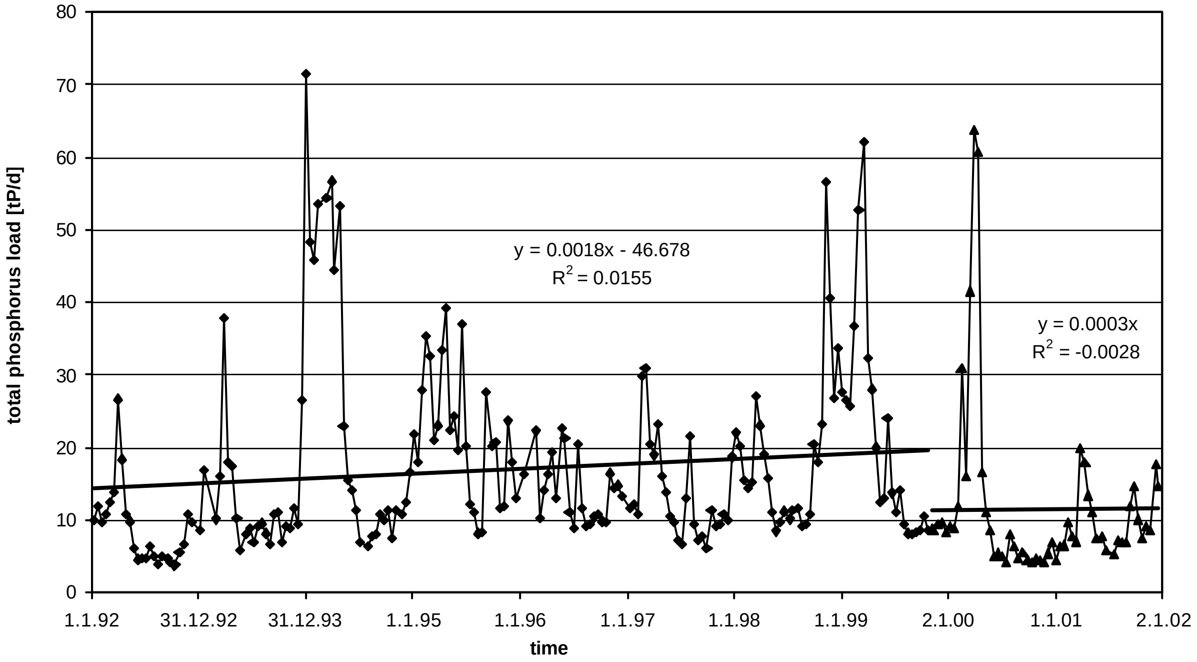


Fig. 2.2.8: Total phosphorus load (t P/d) at Cuxhaven (km 725.2) from 1992 to 2001 (ARGE Elbe)

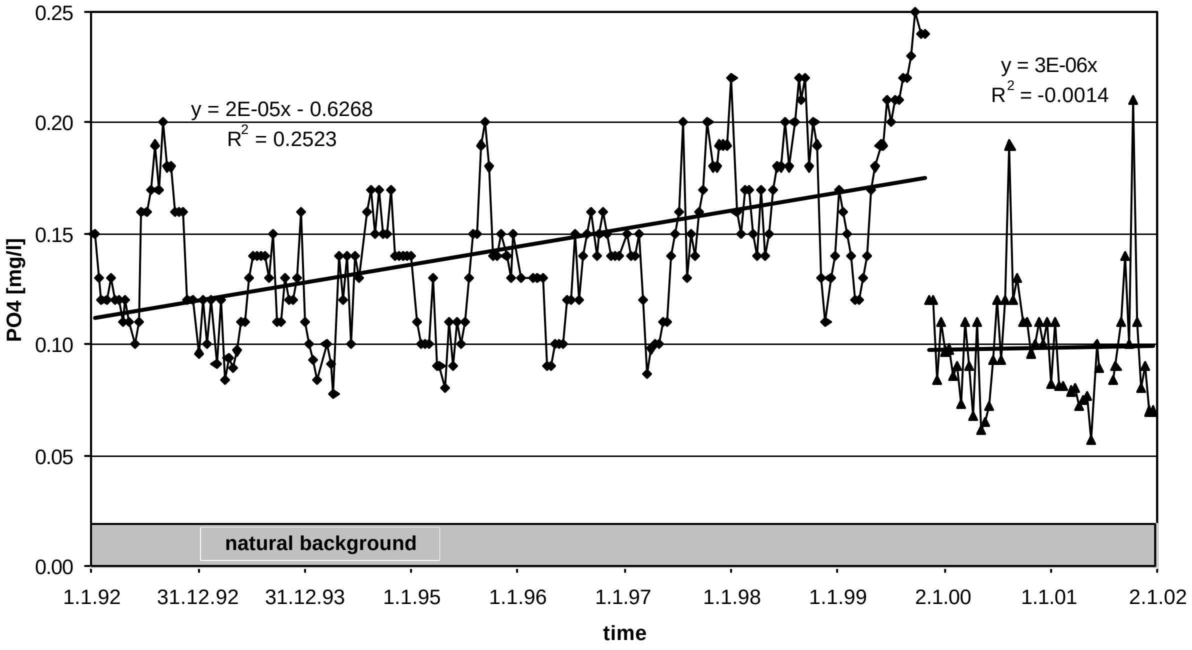


Fig. 2.2.9: Phosphate concentrations (mg P/L) at Cuxhaven (km 725.2) from 1992 to 2001 (ARGE Elbe)

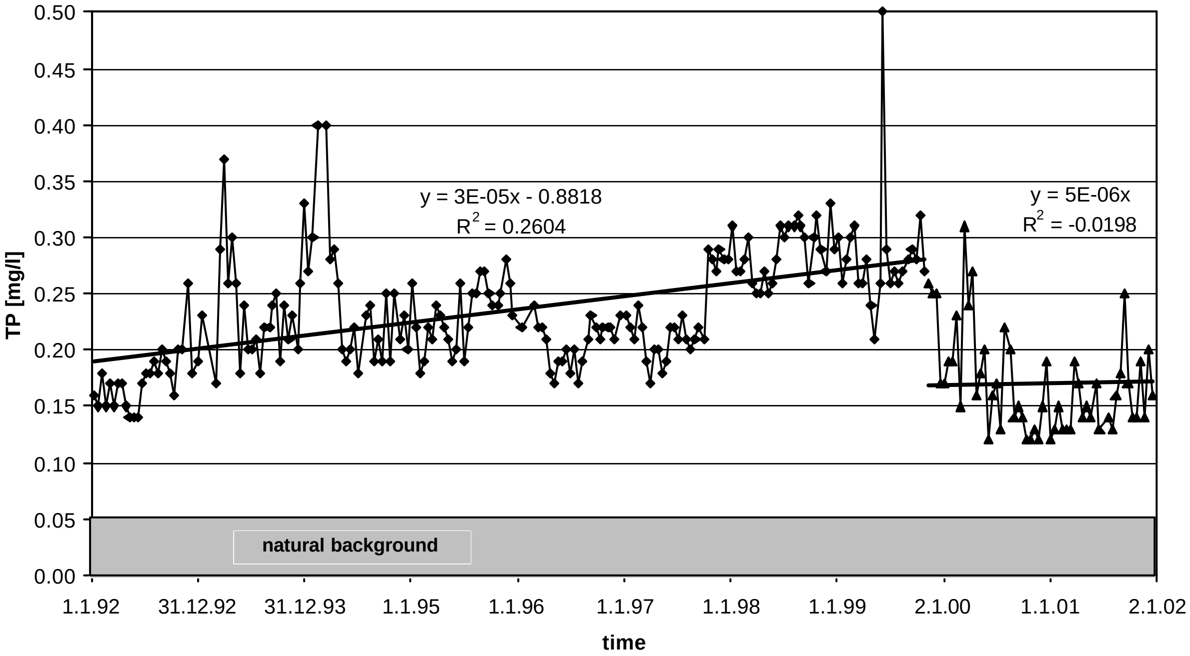


Fig. 2.2.10: Total phosphorus concentrations (mg P/L) at Cuxhaven (km 725.2) from 1992 to 2001 (ARGE Elbe)

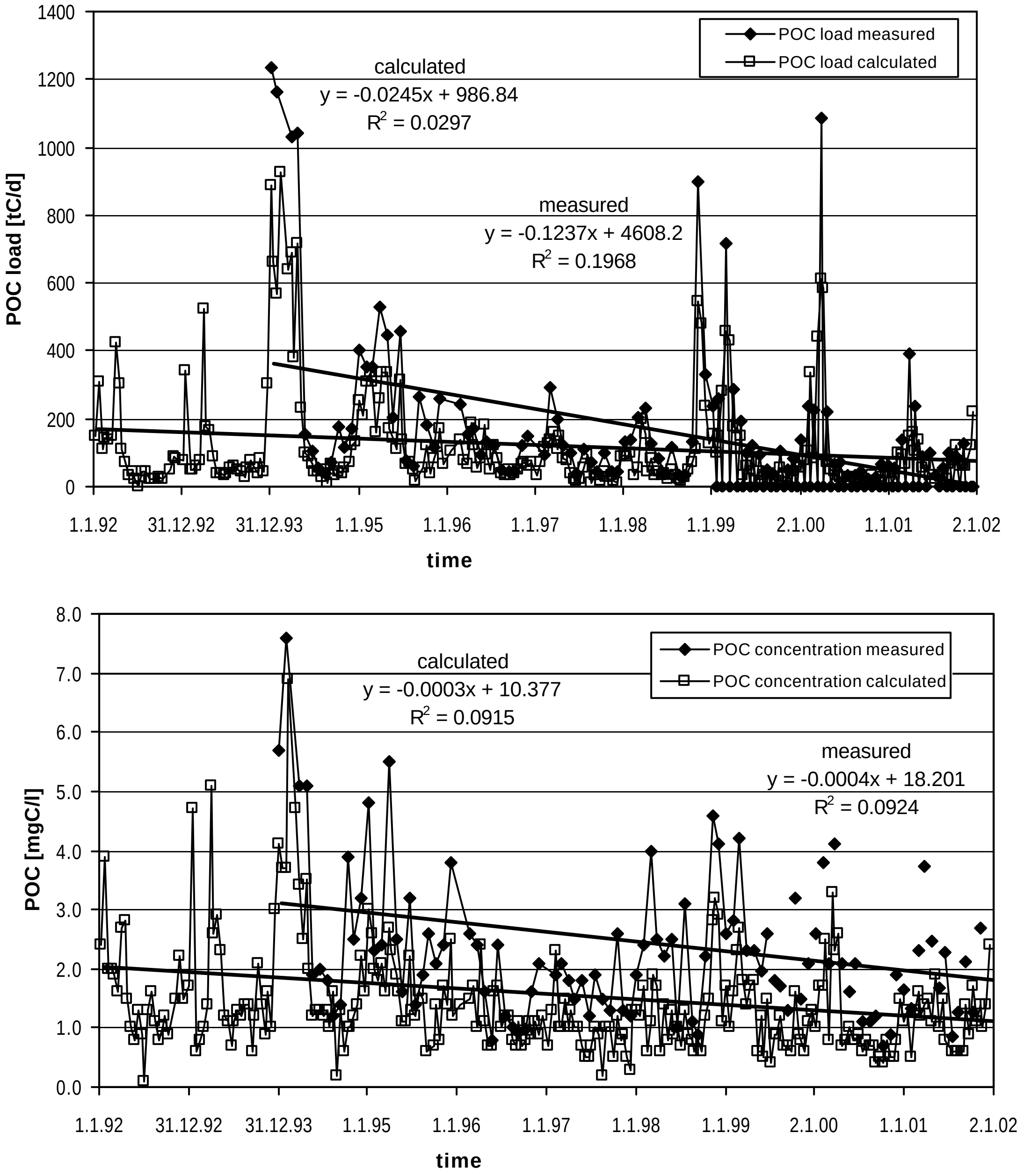


Fig. 2.2.11: Daily load (t C/d) and concentrations (mg C/l) of POC (particulate organic carbon) at Cuxhaven (km 725.2) [ARGE Elbe].

POC was directly measured from 1994 to 2001 and was calculated as difference of total and dissolved organic carbon since 1992

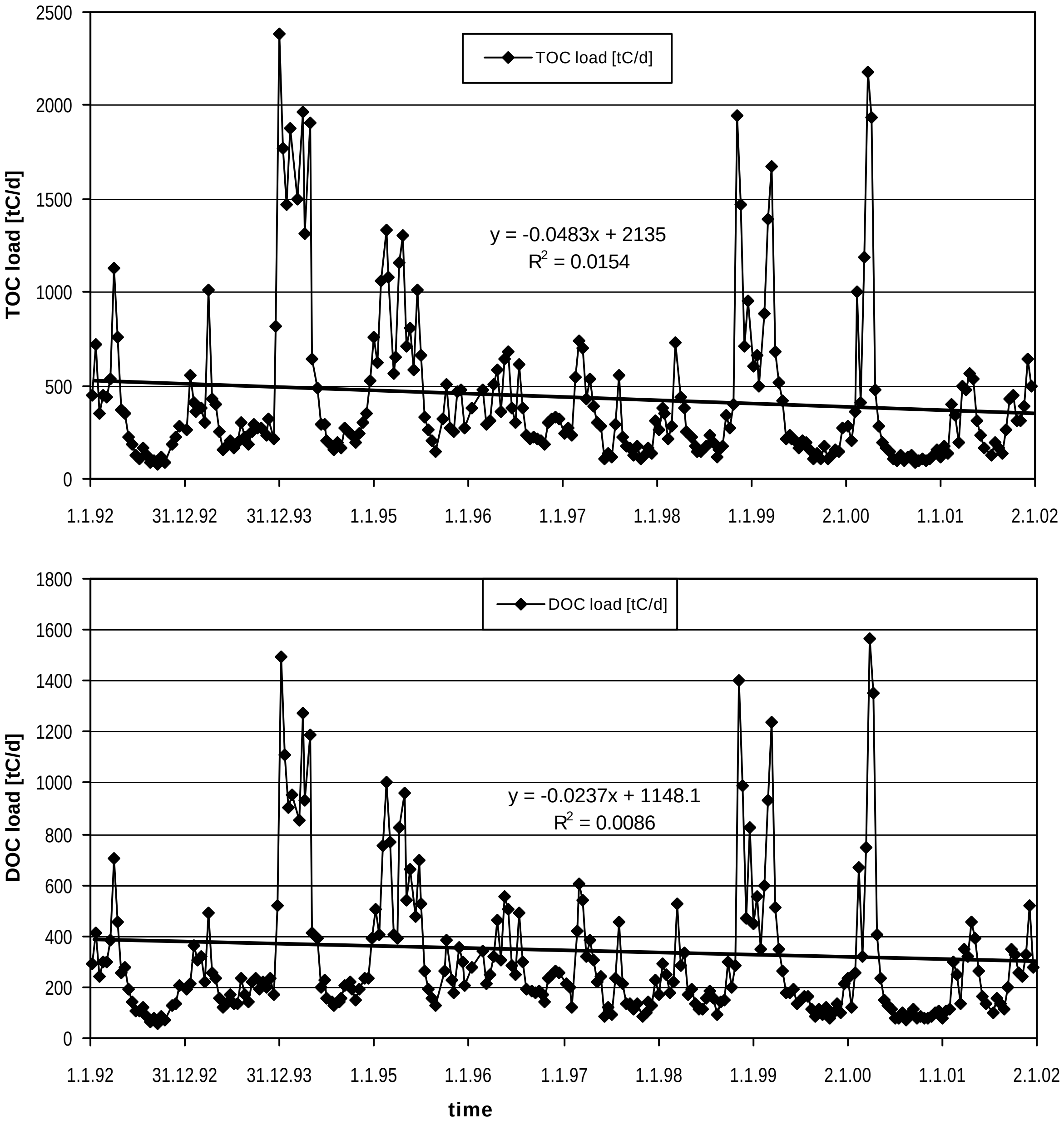


Fig. 2.2.12: Comparison of loads of total (TOC) and dissolved (DOC) organic carbon (t C/d) at Cuxhaven (km 725.2) [ARGE Elbe]

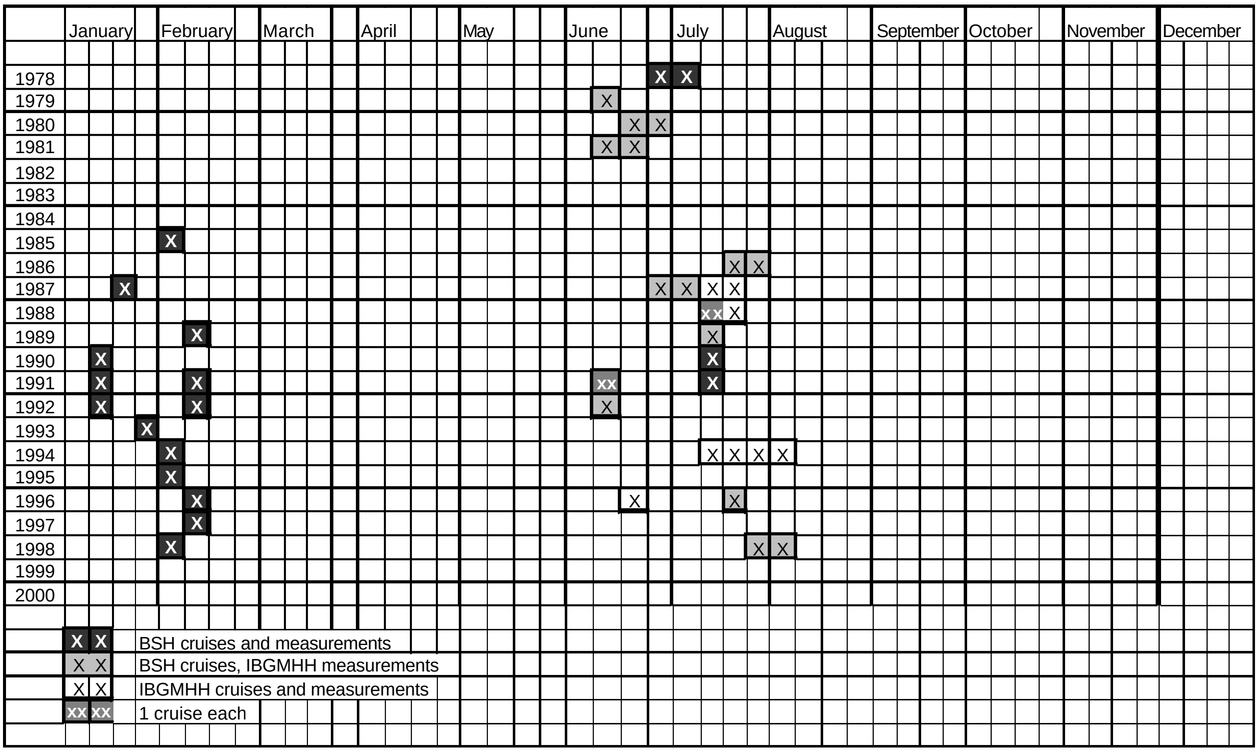
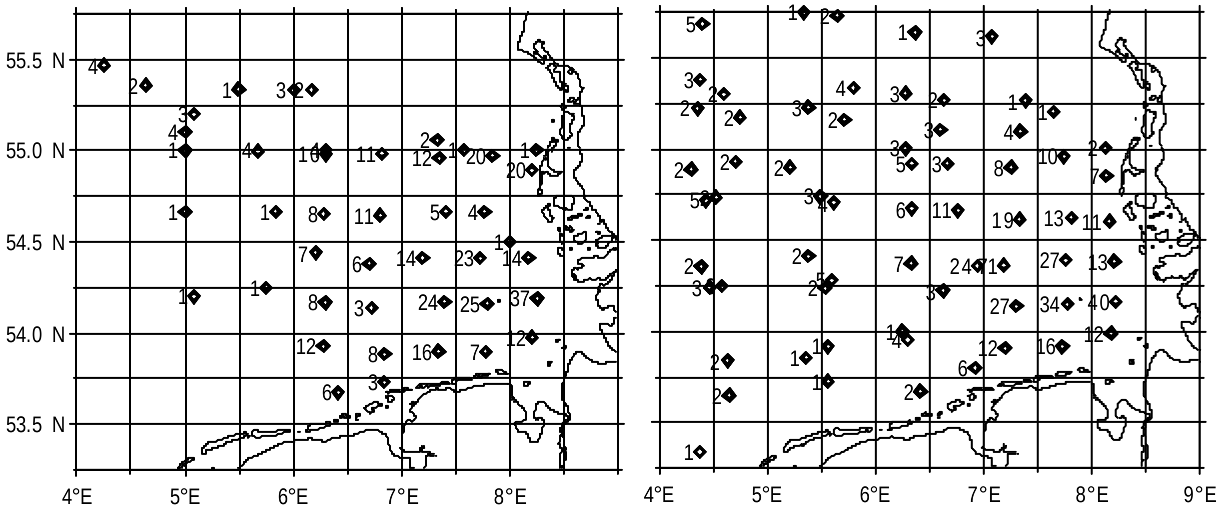


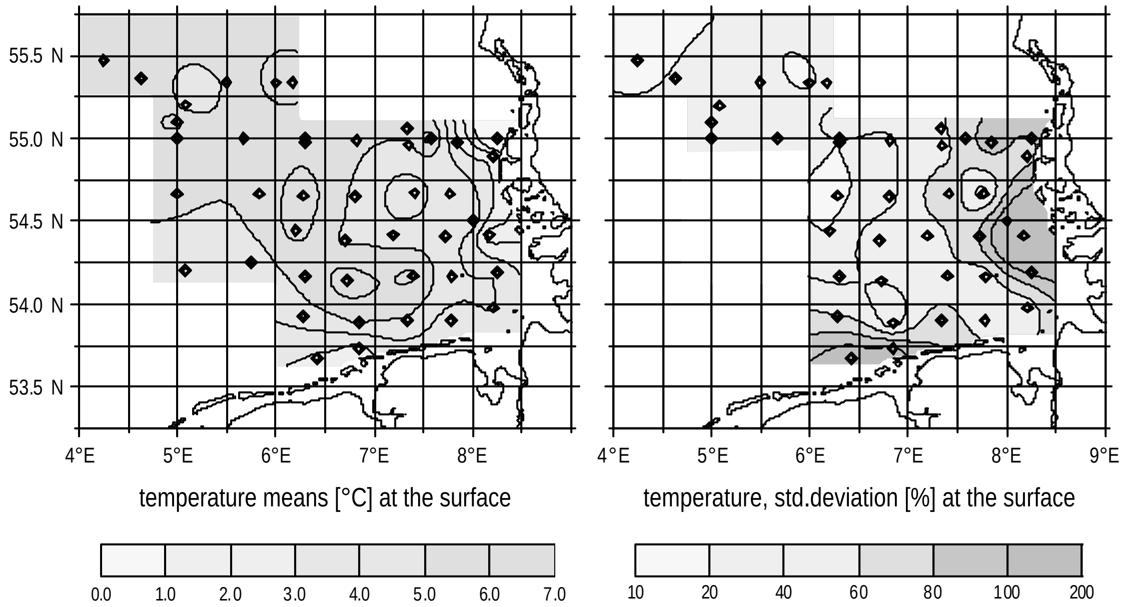
Fig. 2.3.0.1: Time schedule of winter and summer cruises



German Bight in January and Februay 1985 - 1998

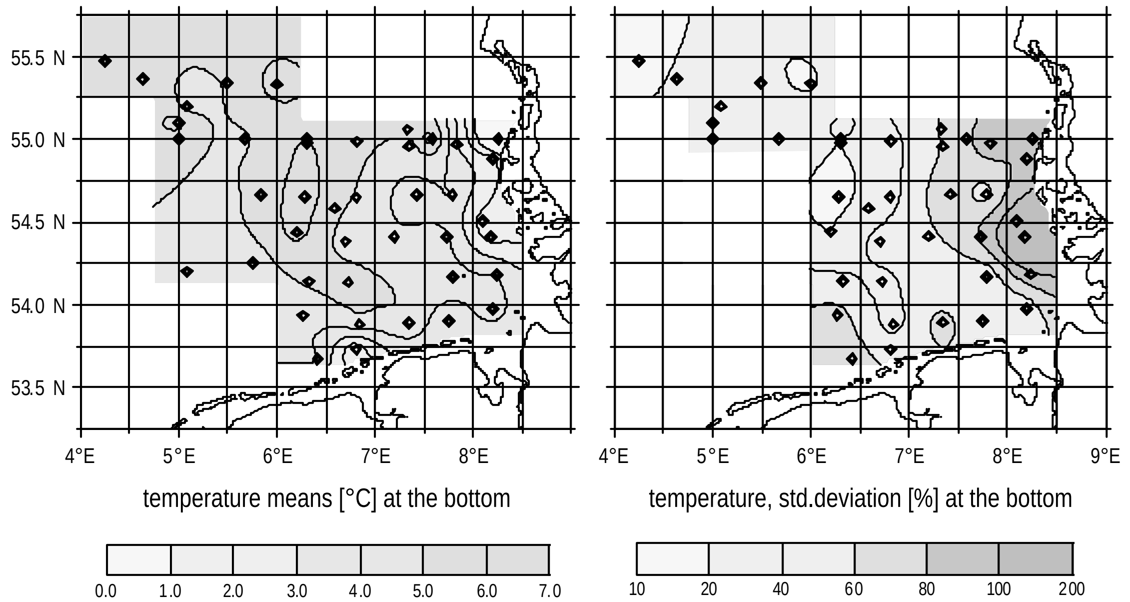
German Bight in June and July 1985 - 1998

Fig. 2.3.0.2: Station net with number of stations per quadrant used for calcuation of means and standard deviations during winter and summer



temperature means [°C] at the surface

temperature, std.deviation [%] at the surface

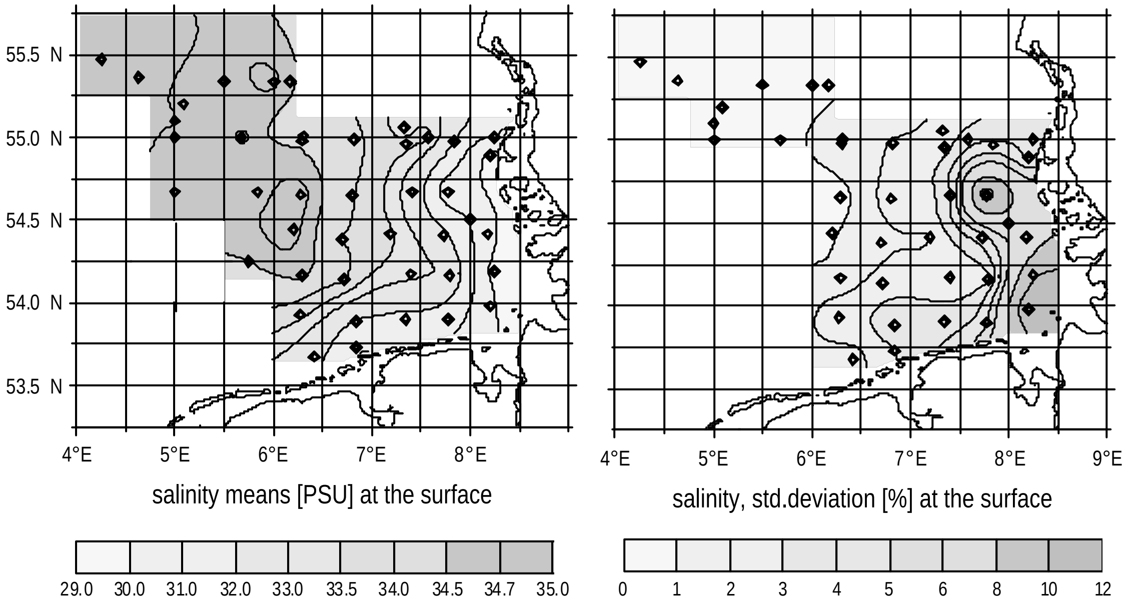


temperature means [°C] at the bottom

temperature, std.deviation [%] at the bottom

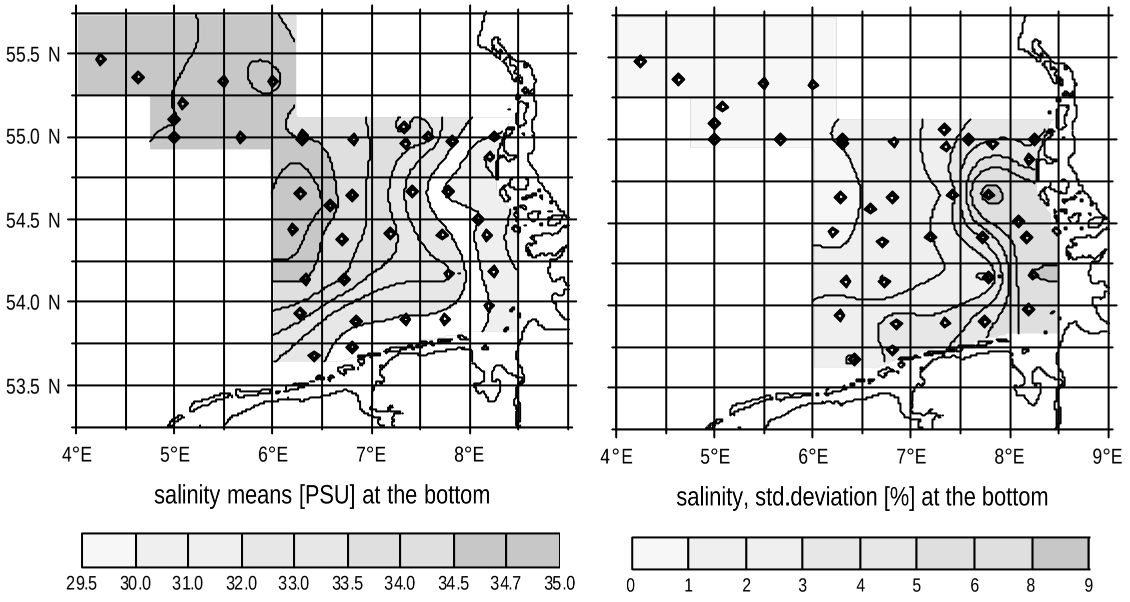
The German Bight in January and February 1985 - 1998

Fig. 2.3.1.1: Mean winter temperature gradients [°C] and their variability [%]



salinity means [PSU] at the surface

salinity, std.deviation [%] at the surface

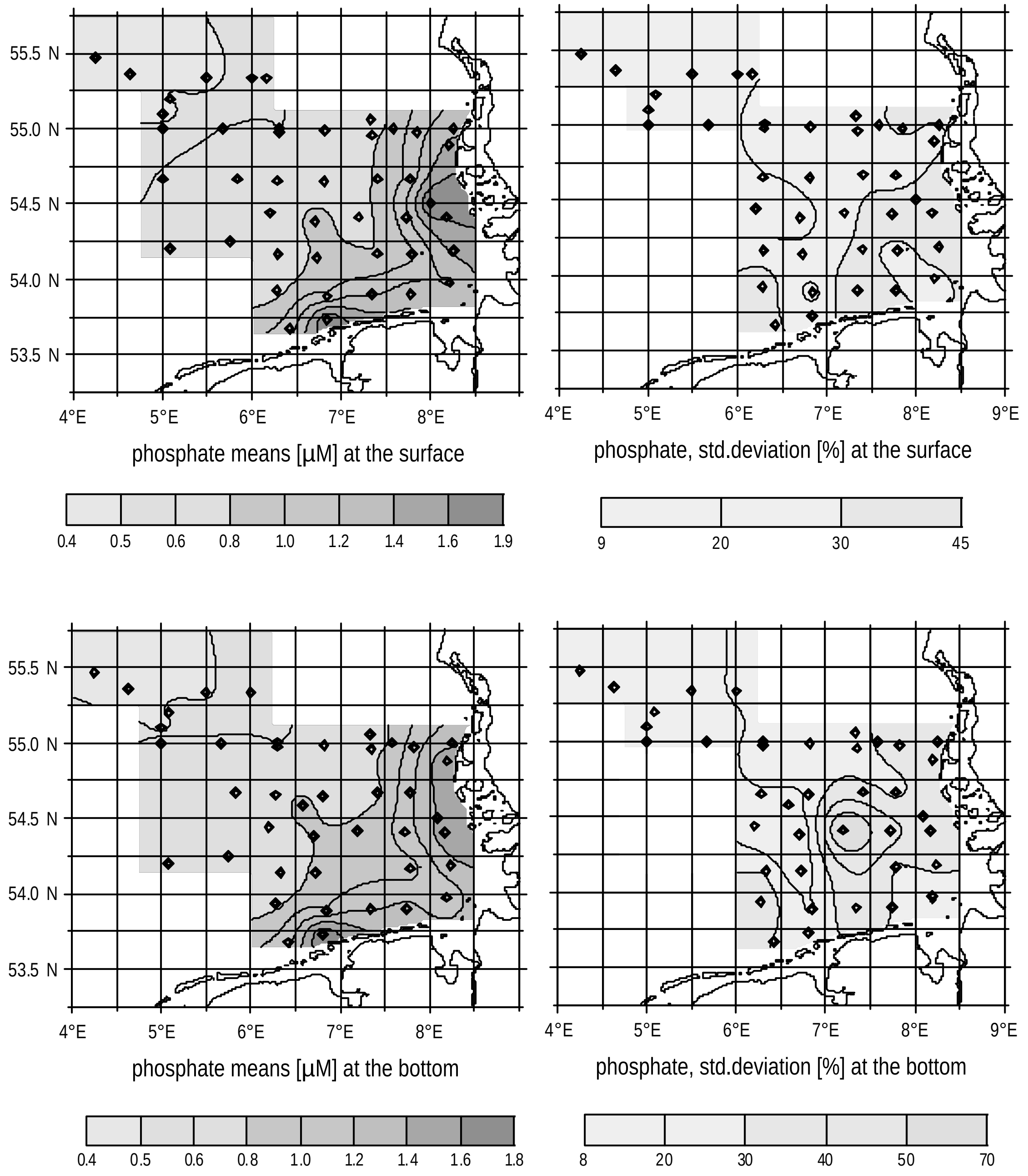


salinity means [PSU] at the bottom

salinity, std.deviation [%] at the bottom

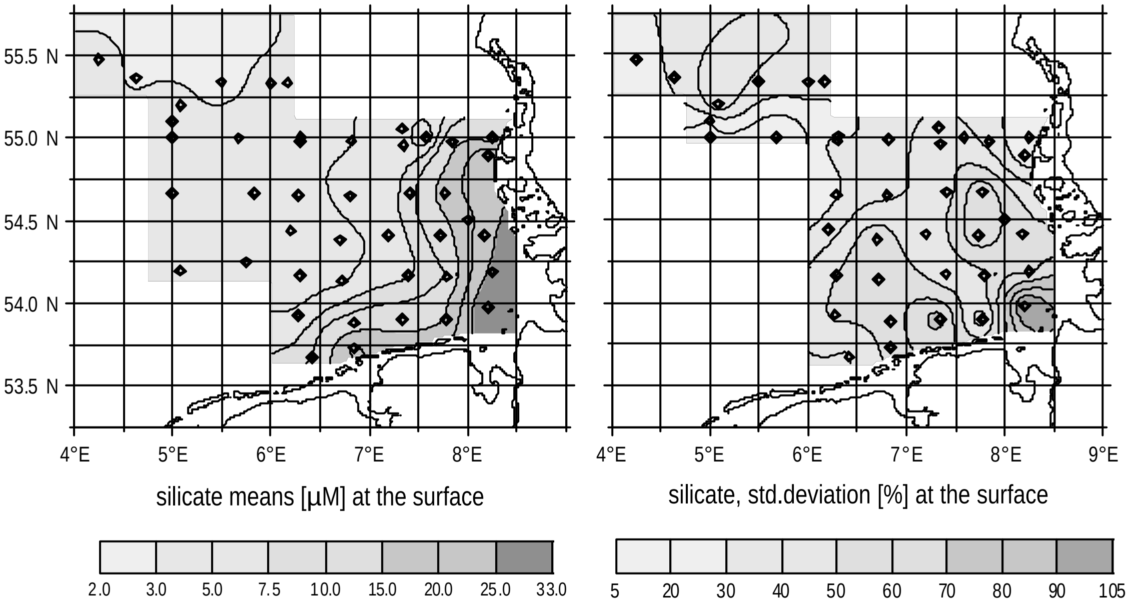
The German Bight in January and February 1985 - 1998

Fig. 2.3.1.2: Mean winter salinity gradients and their variability [%]



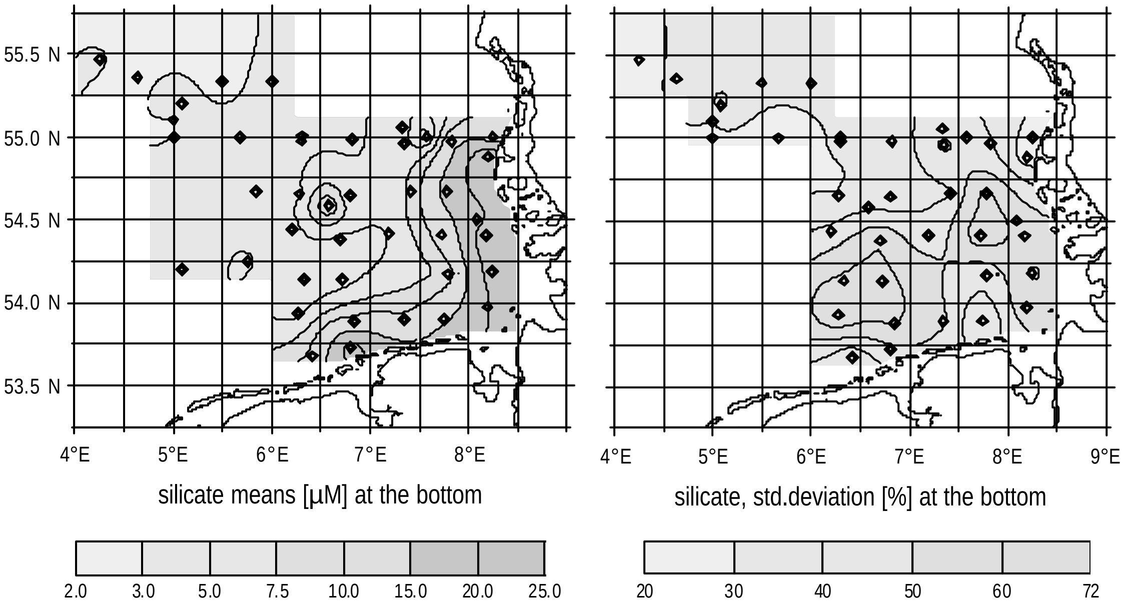
The German Bight in January and February 1985 - 1998

Fig. 2.3.1.3: Mean winter phosphate gradients [µM] and their variability [%]



silicate means [µM] at the surface

silicate, std.deviation [%] at the surface

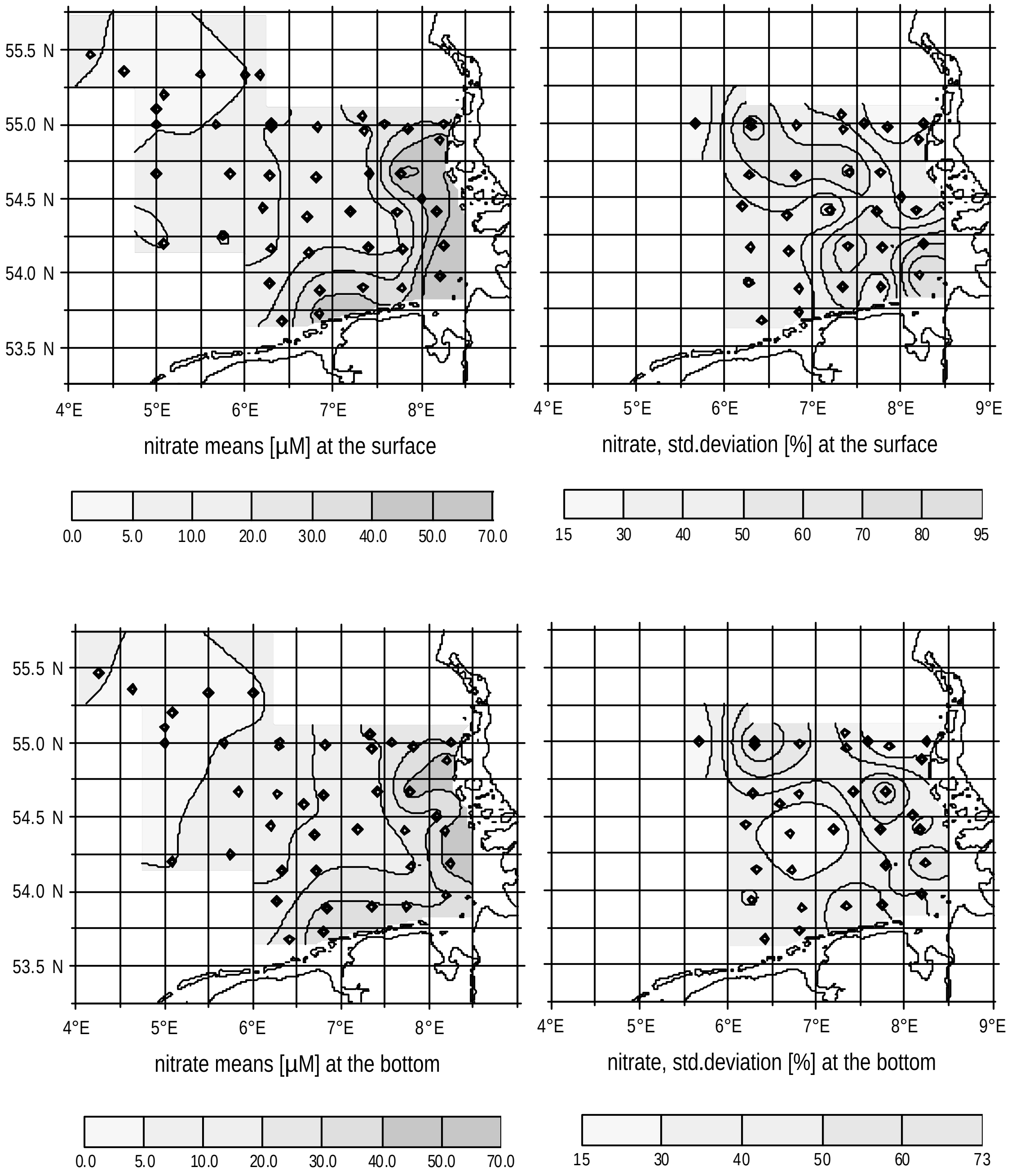


silicate means [µM] at the bottom

silicate, std.deviation [%] at the bottom

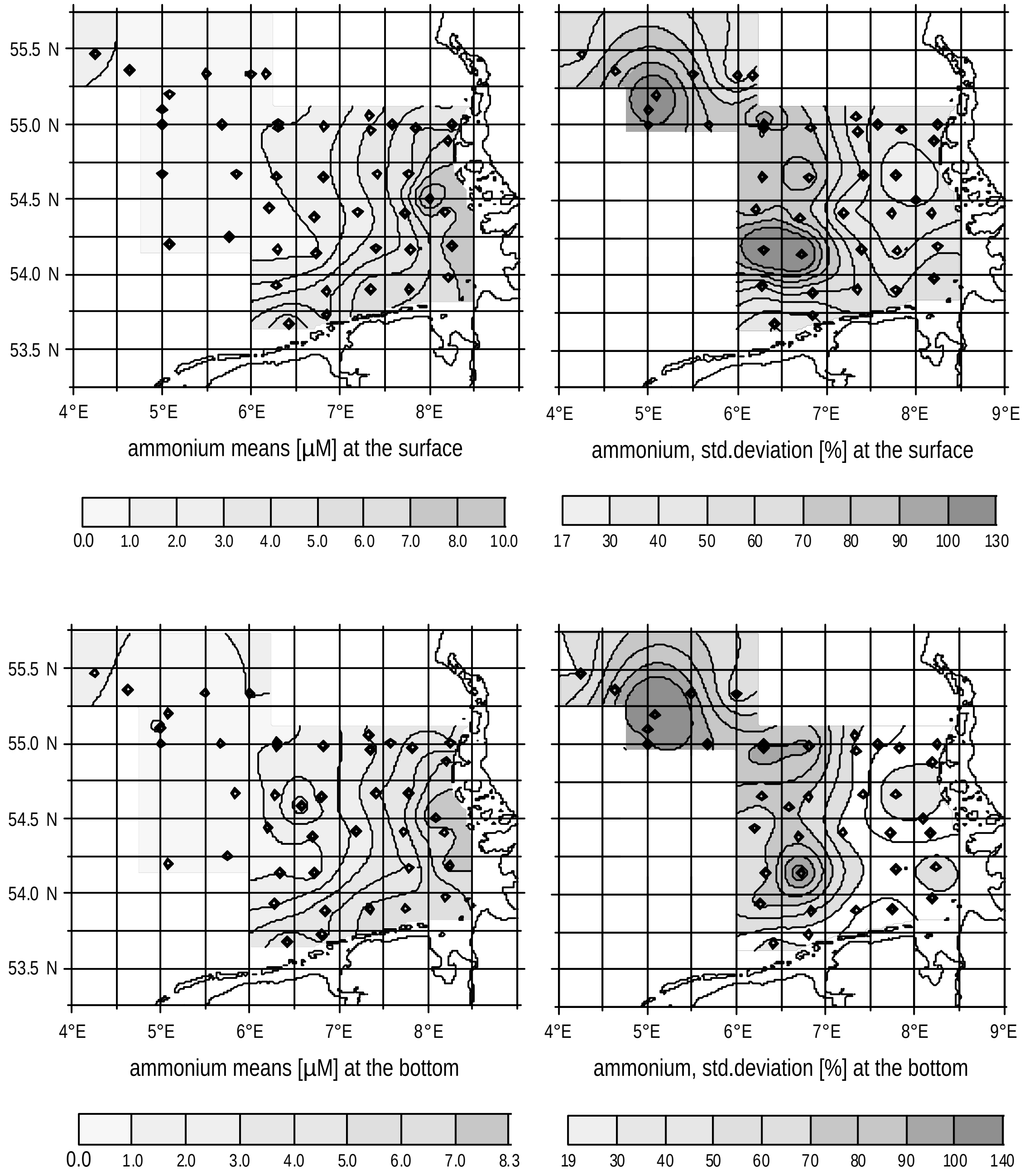
The German Bight in January and February 1985 - 1998

Fig. 2.3.1.4: Mean winter silicate [µM] gradients and their variability [%]



The German Bight in January and February 1985 - 1998

Fig. 2.3.1.5: Mean winter nitrate gradients [µM] and their variability [%]



ammonium means [µM] at the surface

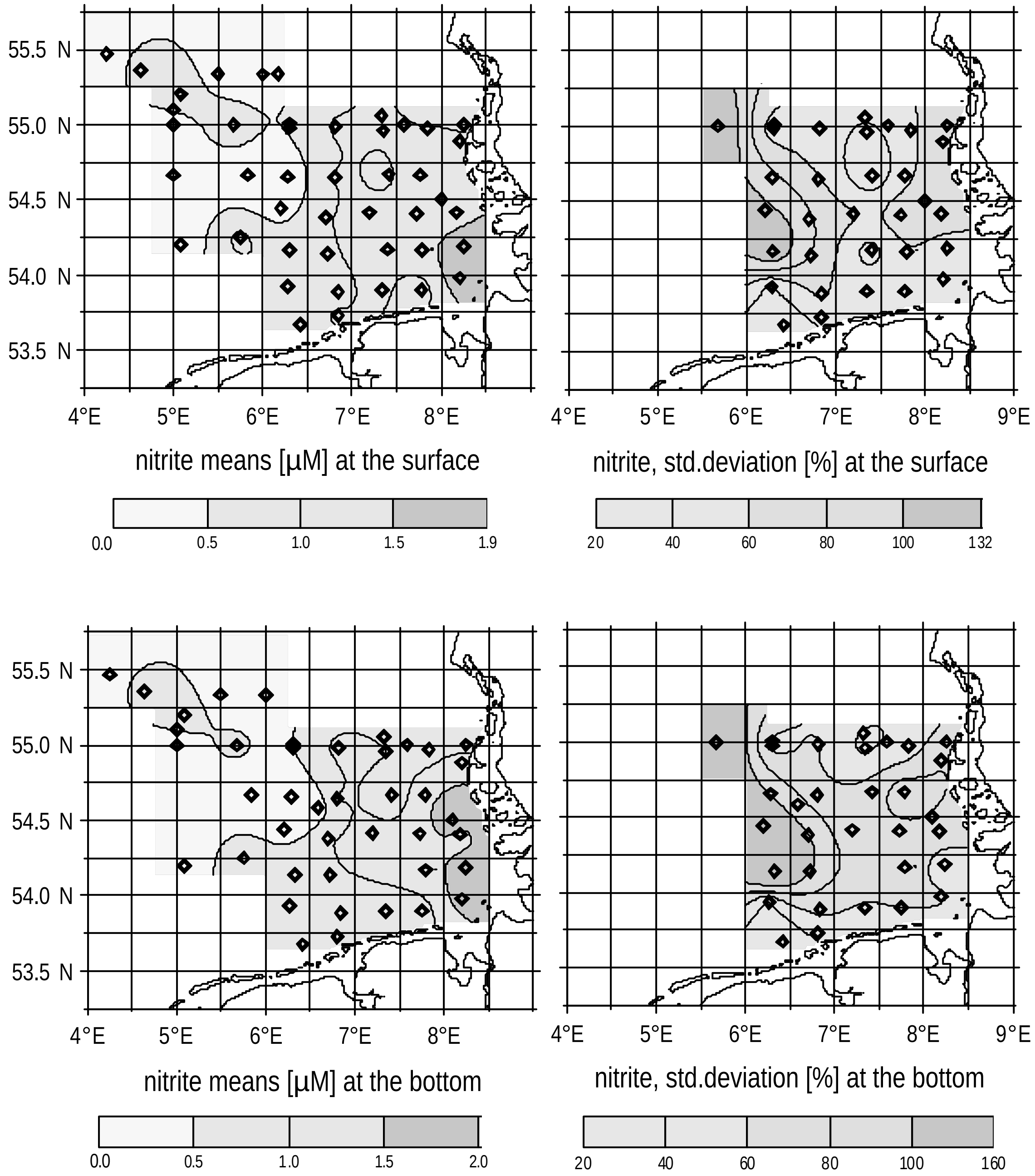
ammonium, std.deviation [%] at the surface

ammonium means [µM] at the bottom

ammonium, std.deviation [%] at the bottom

The German Bight in January and February 1985 - 1998

Fig. 2.3.1.6: Mean winter ammonium gradients [µM] and their variability [%]



nitrite means [µM] at the surface

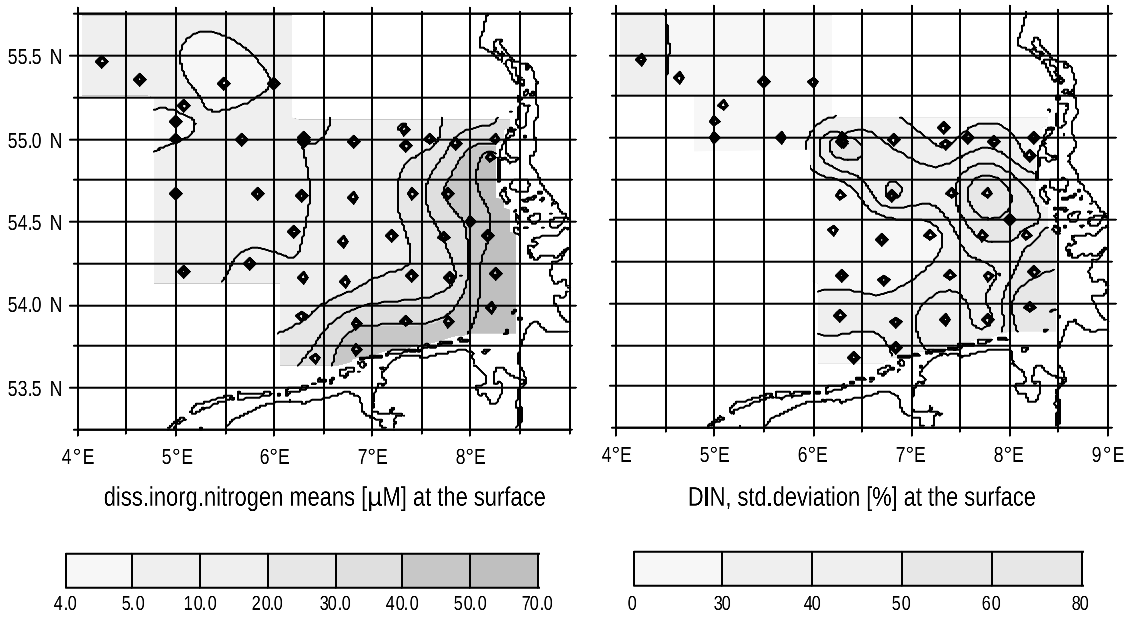
nitrite, std.deviation [%] at the surface

nitrite means [µM] at the bottom

nitrite, std.deviation [%] at the bottom

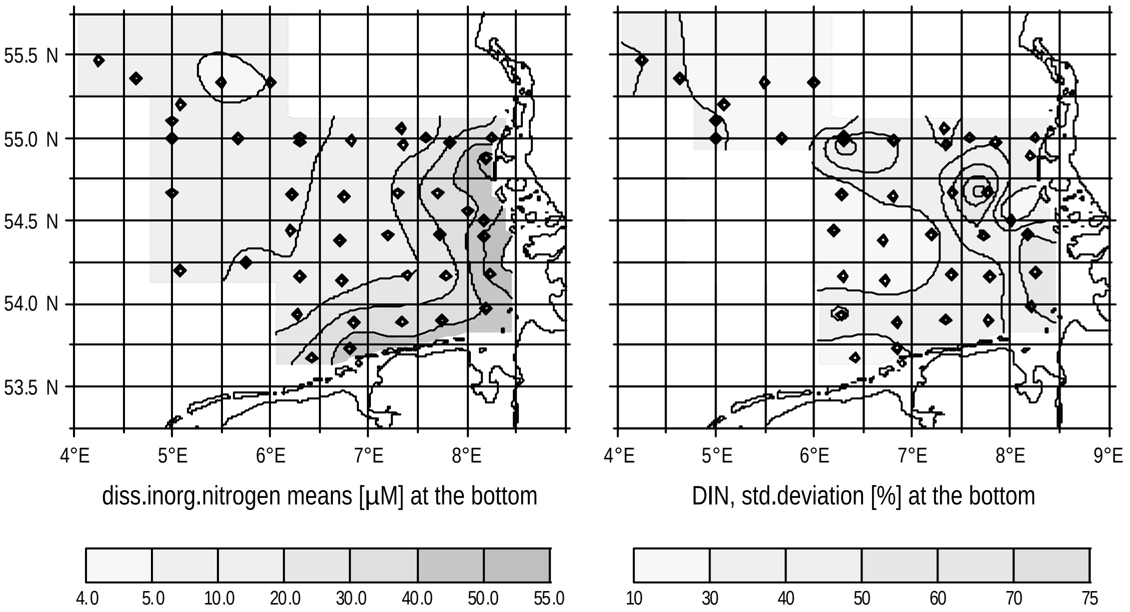
The German Bight in January and February 1985 - 1998:

Fig. 2.3.1.7: Mean winter nitrite gradients [µM] and their variability [%]



diss.inorg.nitrogen means [µM] at the surface

DIN, std.deviation [%] at the surface

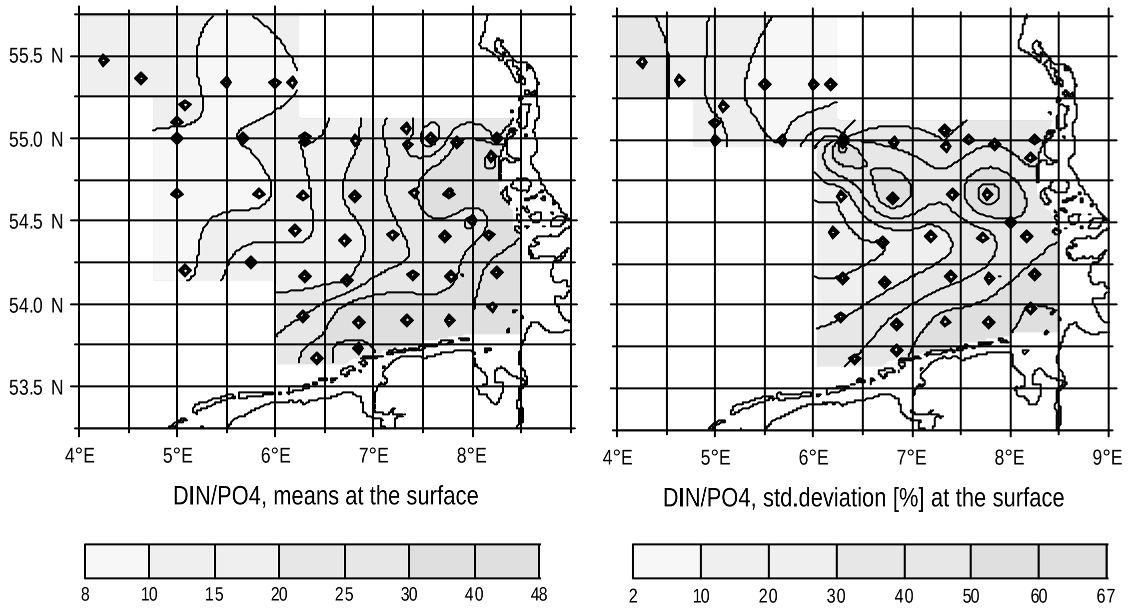


diss.inorg.nitrogen means [µM] at the bottom

DIN, std.deviation [%] at the bottom

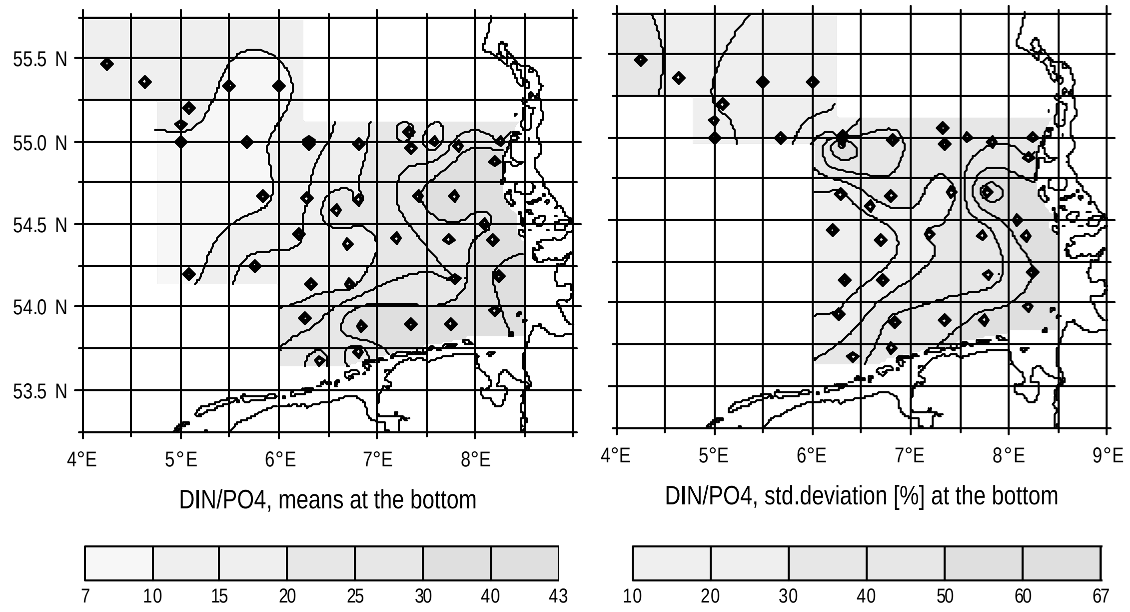
The German Bight in January and February 1985 - 1998

Fig. 2.3.1.8: Mean winter DIN gradients [µM] and their variability [%]



DIN/PO4, means at the surface

DIN/PO4, std.deviation [%] at the surface

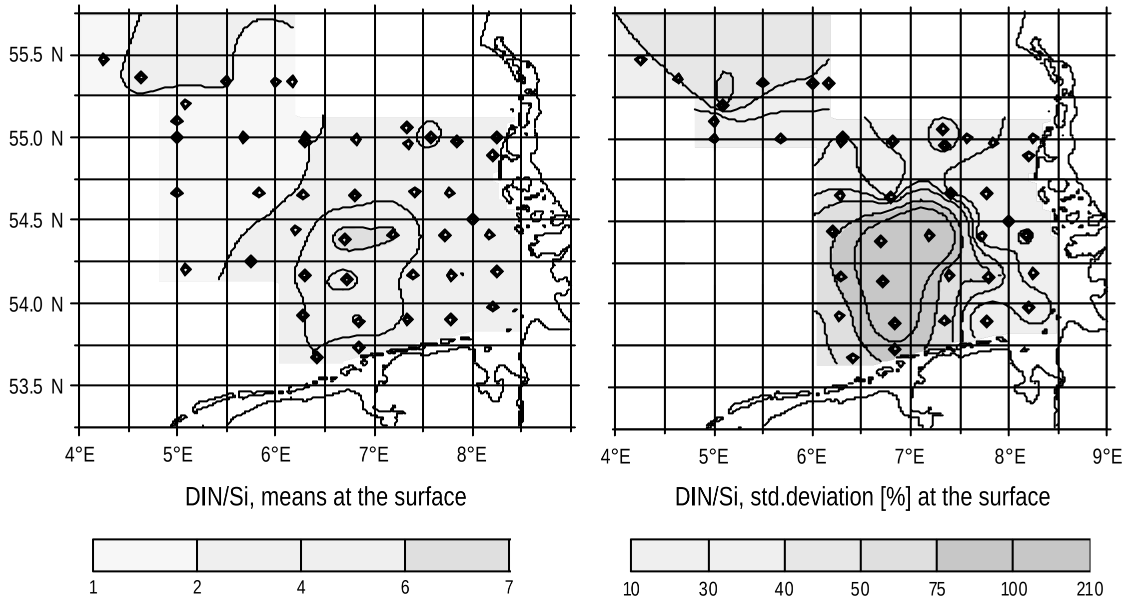


DIN/PO4, means at the bottom

DIN/PO4, std.deviation [%] at the bottom

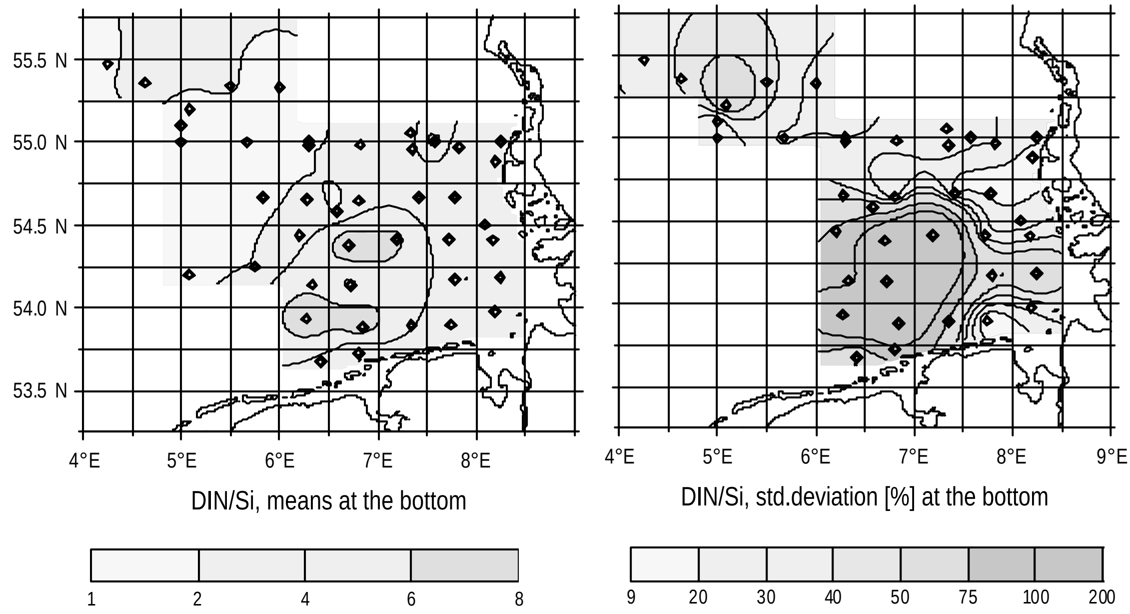
The German Bight in January and February 1985 - 1998:

Fig. 2.3.1.9: Mean winter N/P ratios [M/M] and their variability [%]



DIN/Si, means at the surface

DIN/Si, std.deviation [%] at the surface

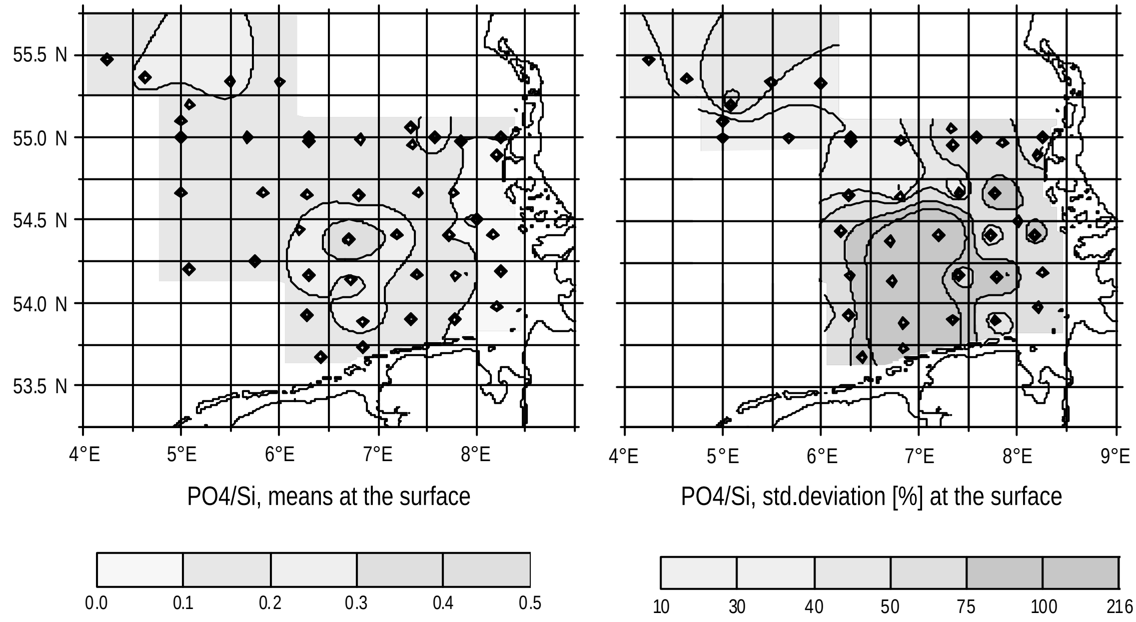


DIN/Si, means at the bottom

DIN/Si, std.deviation [%] at the bottom

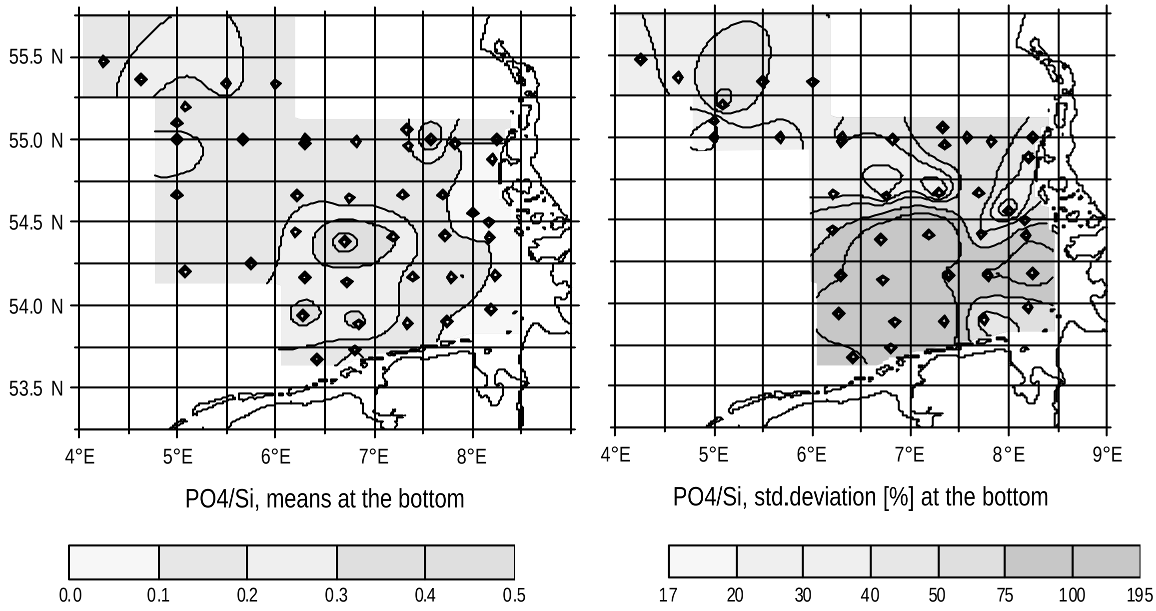
The German Bight in January and February 1985 - 1998:

Fig. 2.3.1.10: Mean winter N/Si ratios [M/M] and their variability [%]



PO4/Si, means at the surface

PO4/Si, std.deviation [%] at the surface

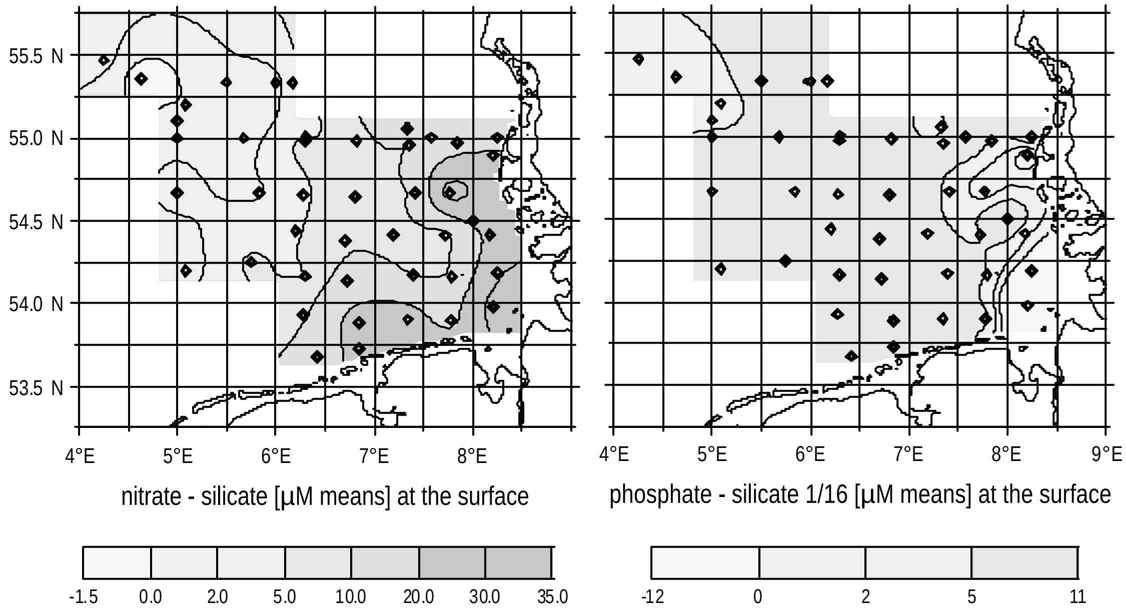


PO4/Si, means at the bottom

PO4/Si, std.deviation [%] at the bottom

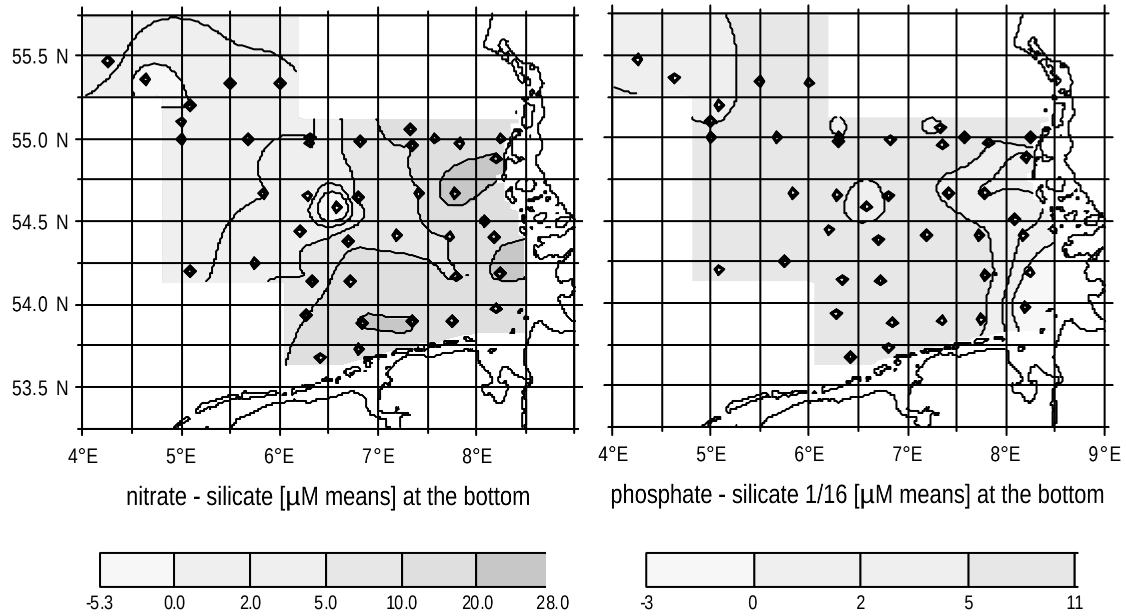
The German Bight in January and February 1985 - 1998:

Fig. 2.3.1.11: Mean winter P/Si ratios [M/M] and their variability [%]



nitrate - silicate [µM means] at the surface

phosphate - silicate 1/16 [µM means] at the surface



nitrate - silicate [µM means] at the bottom

phosphate - silicate 1/16 [µM means] at the bottom

The German Bight in January and February 1985 - 1998

Fig. 2.3.1.12: Differences between nitrate and silicate (left) and phosphate and 1/16 silicate (µM) during winter, indicating possible anthropogenic contributions to nitrate and phosphate in relation to silicate, assuming Redfield ratios of 1:1 and 1:16 (Redfield et al. 1963)

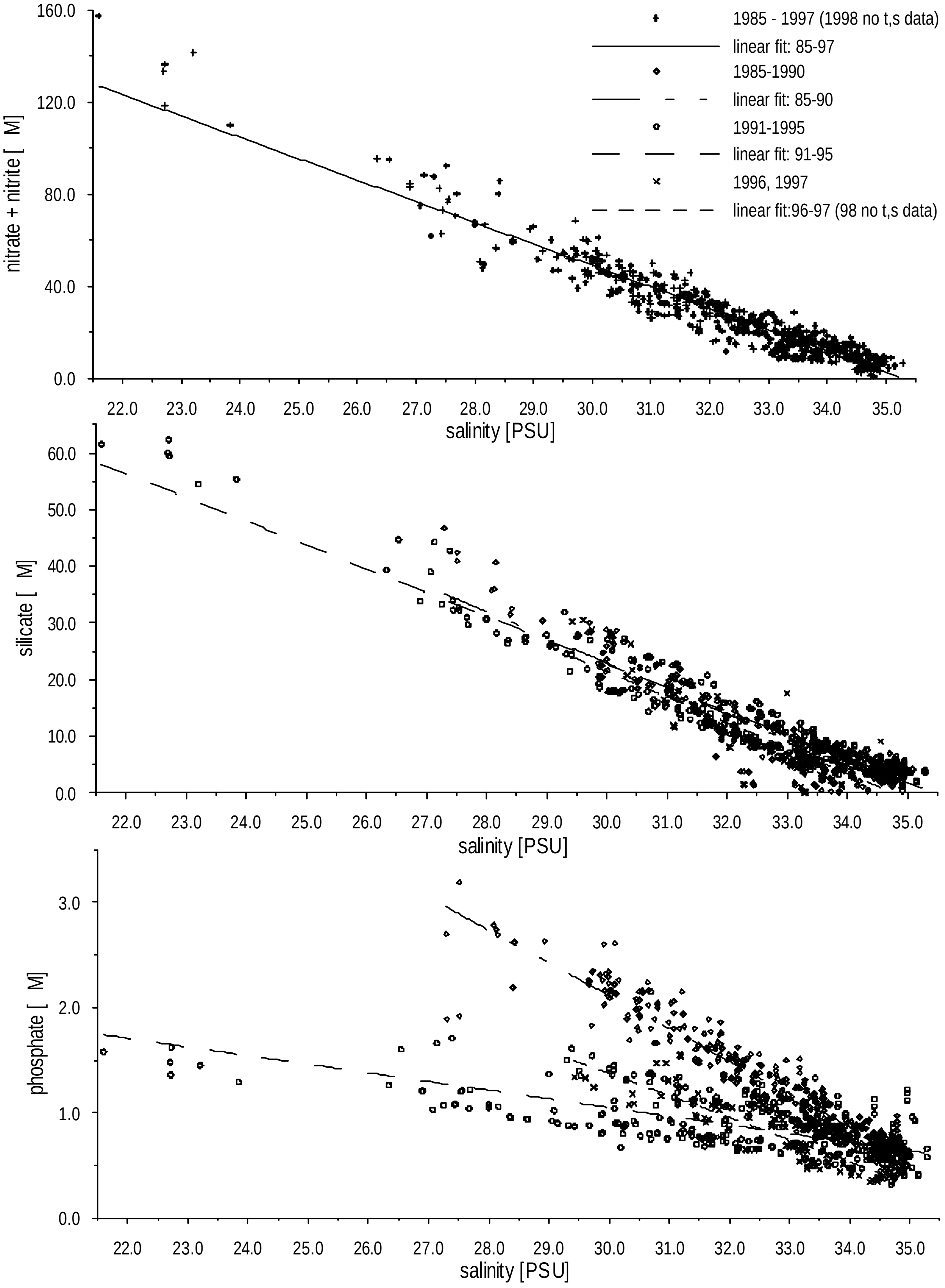


Fig. 2.3.2.1: Mixing diagrams of nitrate + nitrite, silicate and phosphate during winter in the German Bight 1985 - 1997

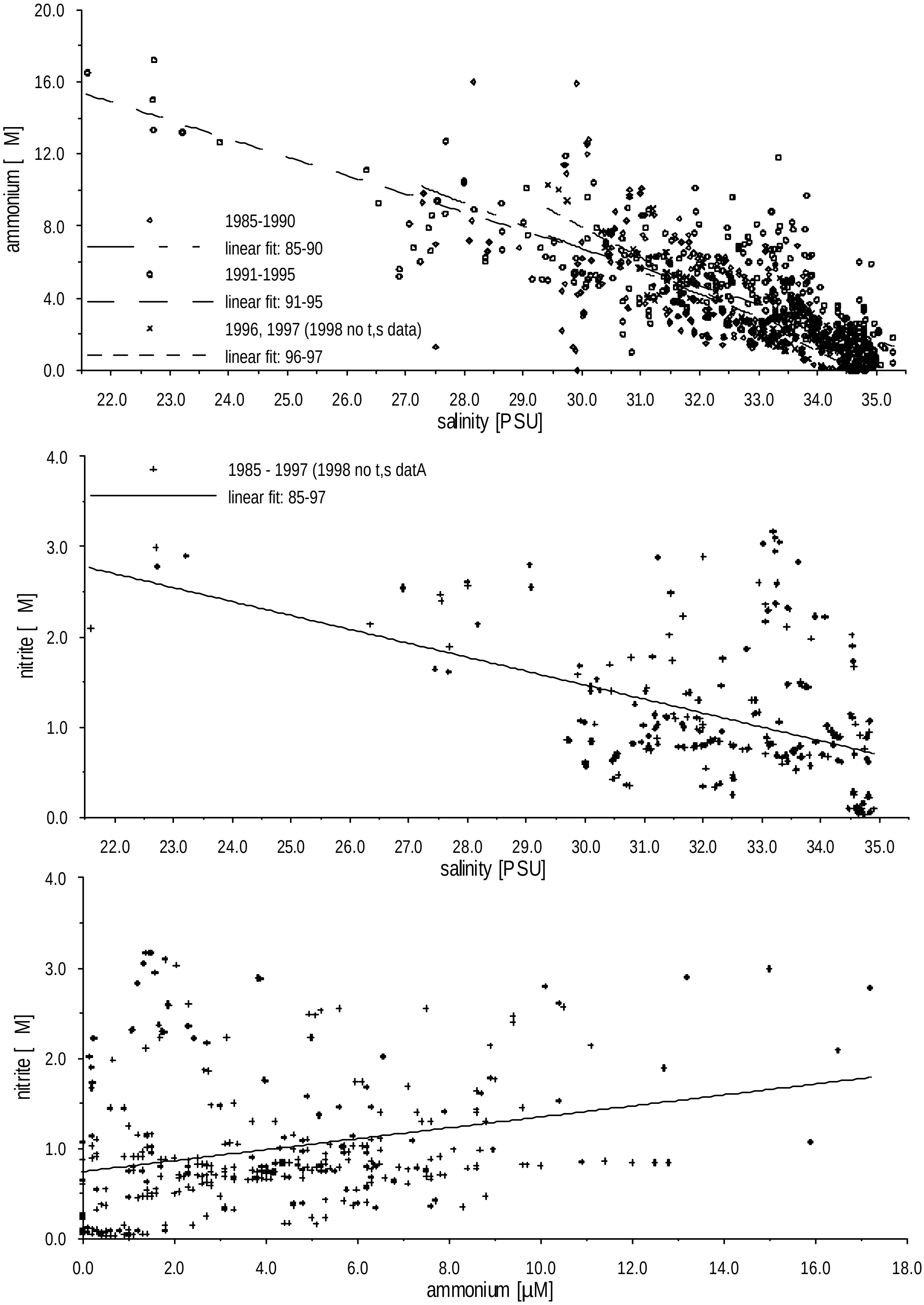


Fig. 2.3.2.2: Correlation diagrams of silicate with nitrate+nitrite, phosphate with nitrate+nitrite and phosphate with silicate during winter in the German Bight 1985 - 1998

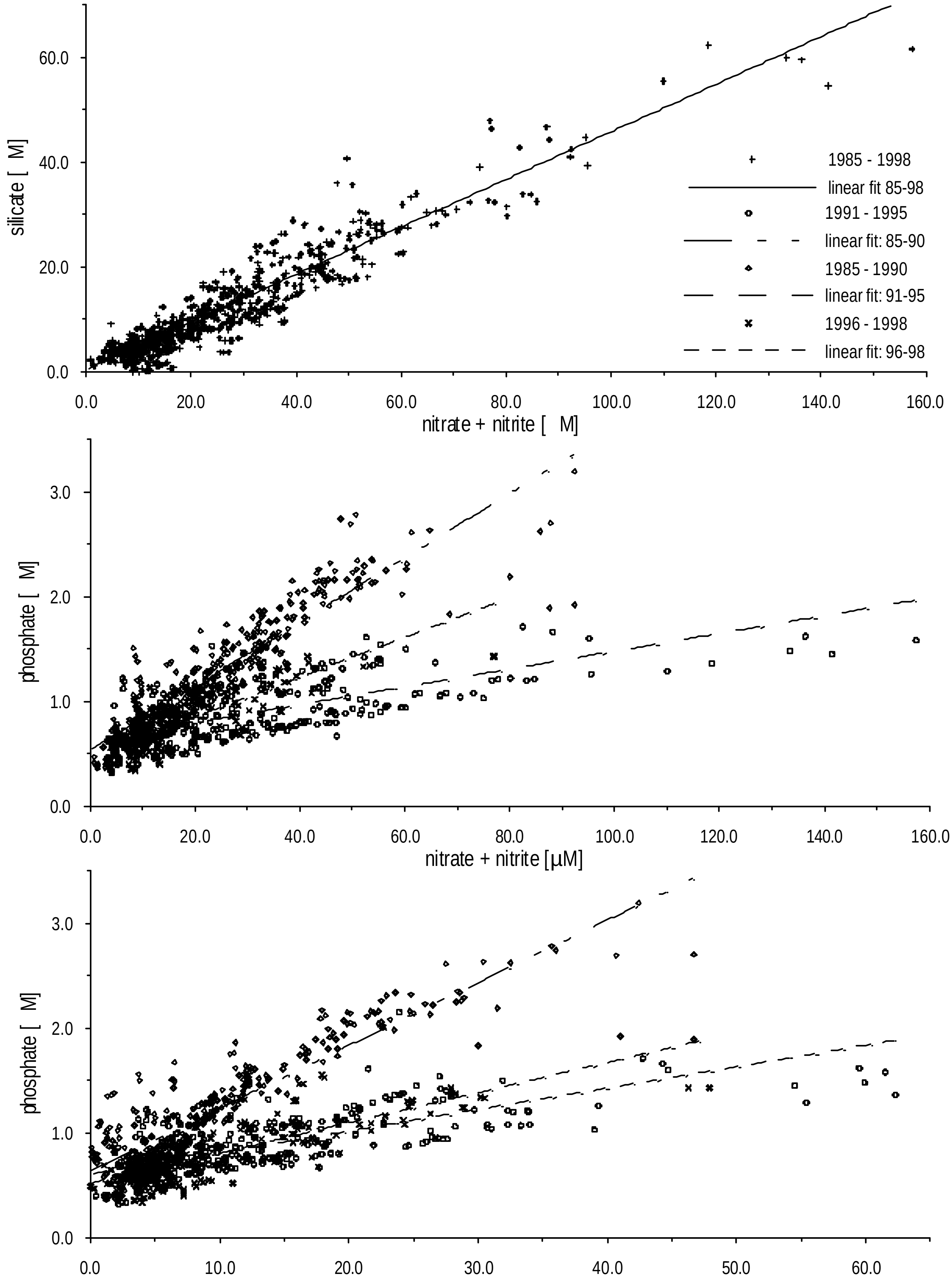


Fig. 2.3.2.3: Mixing diagrams of ammonium and nitrite and correlation diagram of nitrite with ammonium during winter in the German Bight 1985 –1998

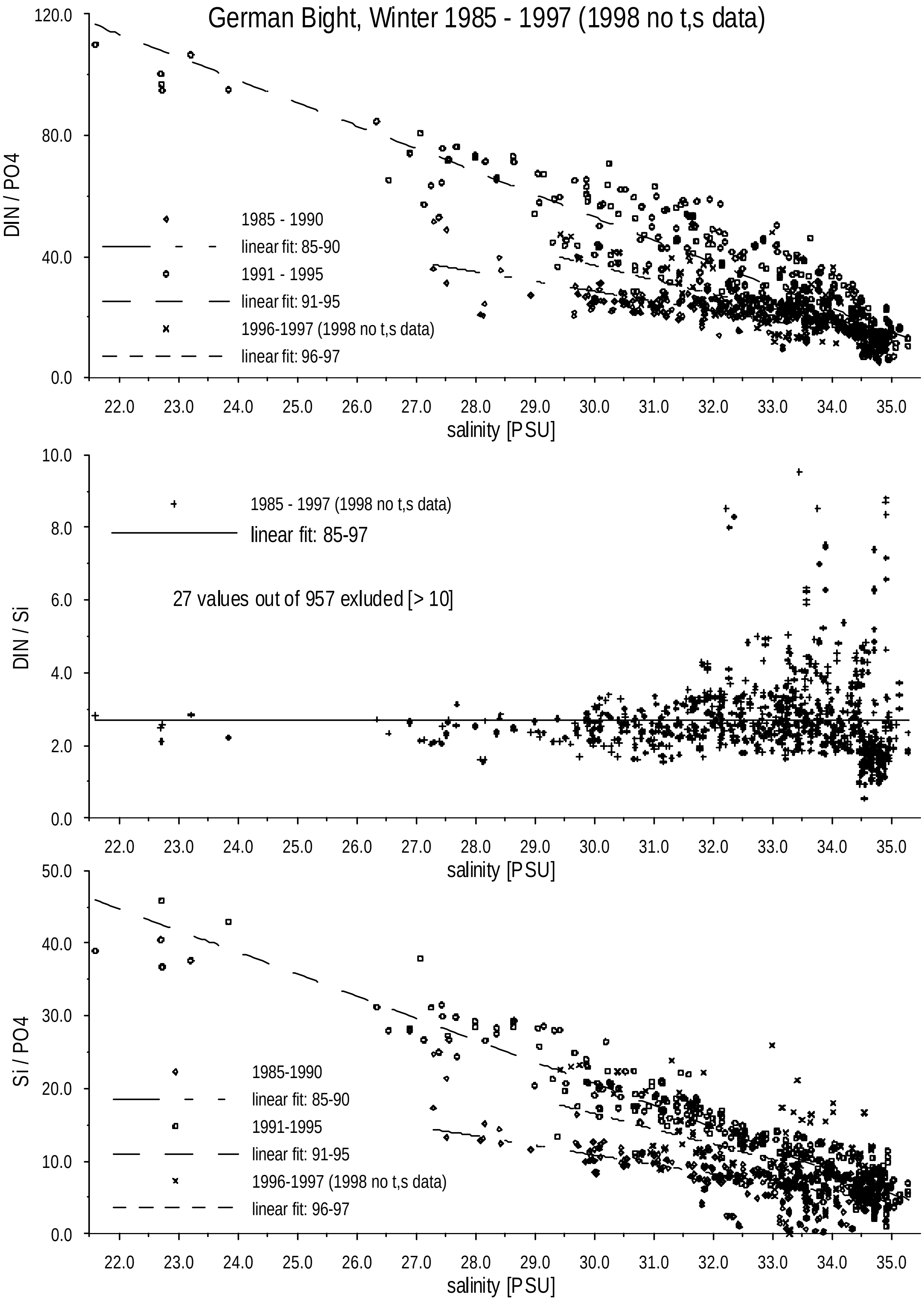


Fig. 2.3.2.4: Mixing diagrams DIN/P, DIN/Si and Si/P ratios [M/M] during winter in the German Bight 1985 –1997

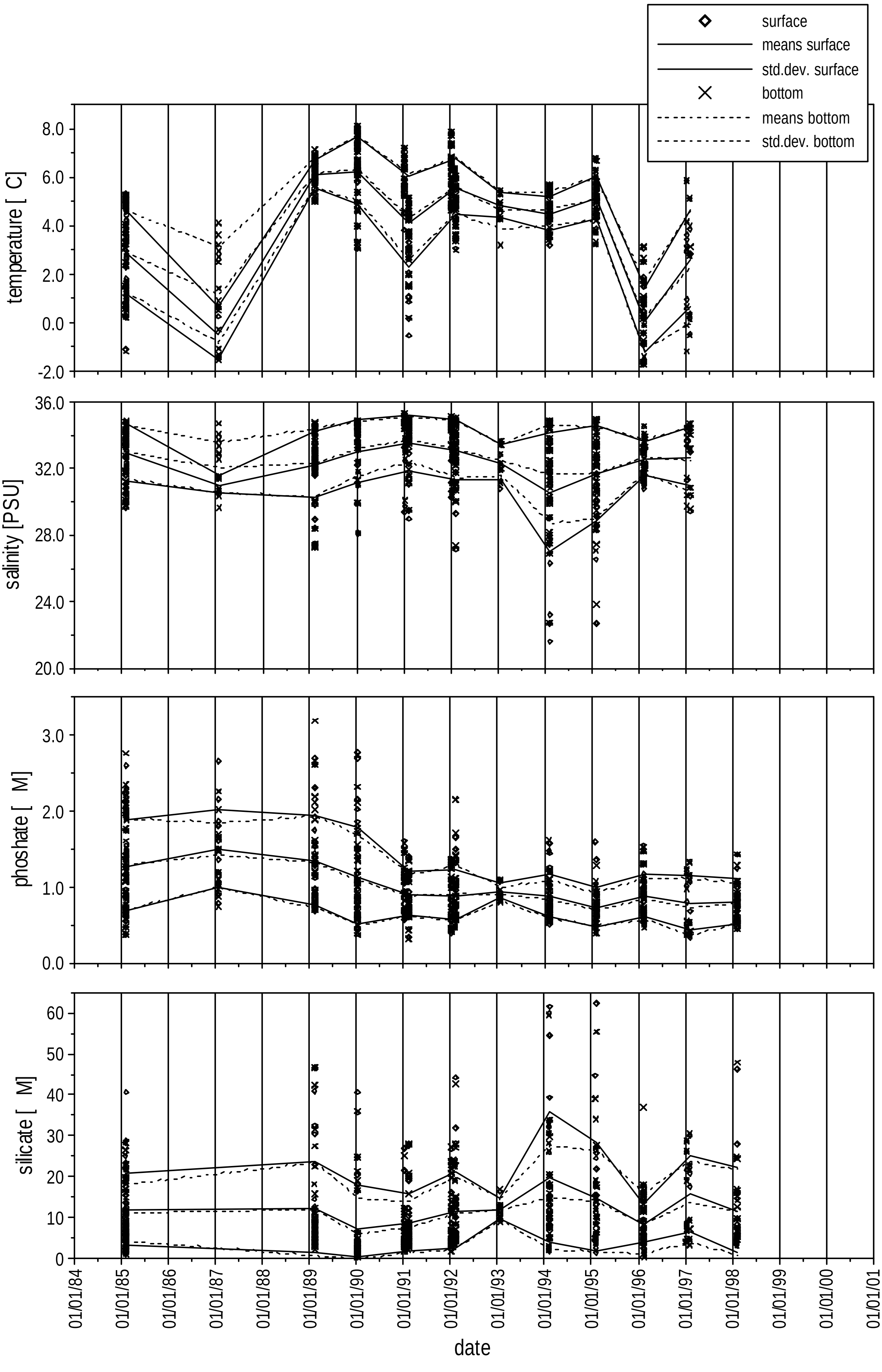


Fig. 2.3.3.1: Time series of winter concentrations in the total German Bight 1985 – 1998

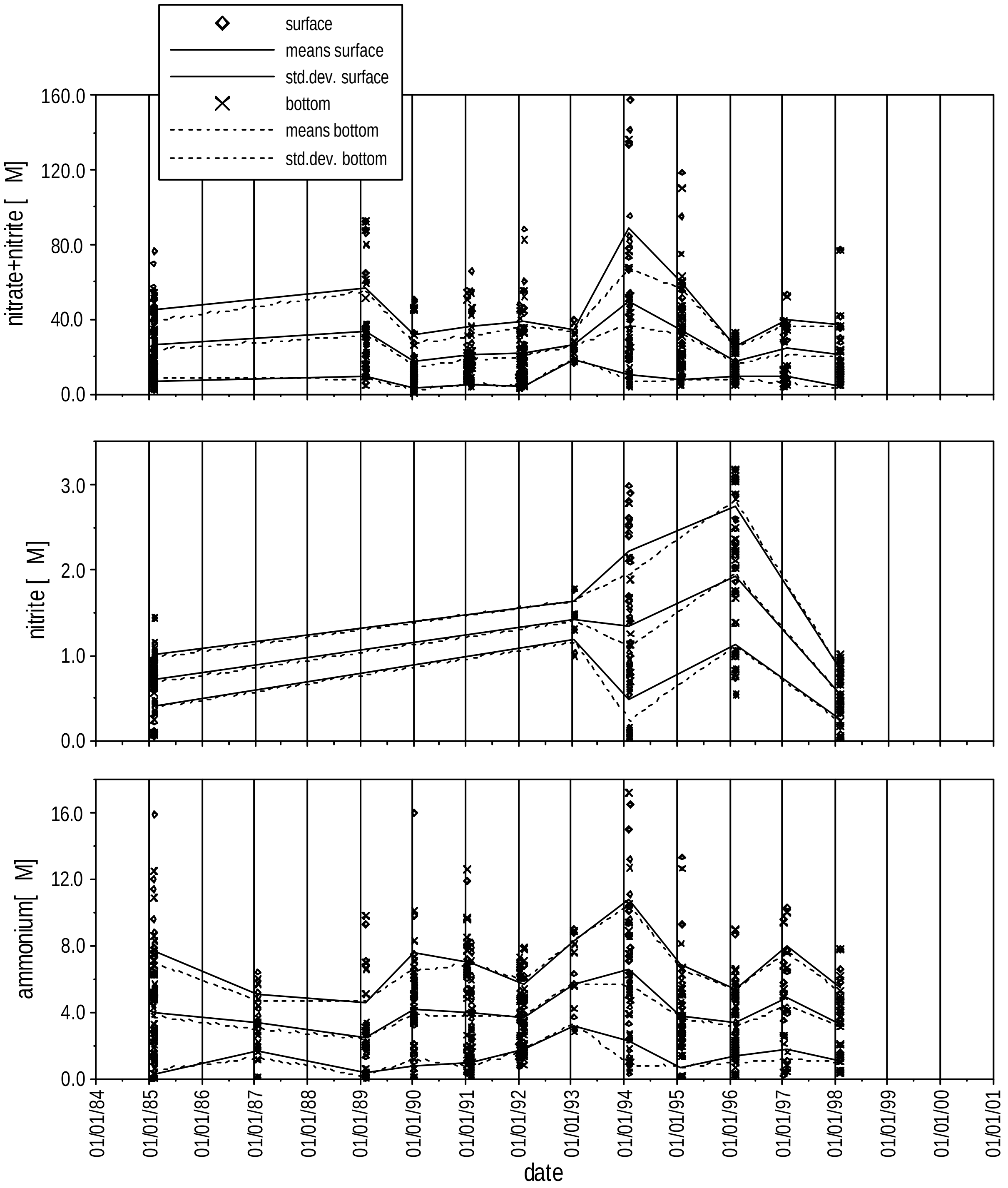


Fig. 2.3.3.2: Time series of winter concentrations in the total German Bight 1985 – 1998

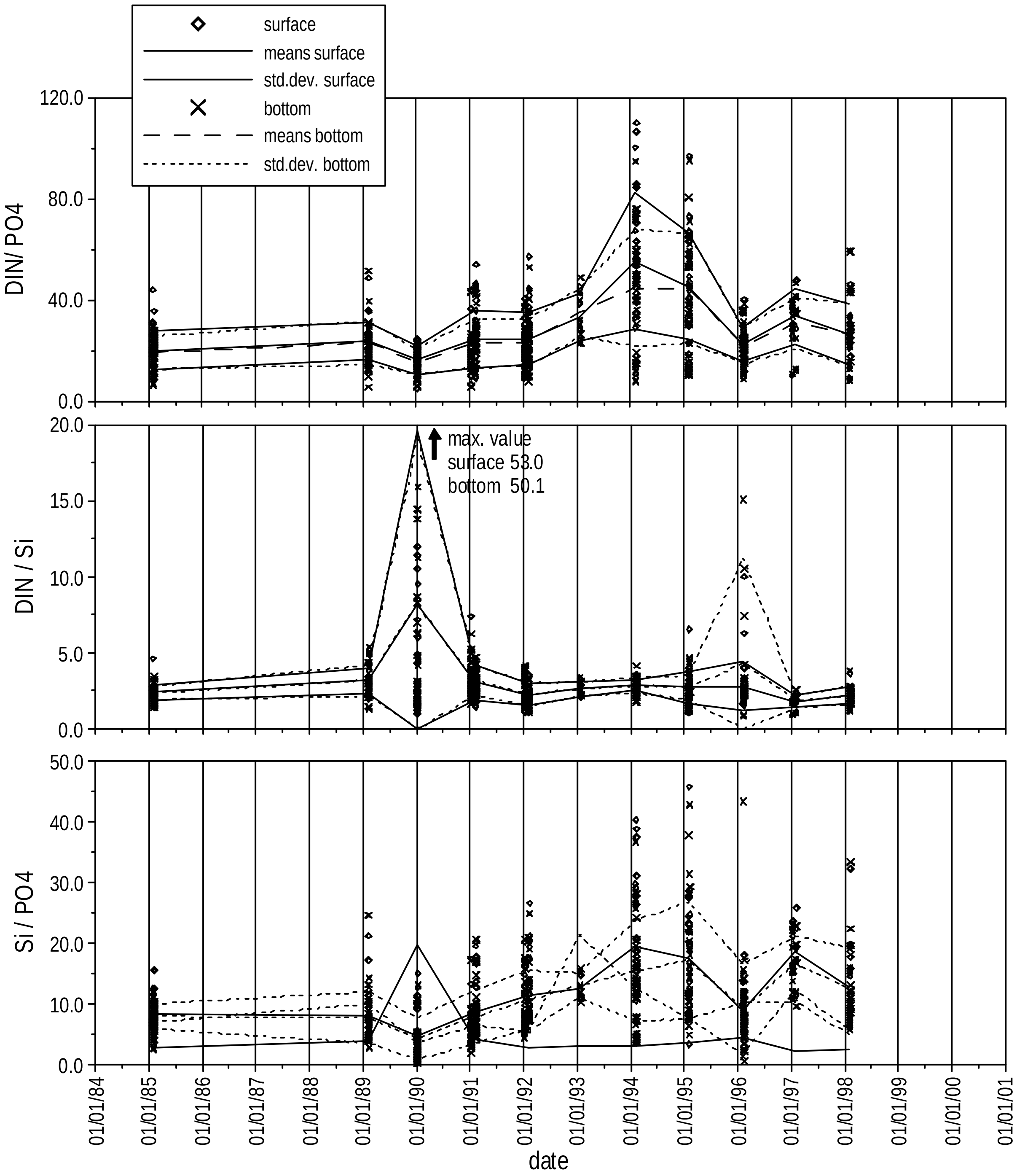


Fig. 2.3.3.3: Time series of nutrient ratios [M/M] in the total German Bight 1985 – 1998 during winter

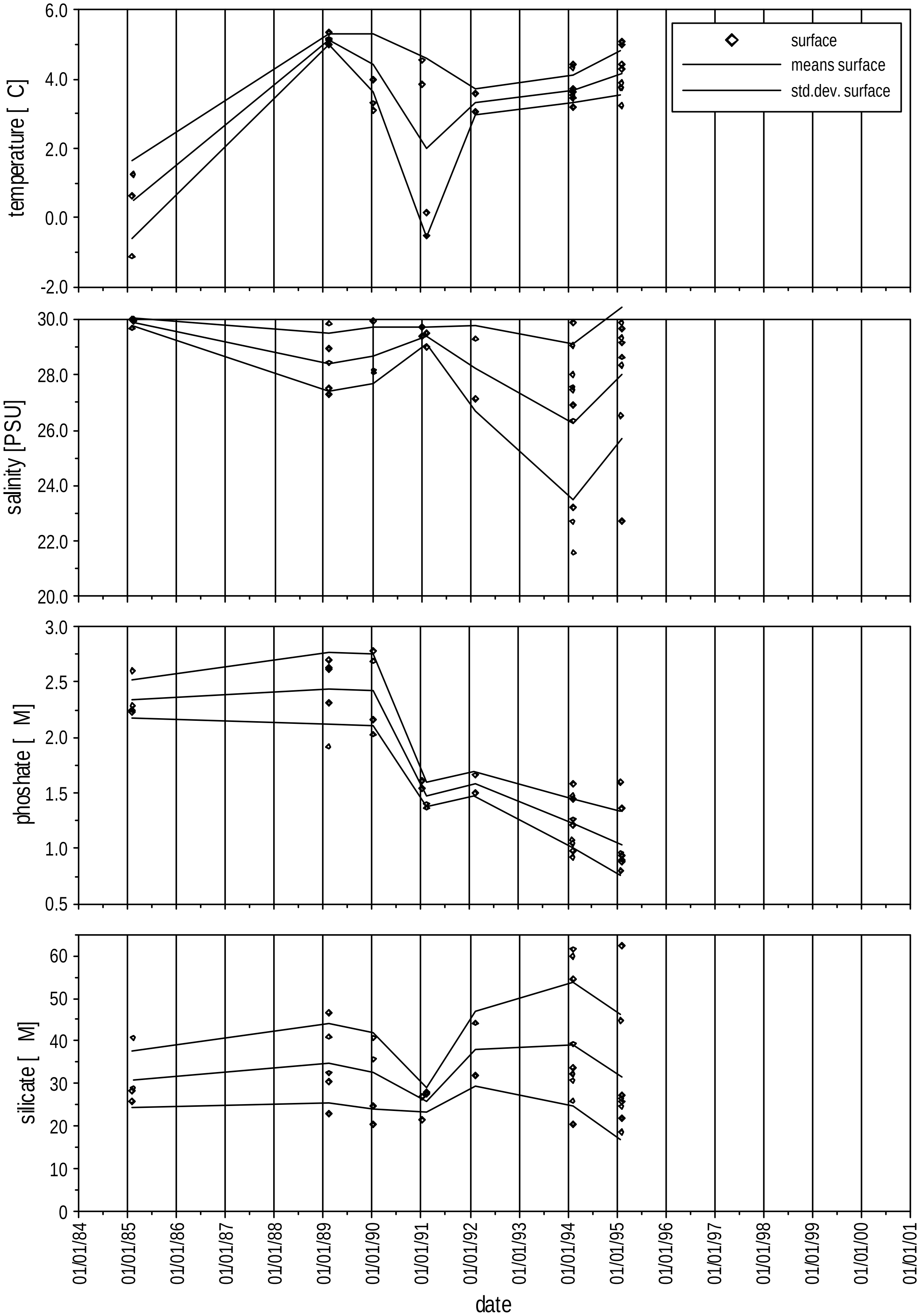


Fig. 2.3.3.4: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinites of 21 – 30

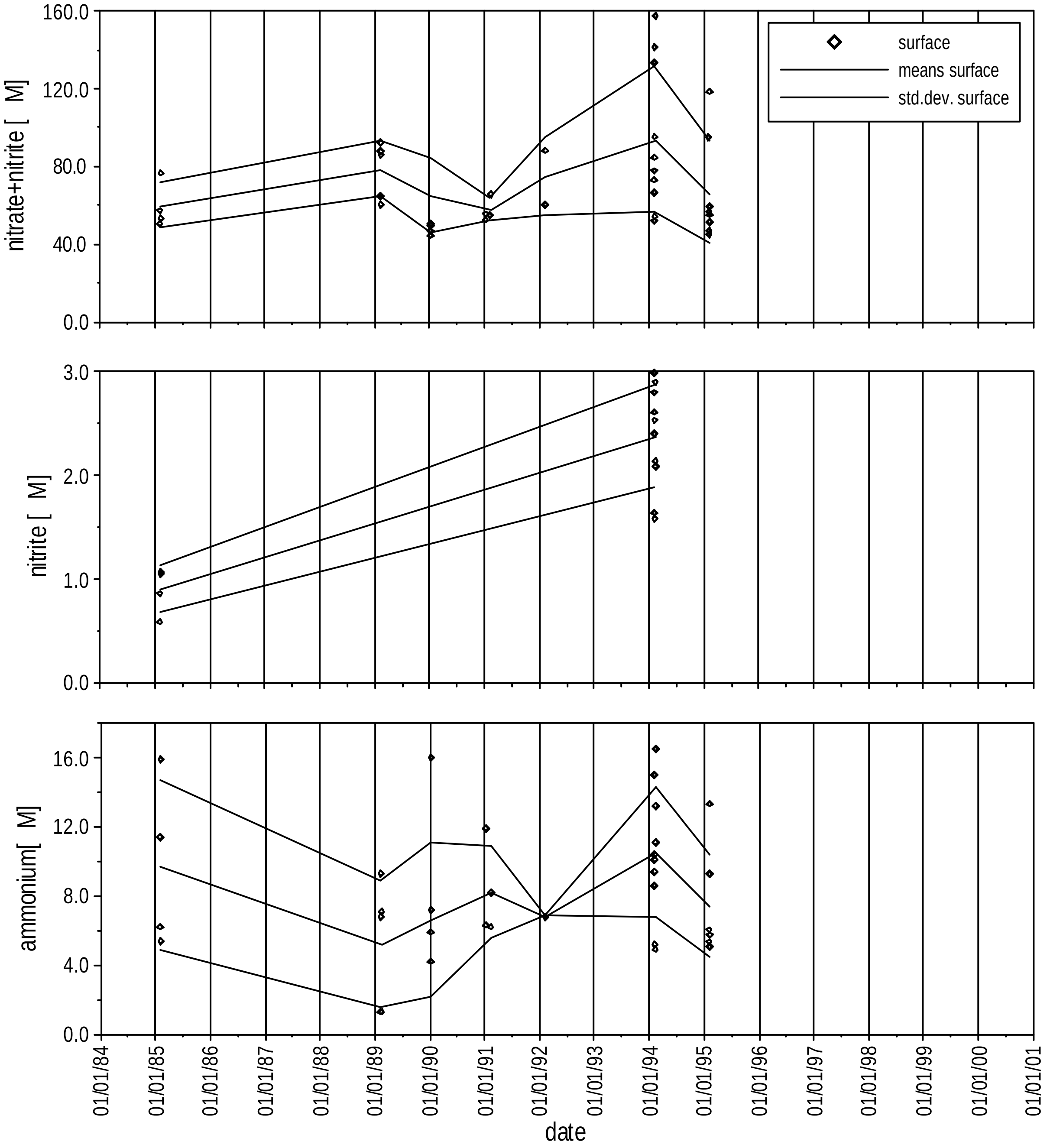


Fig. 2.3.3.5: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 21 – 30

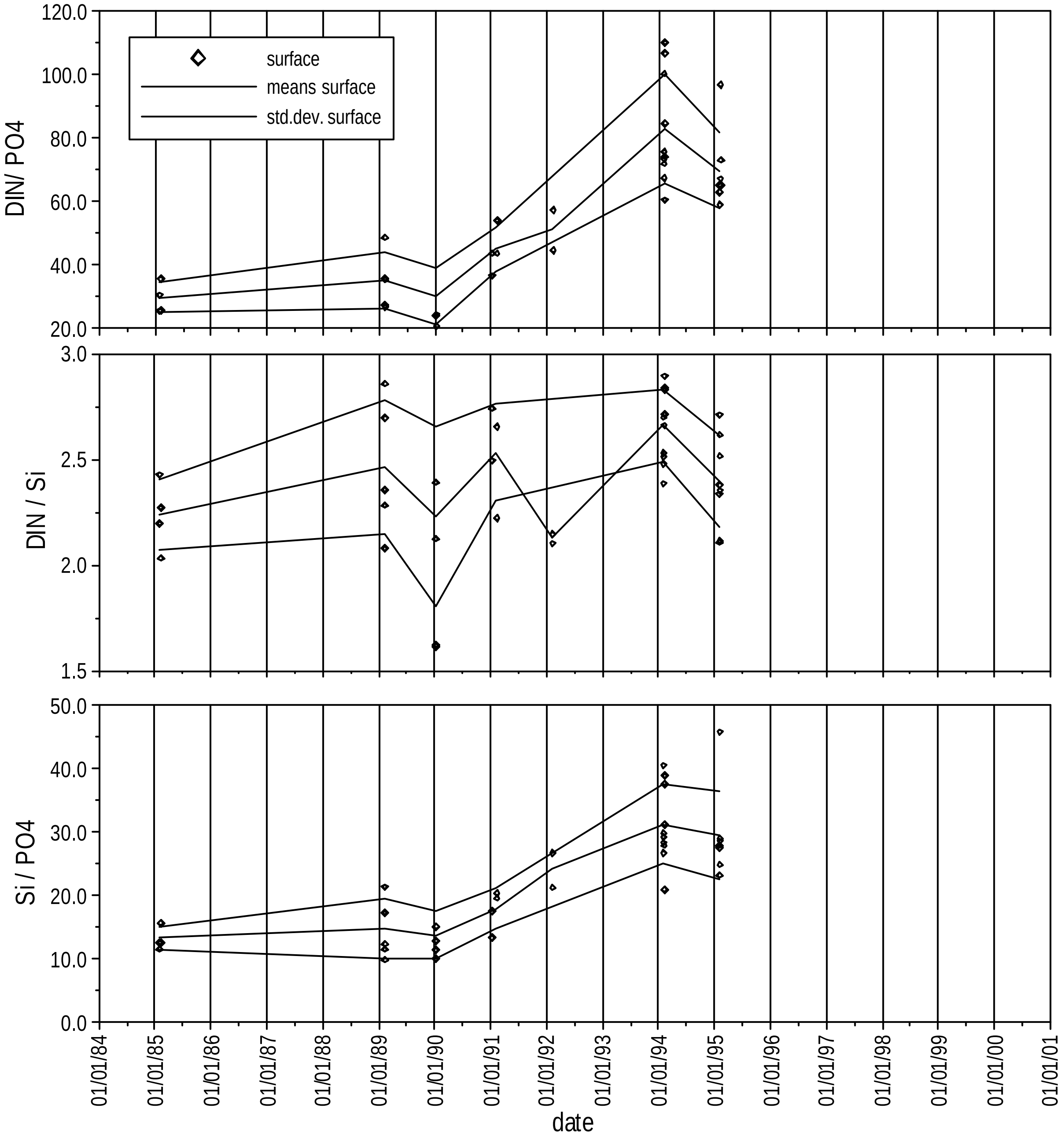


Fig. 2.3.3.6: Time series of nutrient ratios [M/M] in the German Bight 1985 – 1998 at the surface and at salinities of 21 – 30 during winter

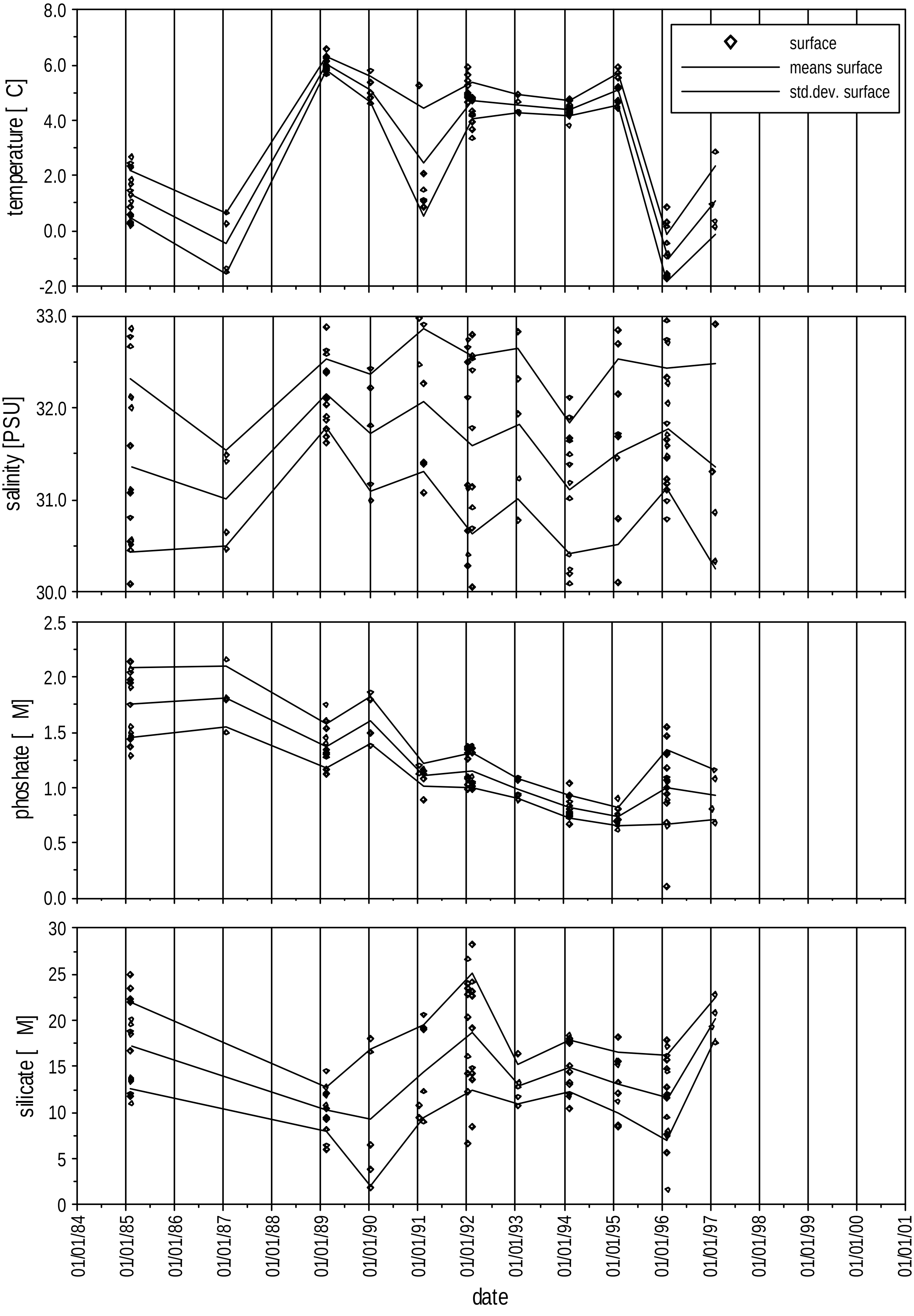


Fig. 2.3.3.7: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 30 - 33

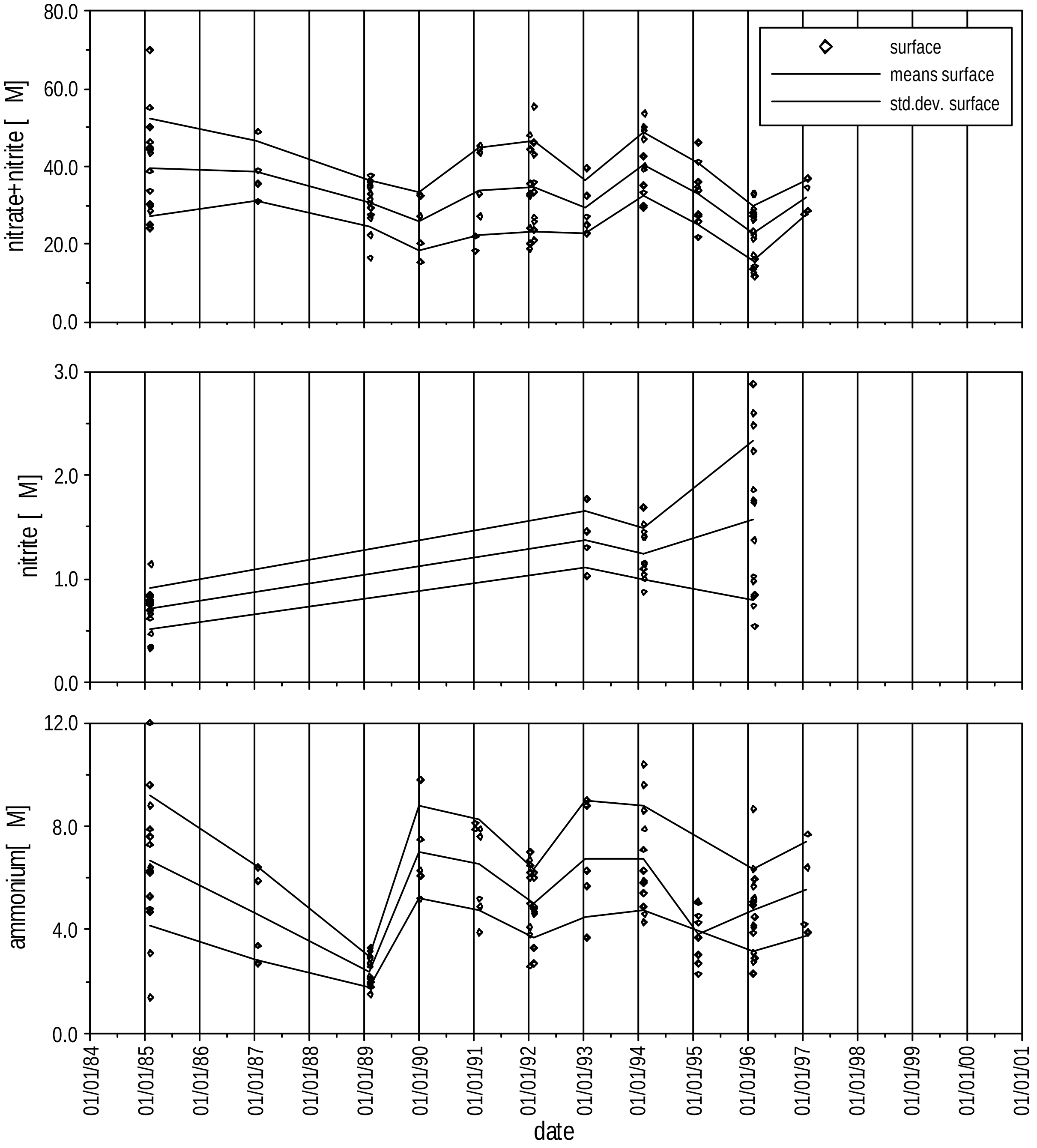


Fig. 2.3.3.8: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 30 - 33

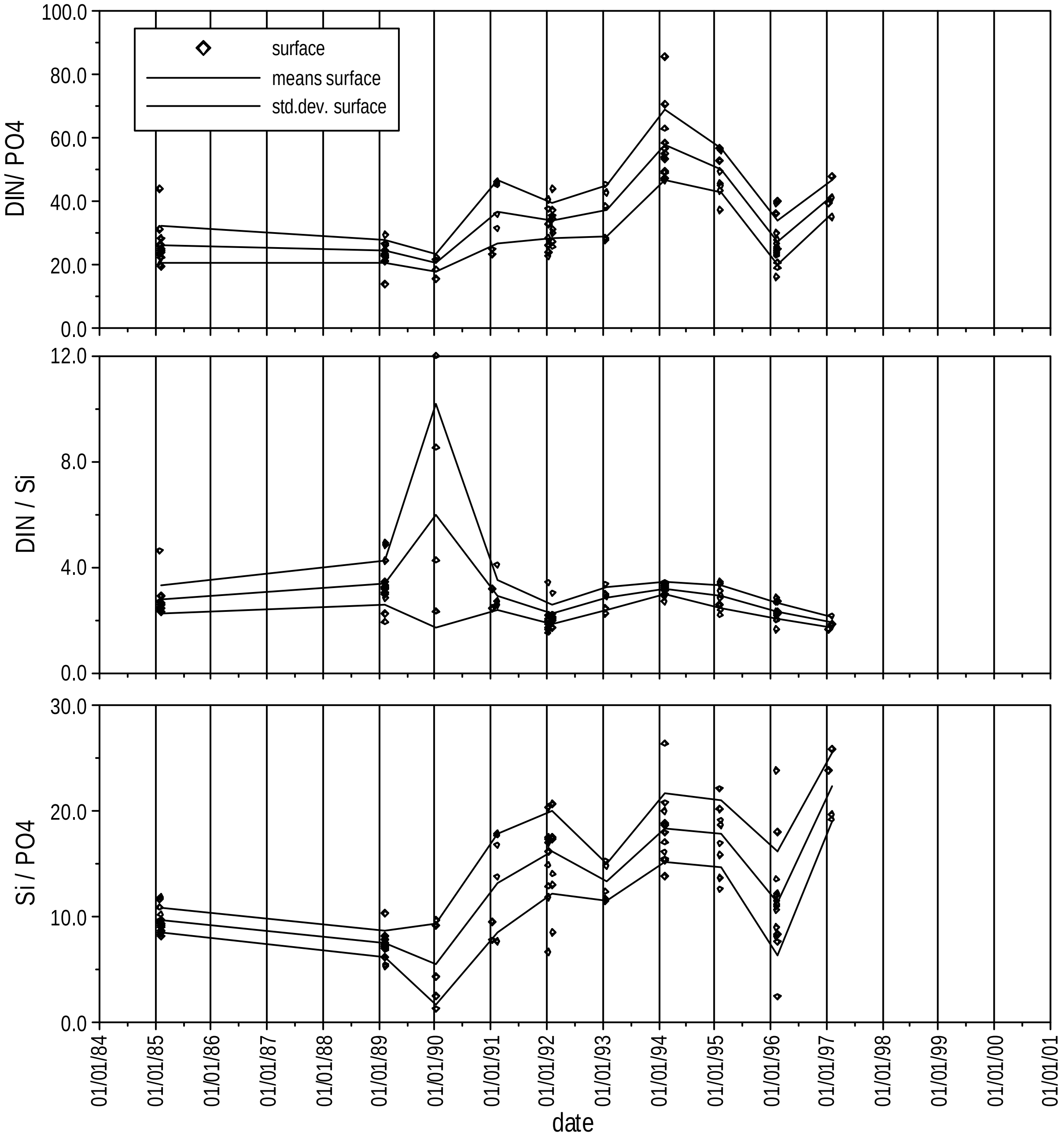


Fig. 2.3.3.9: Time series of nutrient ratios [M/M] in the German Bight 1985 – 1998 at the surface and at salinities of 30 - 33 during winter

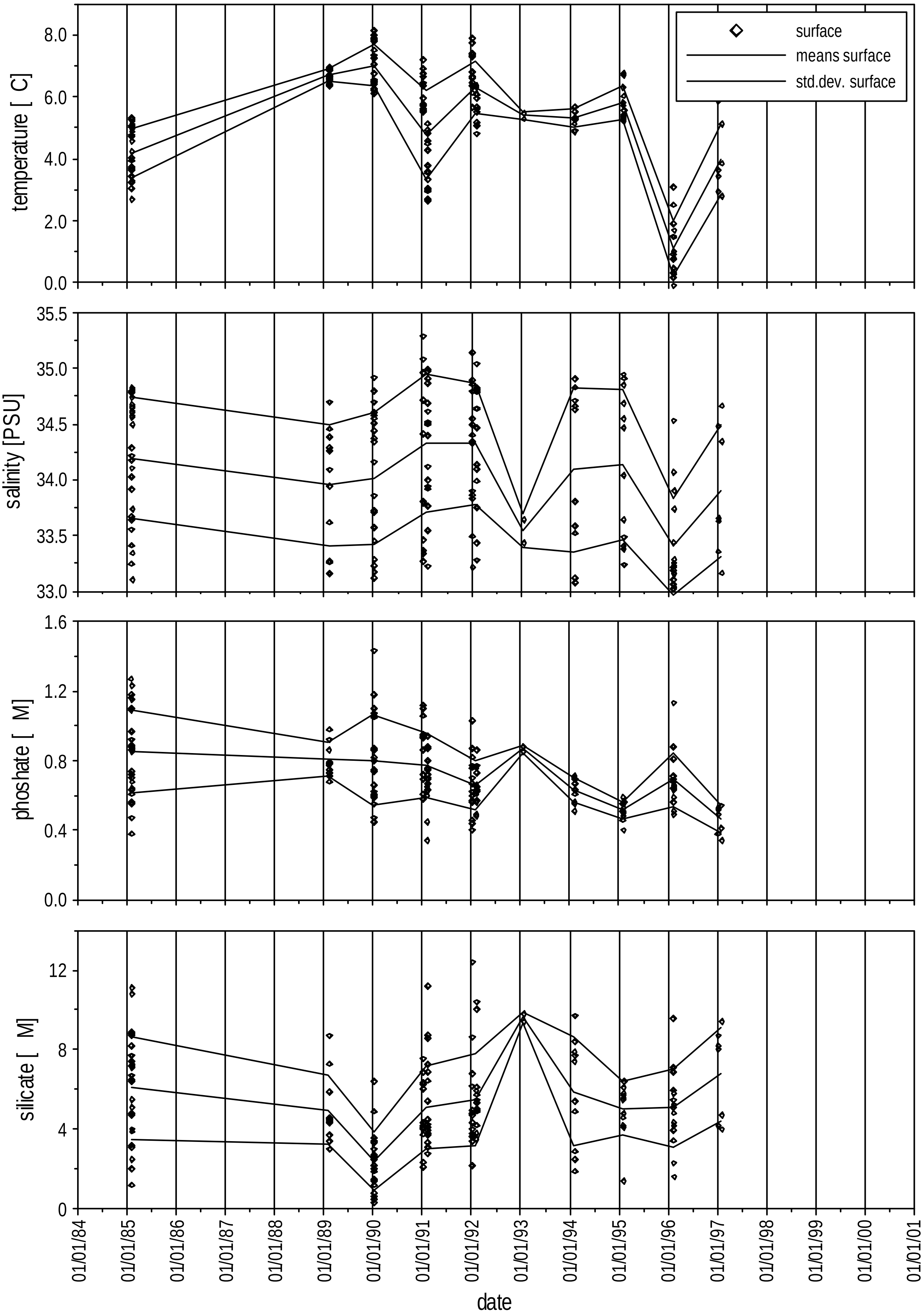


Fig. 2.3.3.10: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 33 – 35.5

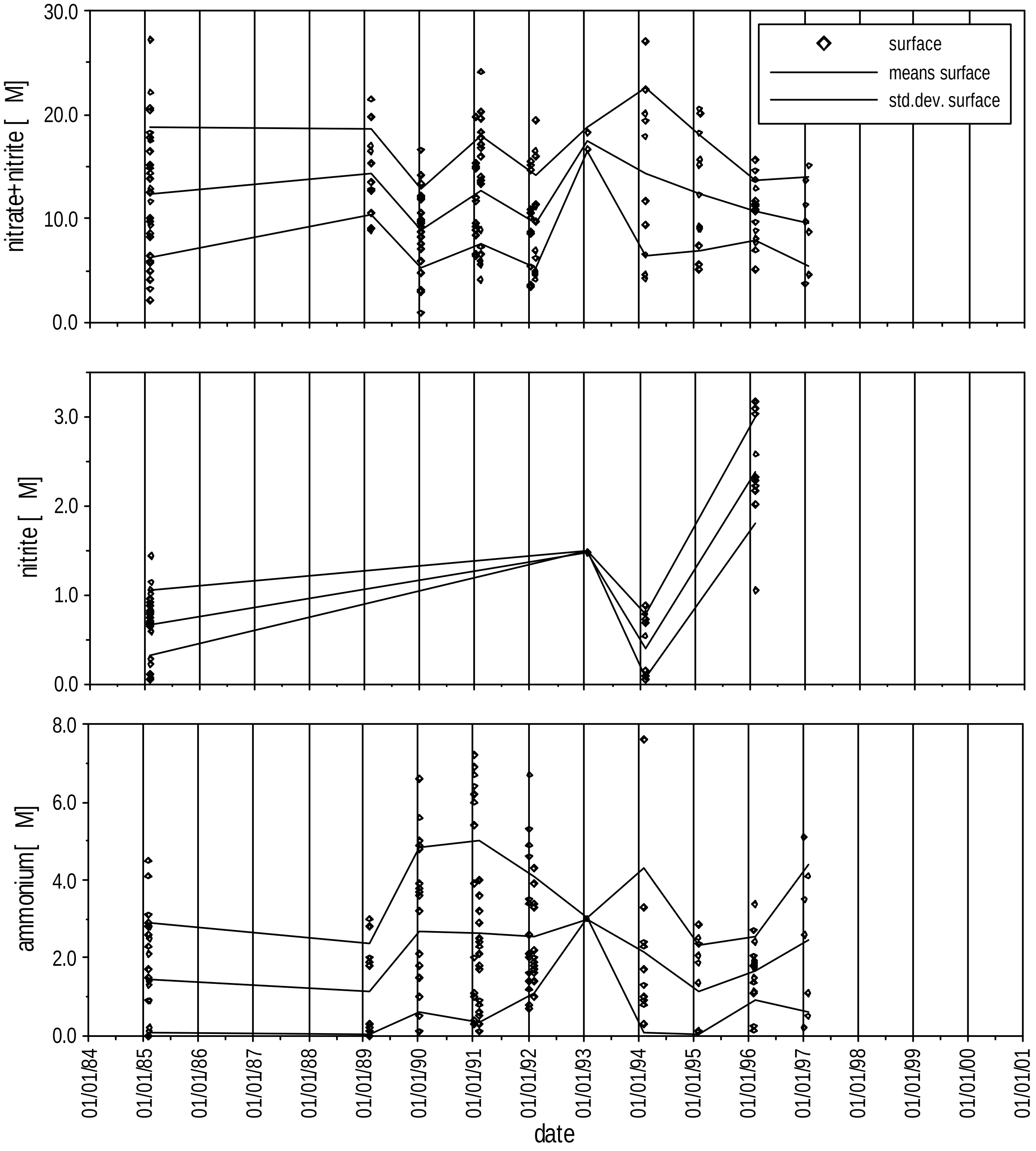


Fig. 2.3.3.11: Time series of winter concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 33 – 35.5

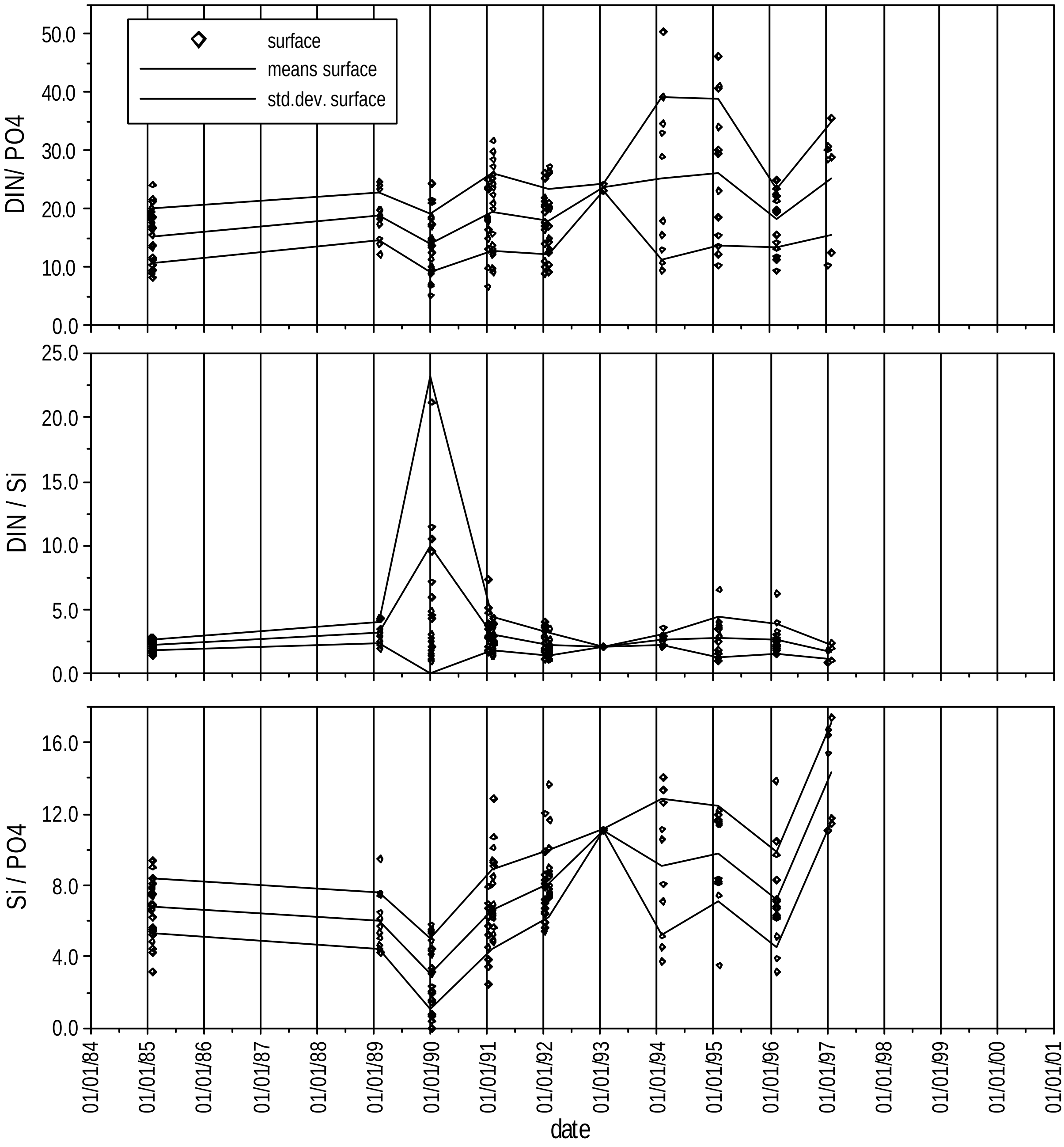
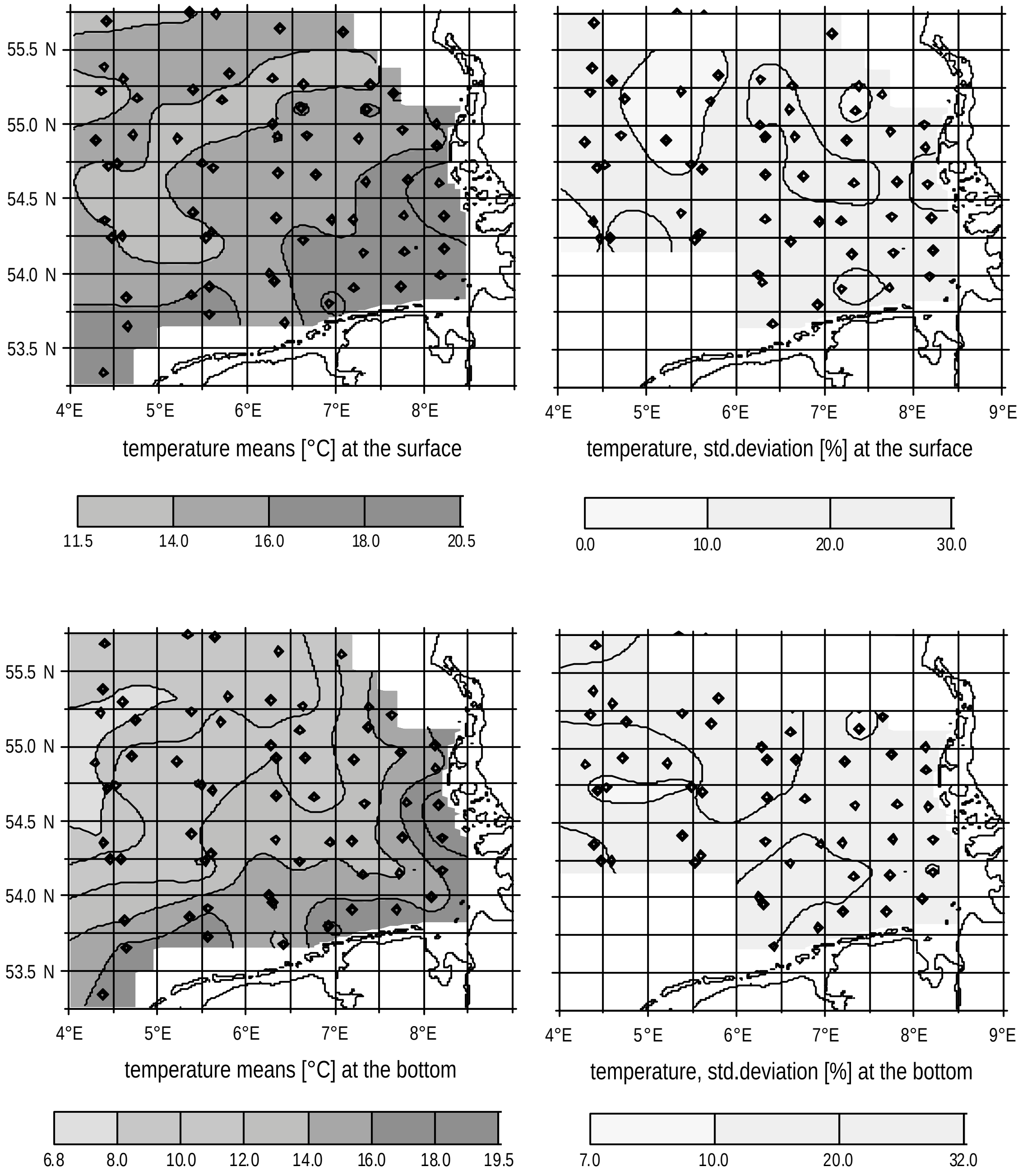
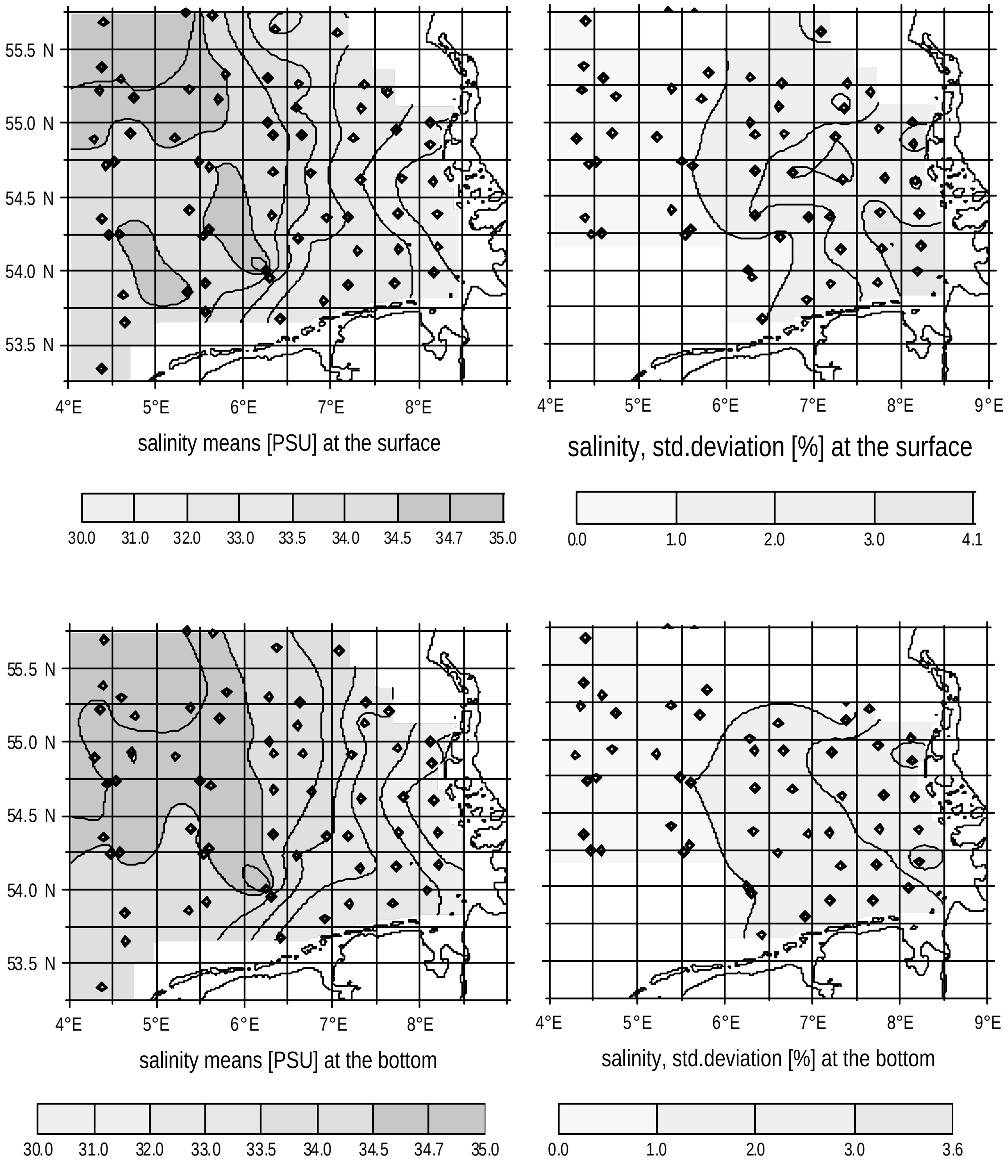


Fig. 2.3.3.12: : Time series of nutrient ratios [M/M] during winter at the surface of the German Bight 1985 – 1998 and at salinities of 33 – 35.5



The German Bight in June and July 1985 - 1998

Fig. 2.3.4.1: Mean summer temperature gradients [°C] and their variability [%]



salinity means [PSU] at the surface

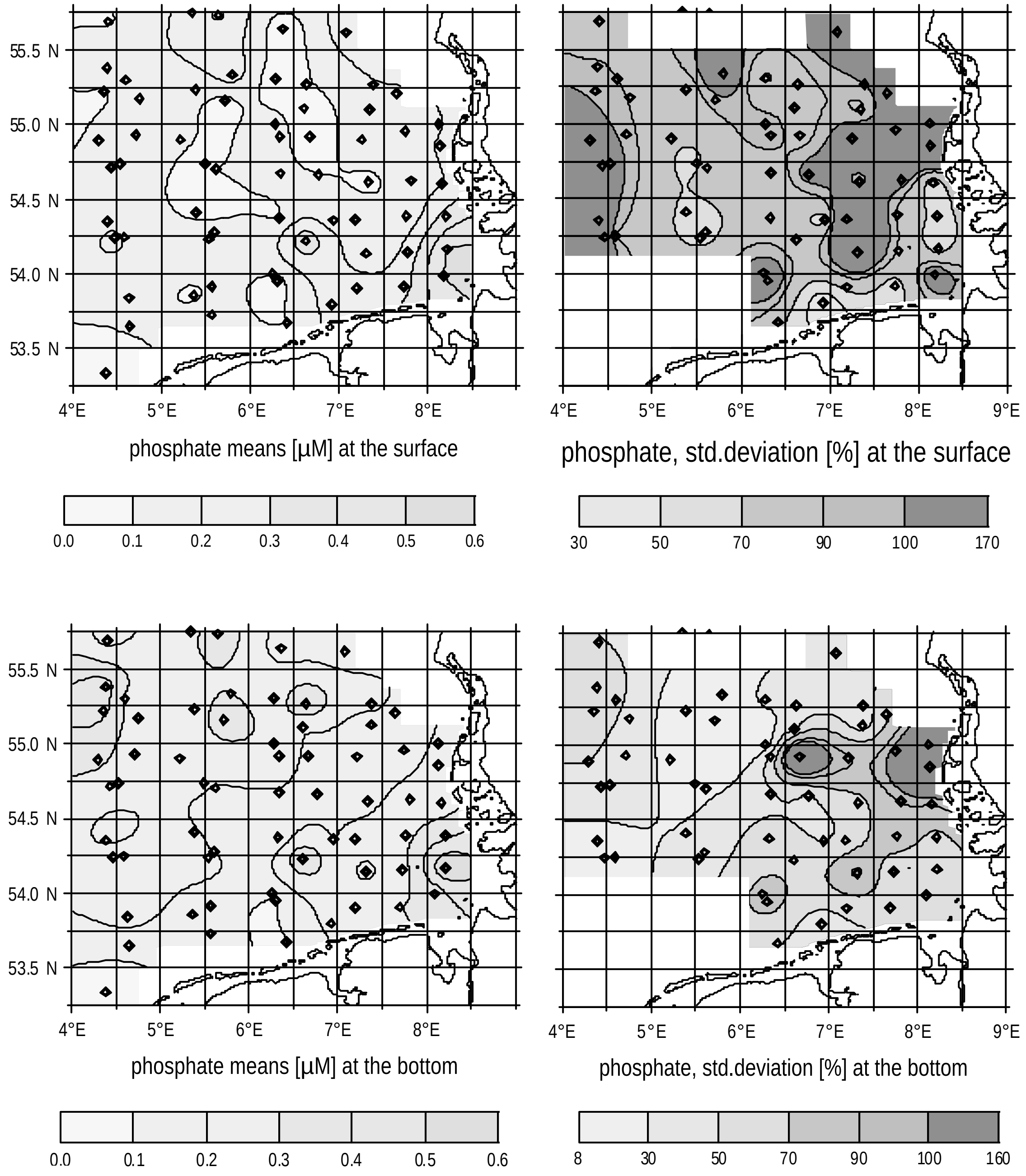
salinity, std.deviation [%] at the surface

salinity means [PSU] at the bottom

salinity, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.2: Mean summer salinity gradients and their variability [%]



phosphate means [µM] at the surface

phosphate, std.deviation [%] at the surface

phosphate means [µM] at the bottom

phosphate, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.3: Mean summer phosphate gradients [µM] and their variability [%]

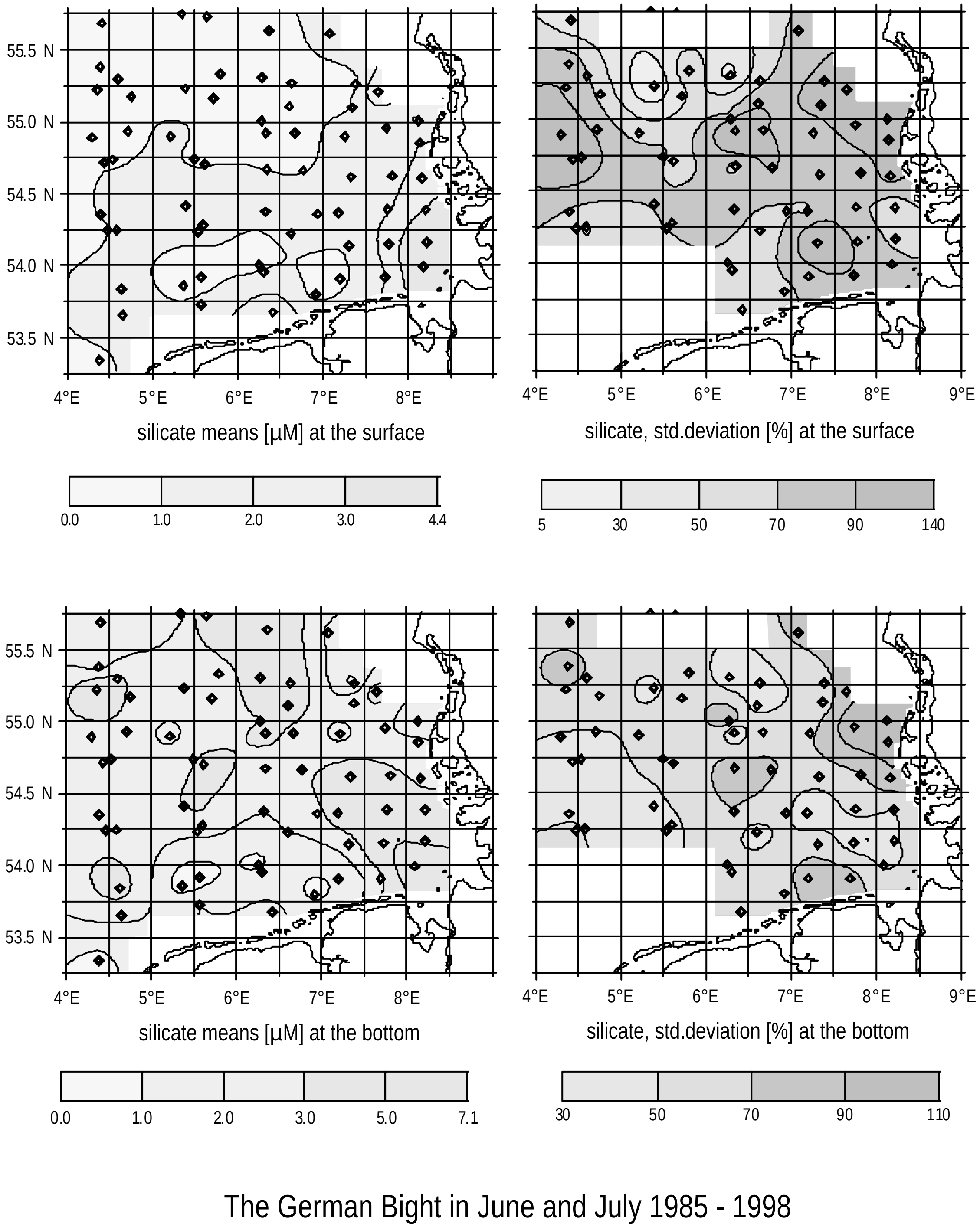
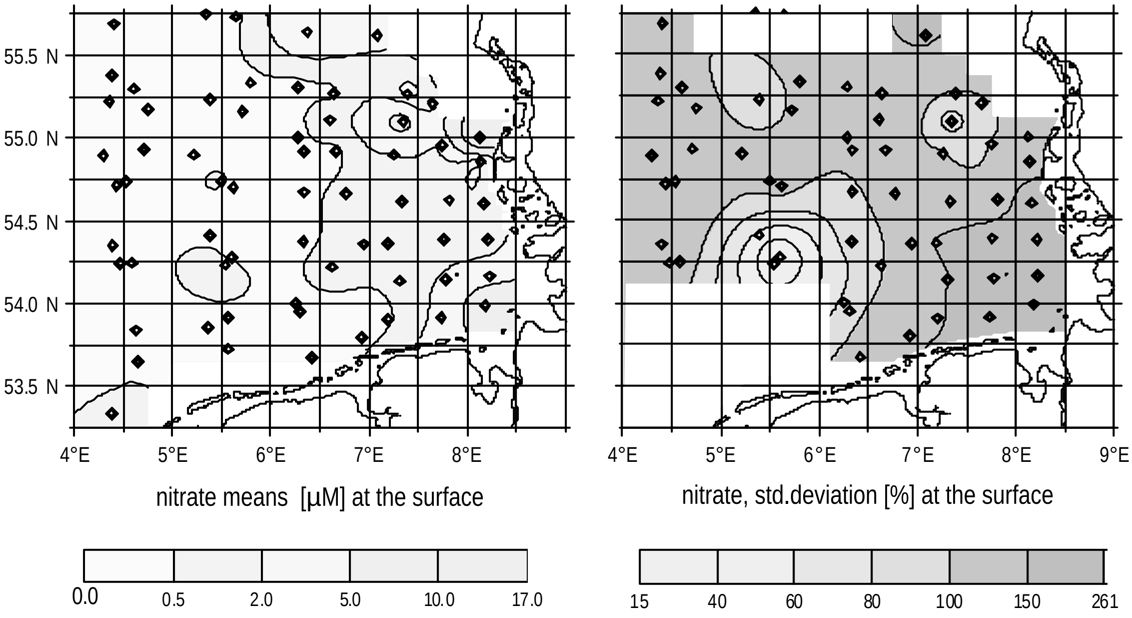
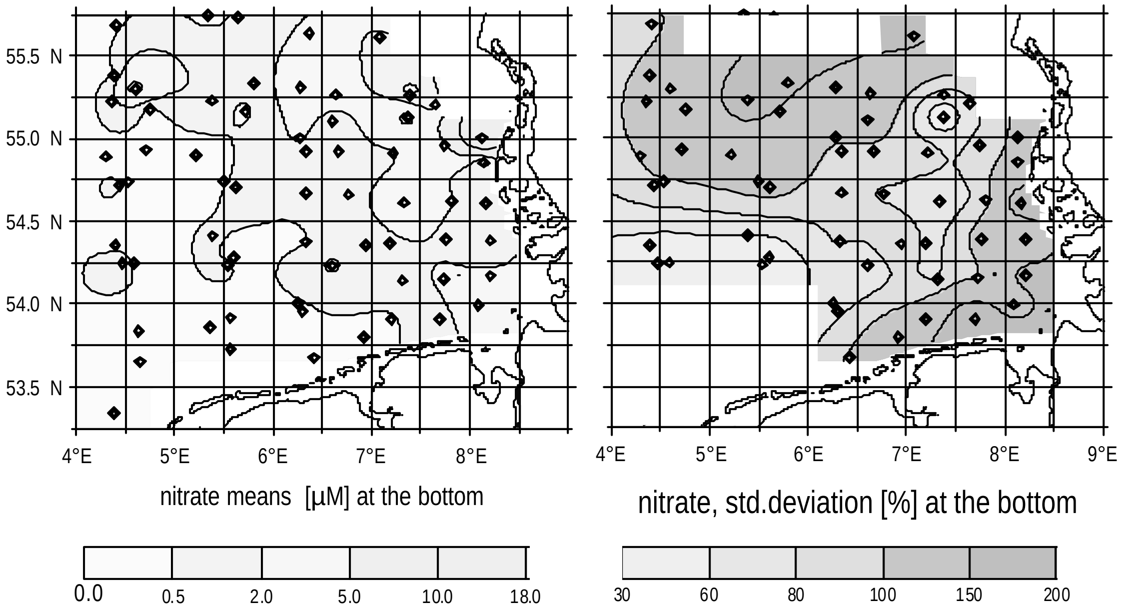


Fig. 2.3.4.4: Mean summer silicate gradients [µM] and their variability [%]



nitrate means [µM] at the surface

nitrate, std.deviation [%] at the surface

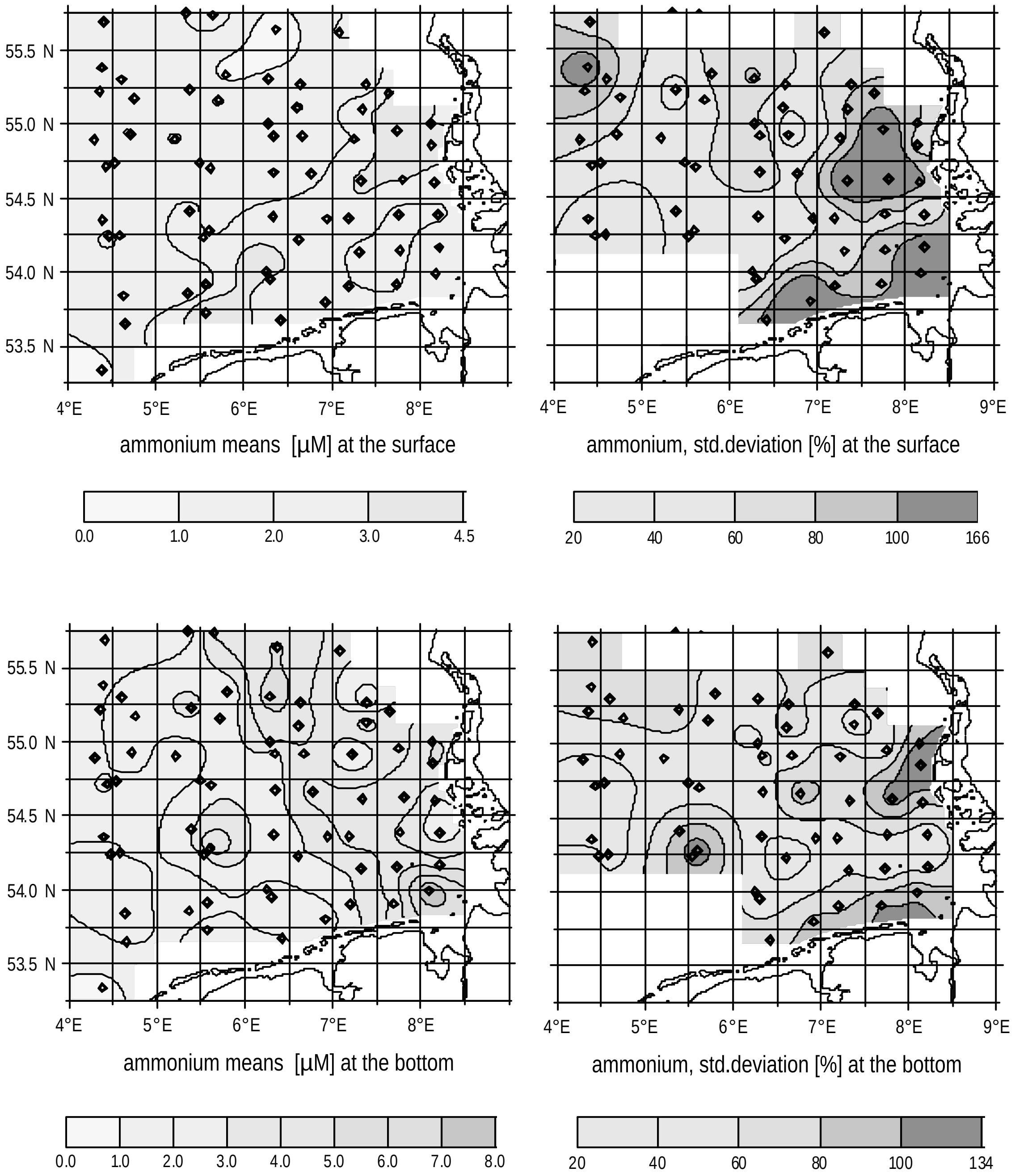


nitrate means [µM] at the bottom

nitrate, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.5: Mean summer nitrate gradients [µM] and their variability [%]



ammonium means [µM] at the surface

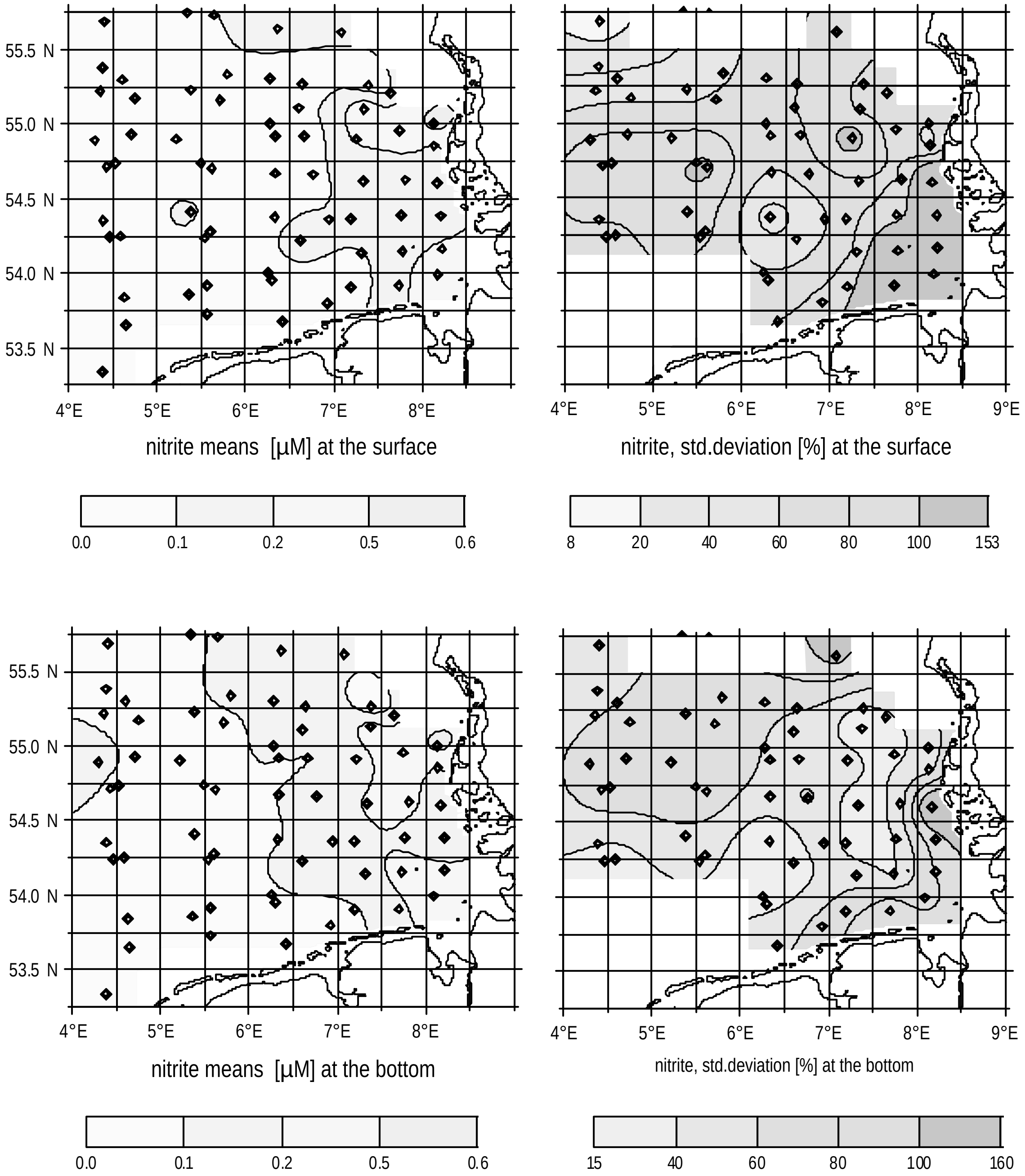
ammonium, std.deviation [%] at the surface

ammonium means [µM] at the bottom

ammonium, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.6: Mean summer ammonium gradients [µM] and their variability [%]



nitrite means [µM] at the surface

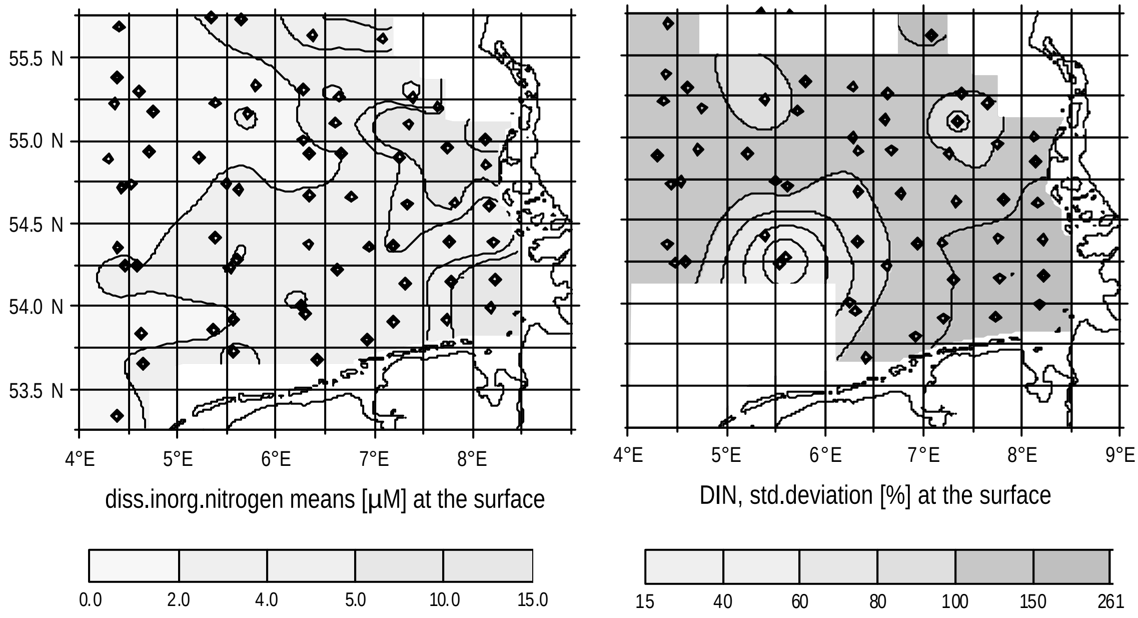
nitrite, std.deviation [%] at the surface

nitrite means [µM] at the bottom

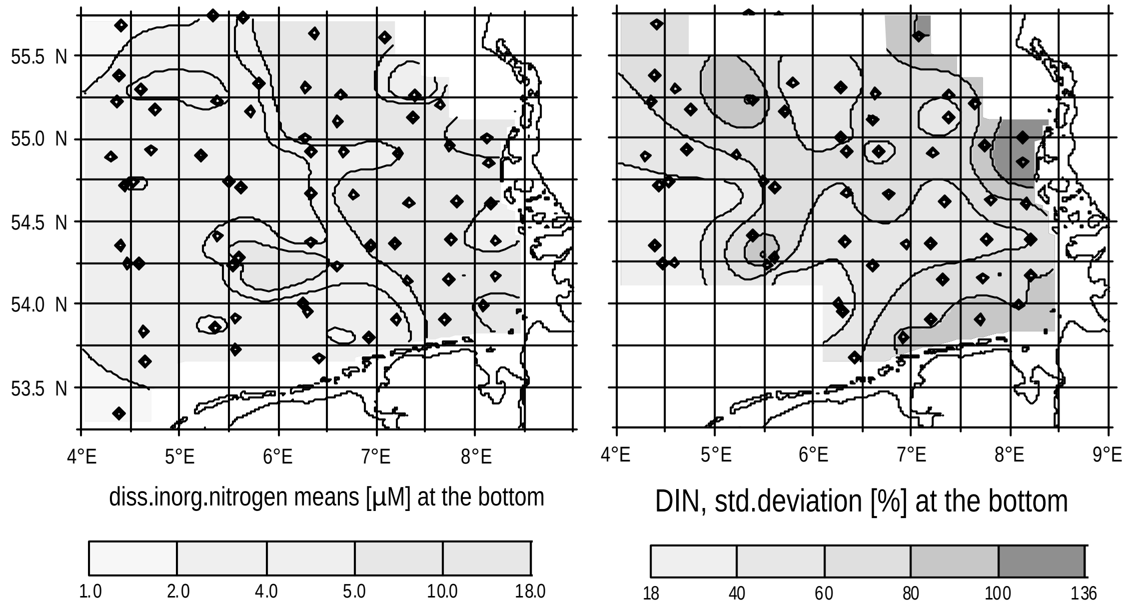
nitrite, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.7: Mean summer nitrite gradients [µM] and their variability [%]


diss.inorg.nitrogen means [µM] at the surface

DIN, std.deviation [%] at the surface


diss.inorg.nitrogen means [µM] at the bottom

DIN, std.deviation [%] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.8: Mean summer DIN gradients [µM] and their variability [%]

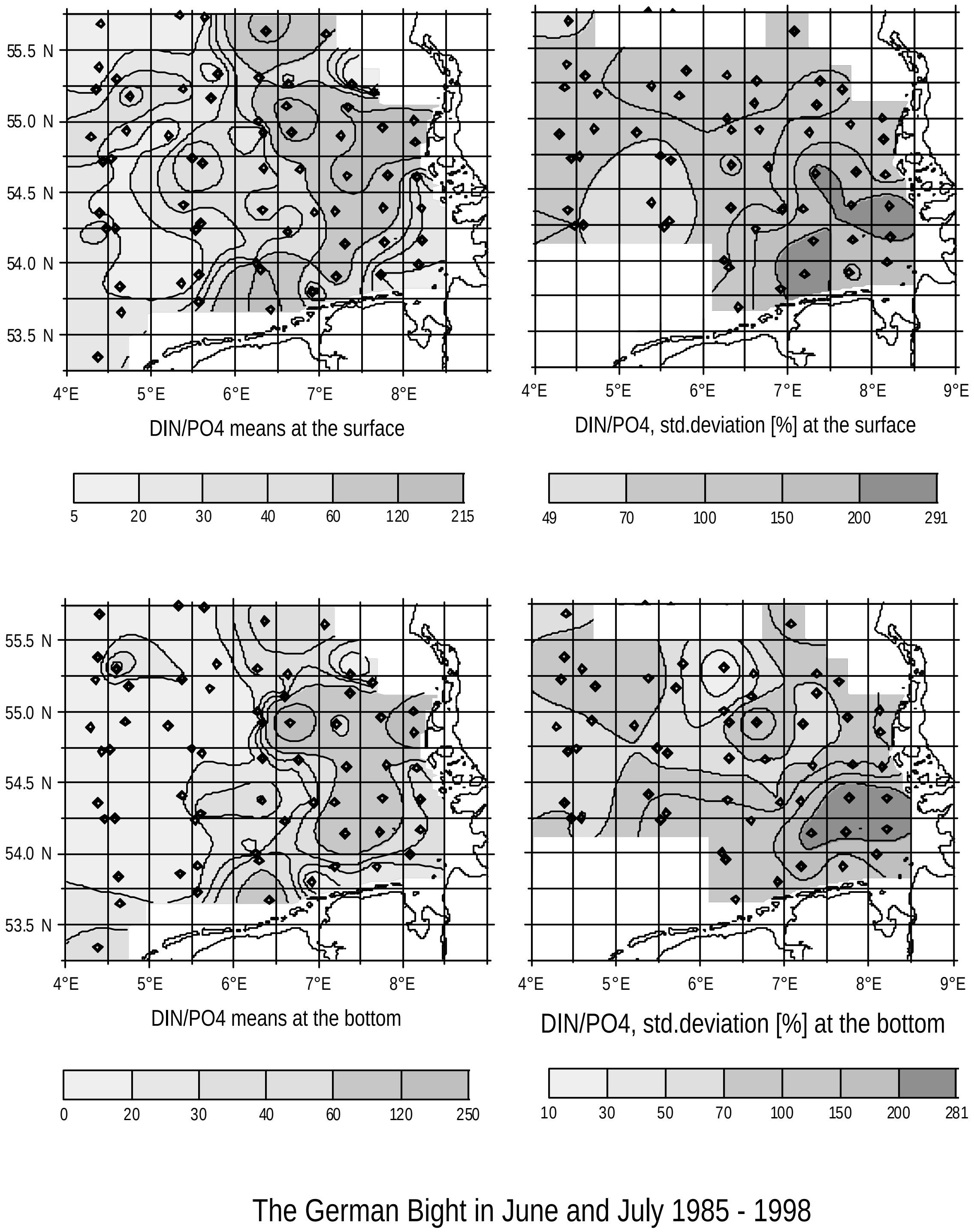
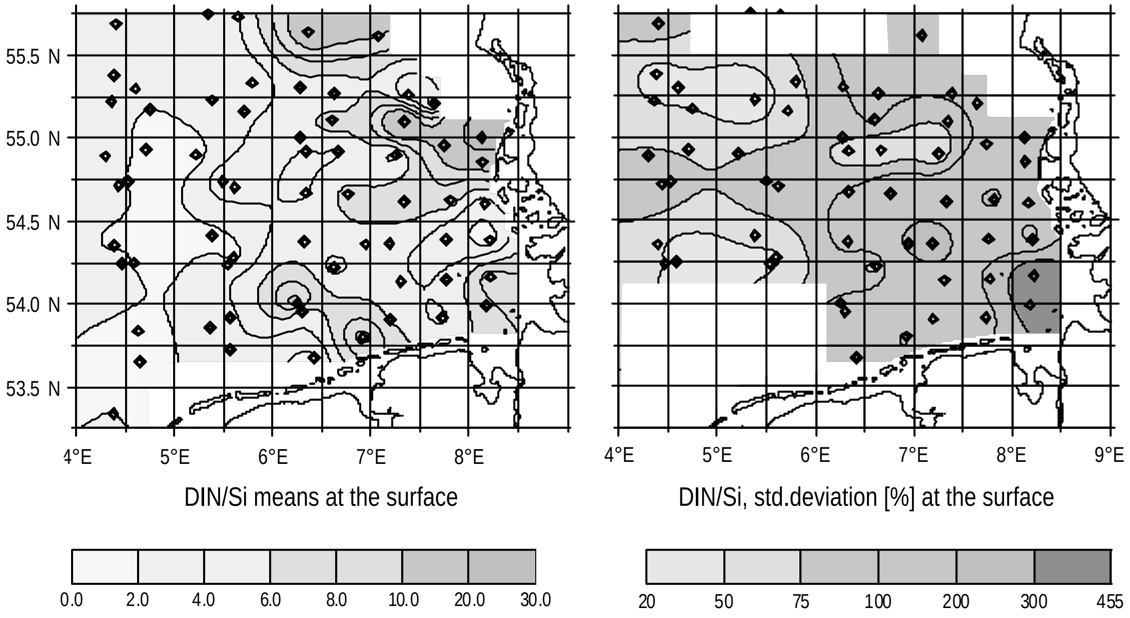
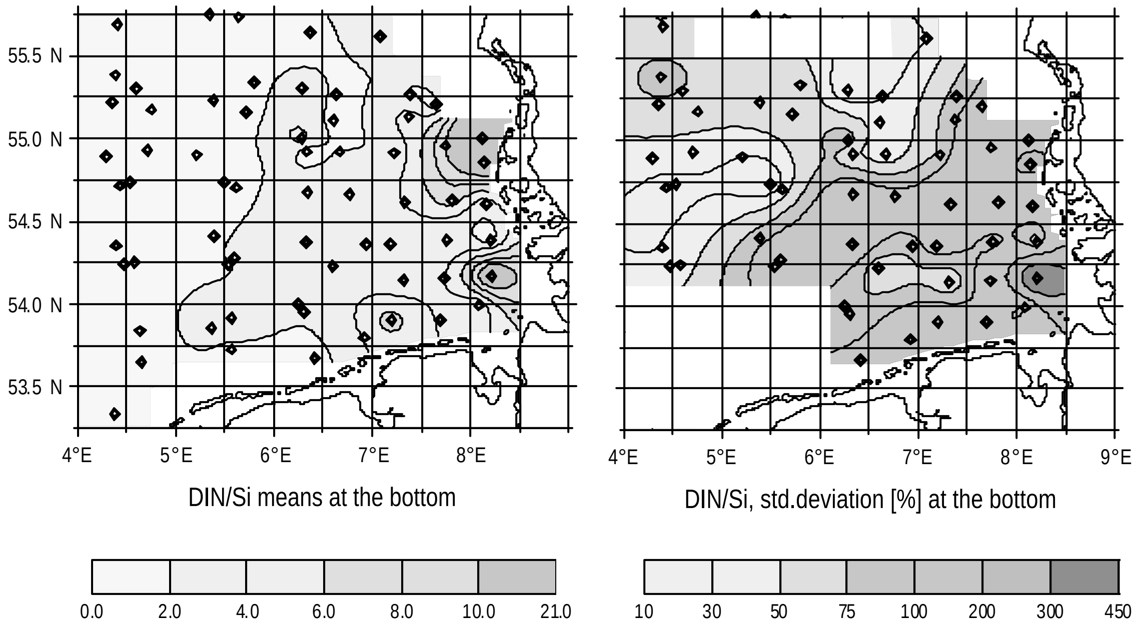


Fig. 2.3.4.9: Mean summer N/P ratios [M/M] and their variability [%]



DIN/Si means at the surface

DIN/Si, std.deviation [%] at the surface

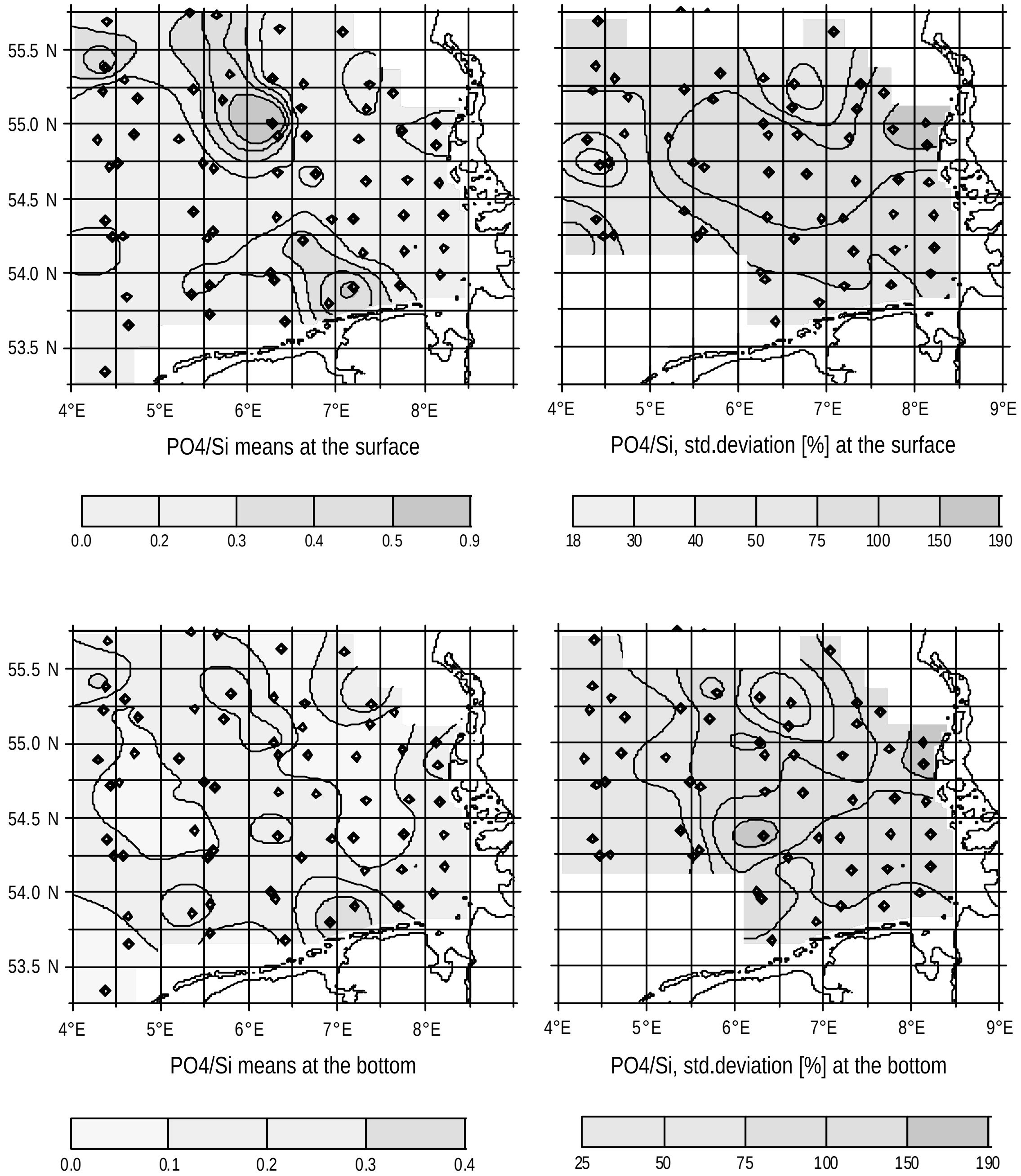


DIN/Si means at the bottom

DIN/Si, std.deviation [%] at the bottom

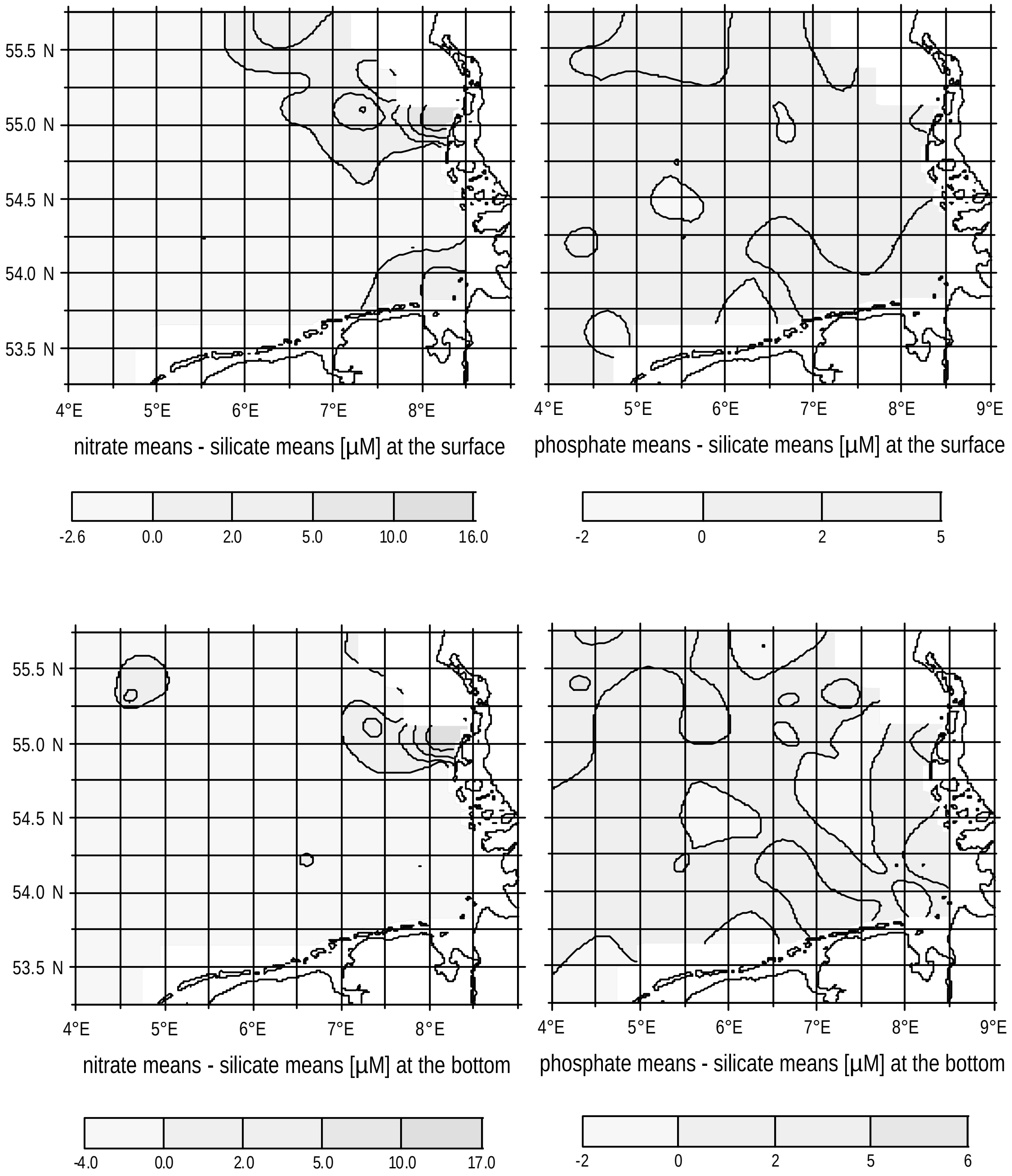
The German Bight in June and July 1985 - 1998

Fig. 2.3.4.10: Mean summer N/Si ratios [M/M] and their variability [%]



The German Bight in June and July 1985 - 1998

Fig. 2.3.4.11: Mean summer P/Si ratios [M/M] and their variability [%]



nitrate means - silicate means [µM] at the surface

phosphate means - silicate means [µM] at the surface

nitrate means - silicate means [µM] at the bottom

phosphate means - silicate means [µM] at the bottom

The German Bight in June and July 1985 - 1998

Fig. 2.3.4.12: Differences between nitrate and silicate (left) and phosphate and 1/16 silicate (µM) during summer, indicating possible anthropogenic contributions to nitrate and phosphate in relation to silicate, assuming Redfield ratios of 1:1 and 1:16 (Redfield et al. 1963)

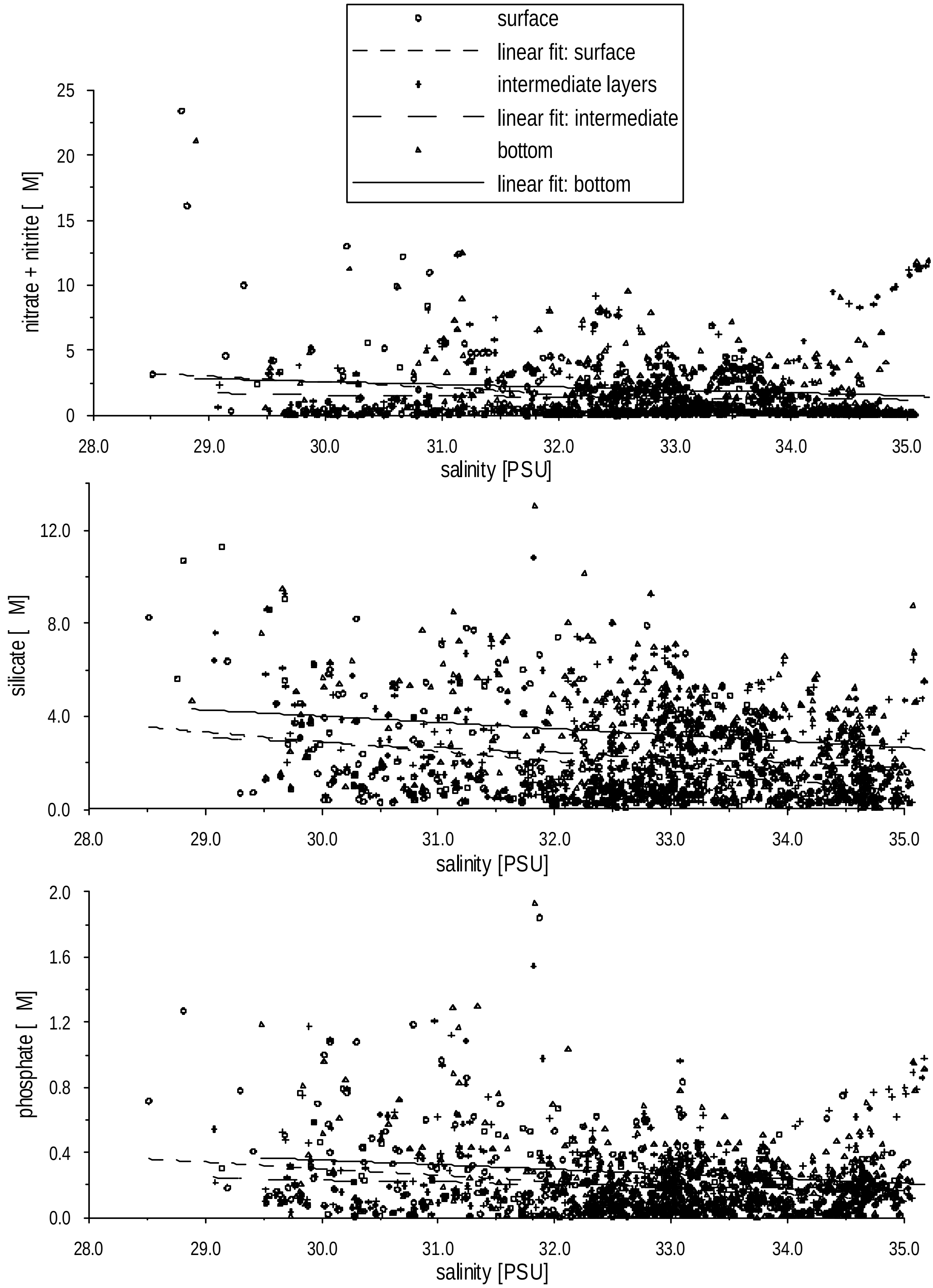


Fig. 2.3.5.1: Mixing diagrams of nitrate + nitrite, silicate and phosphate during summer in the German Bight 1985 - 1998

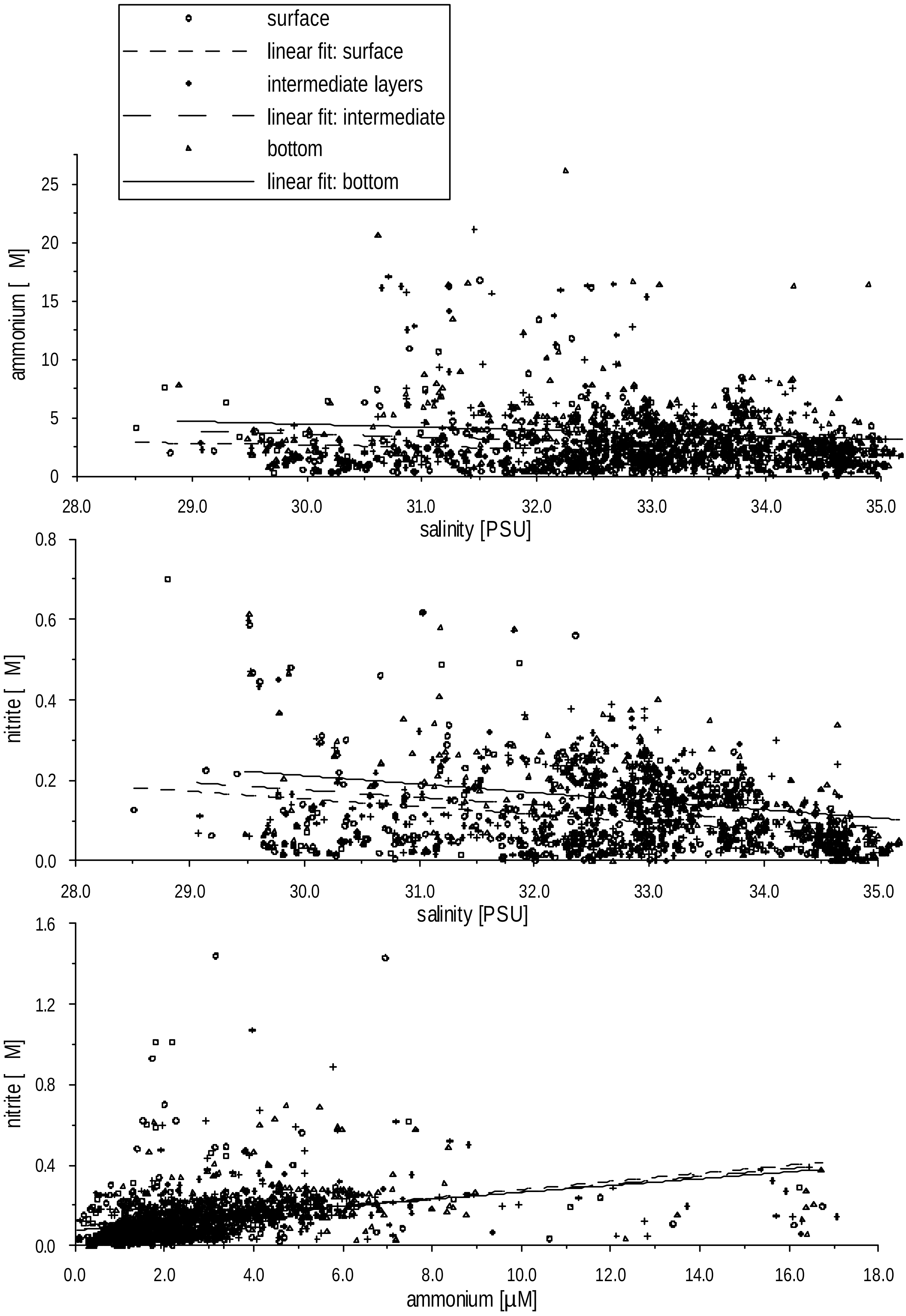


Fig. 2.3.5.2: Mixing diagrams of ammonium and nitrite and correlation diagrams of nitrite with ammonium during summer in the German Bight 1985 - 1998

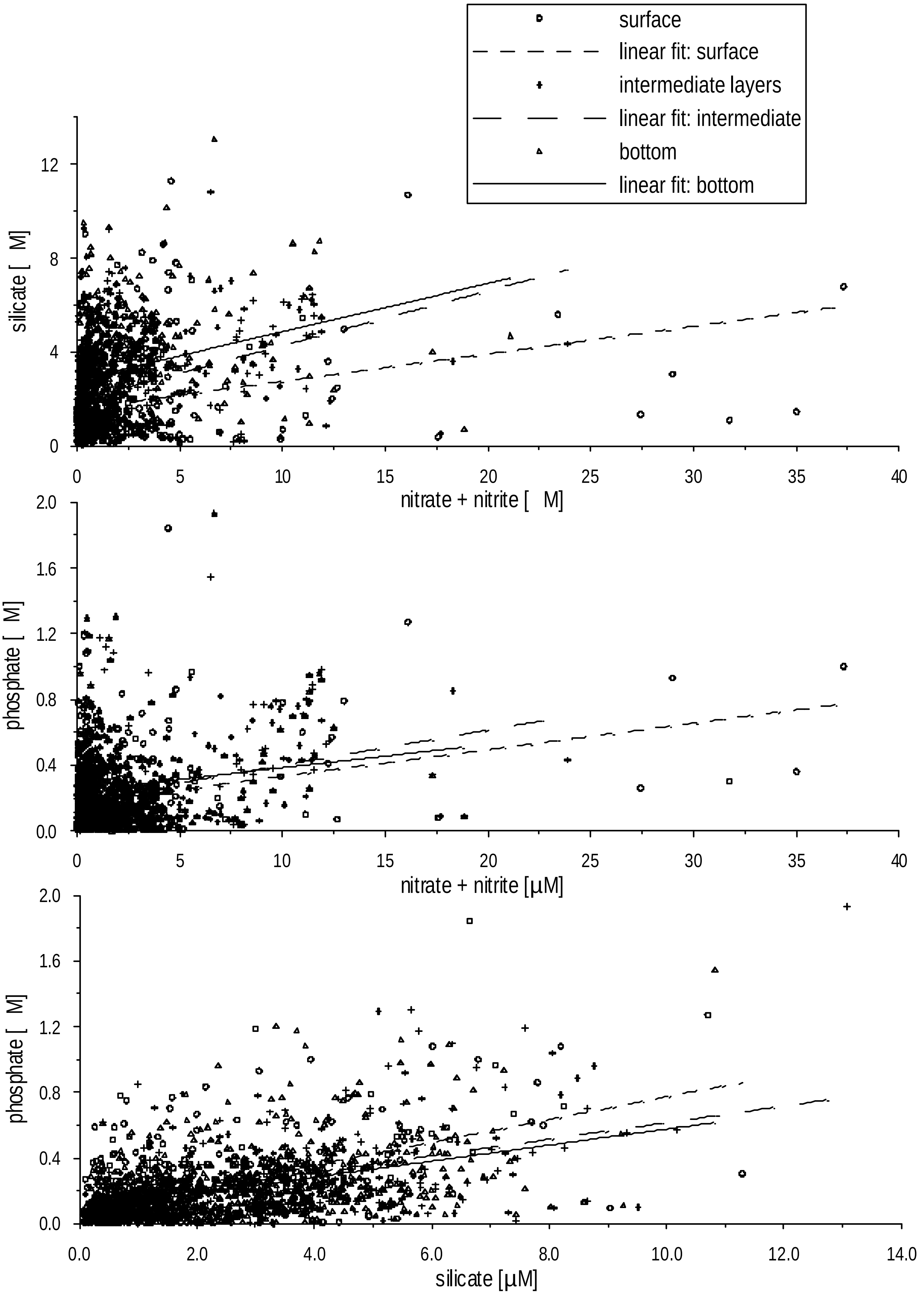


Fig. 2.3.5.3: Correlation diagrams of silicate with nitrate + nitrite, phosphate with nitrate + nitrite and phosphate with silicate during summer in the German Bight 1985 –1998

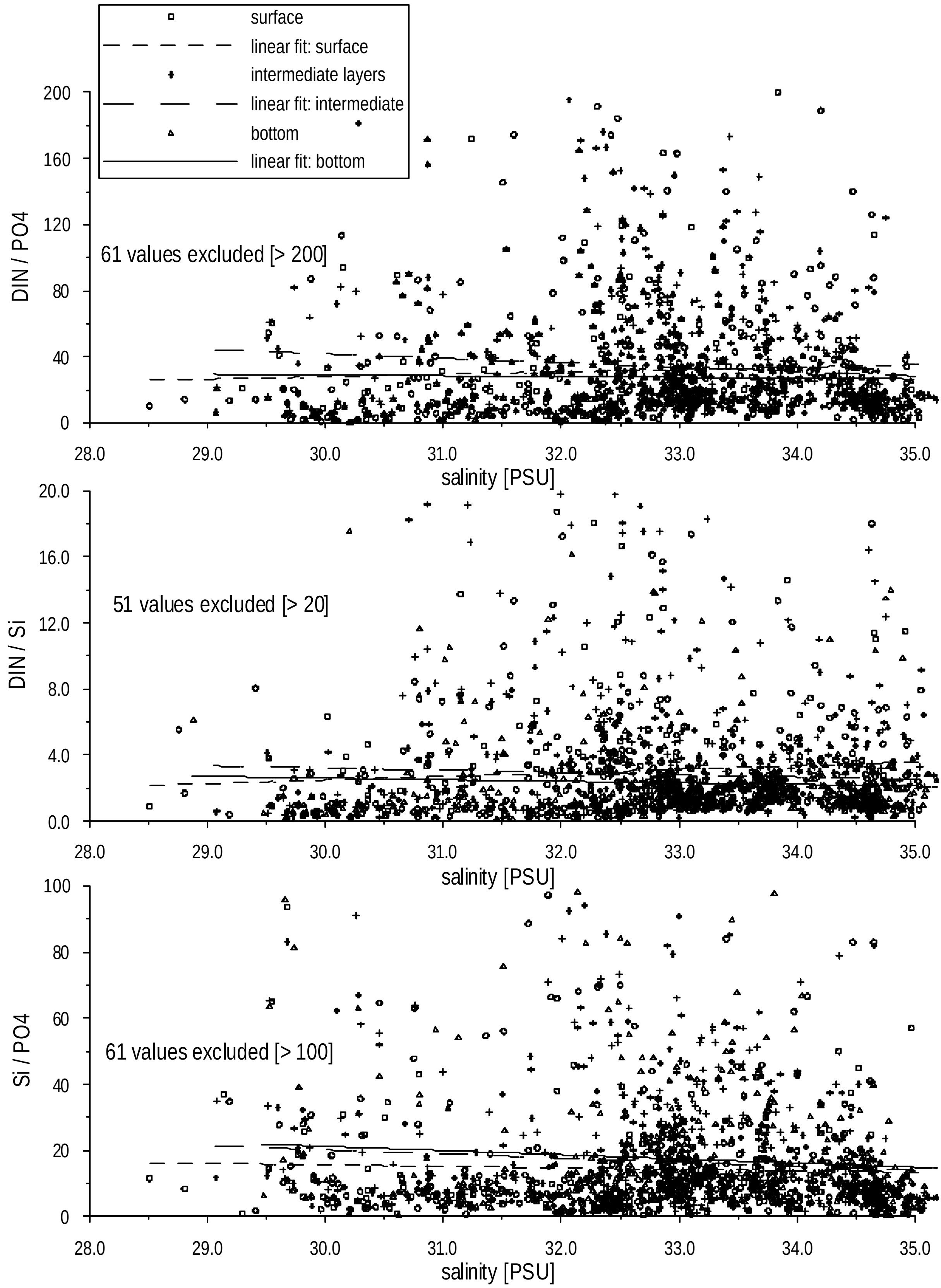


Fig. 2.3.5.4: Mixing diagrams of DIN/P, DIN/Si and Si/P ratios [M/M] during summer in the German Bight 1985 - 1998

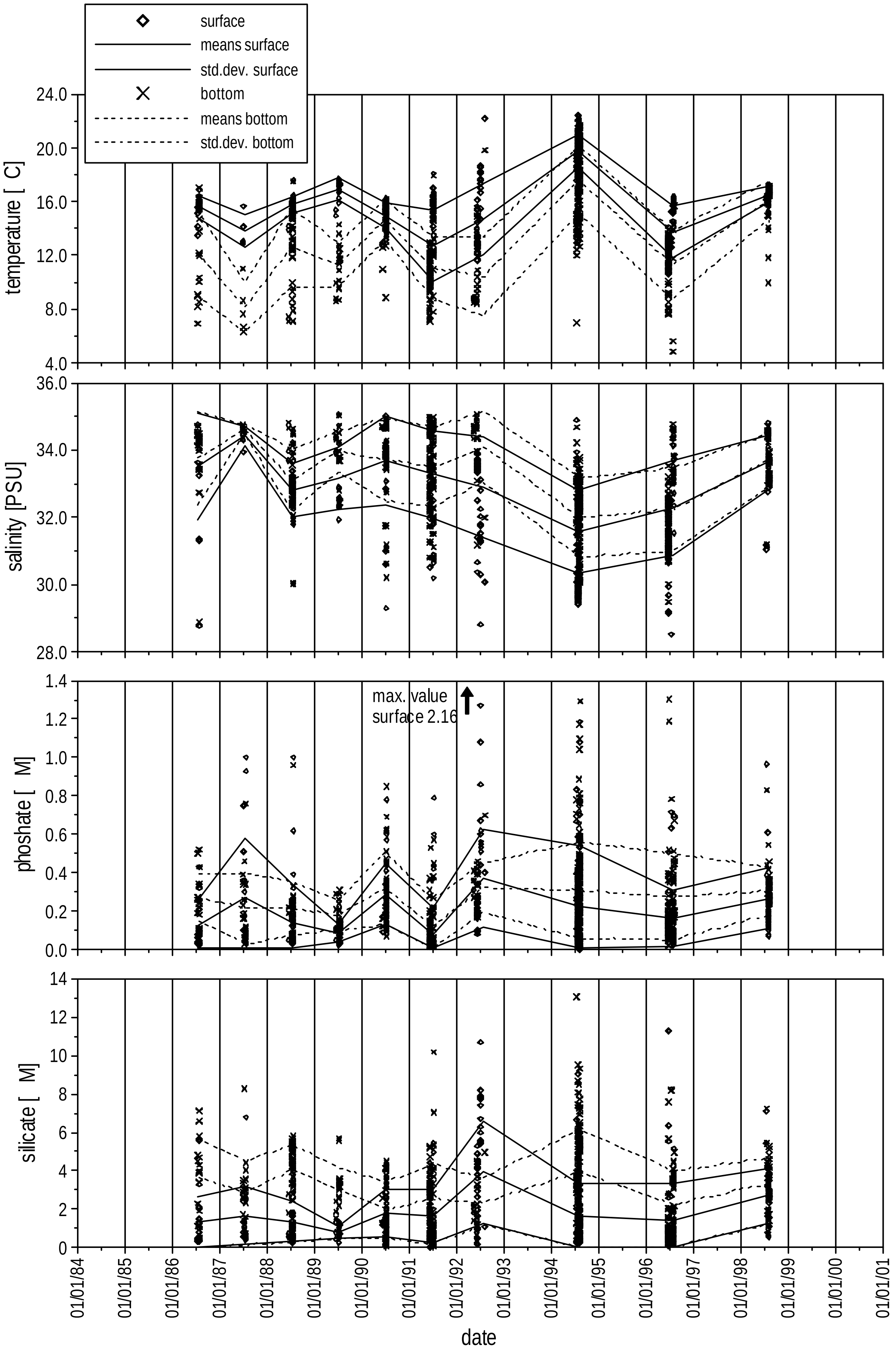


Fig. 2.3.6.1: Time series of summer concentrations in the total German Bight 1985 – 1998

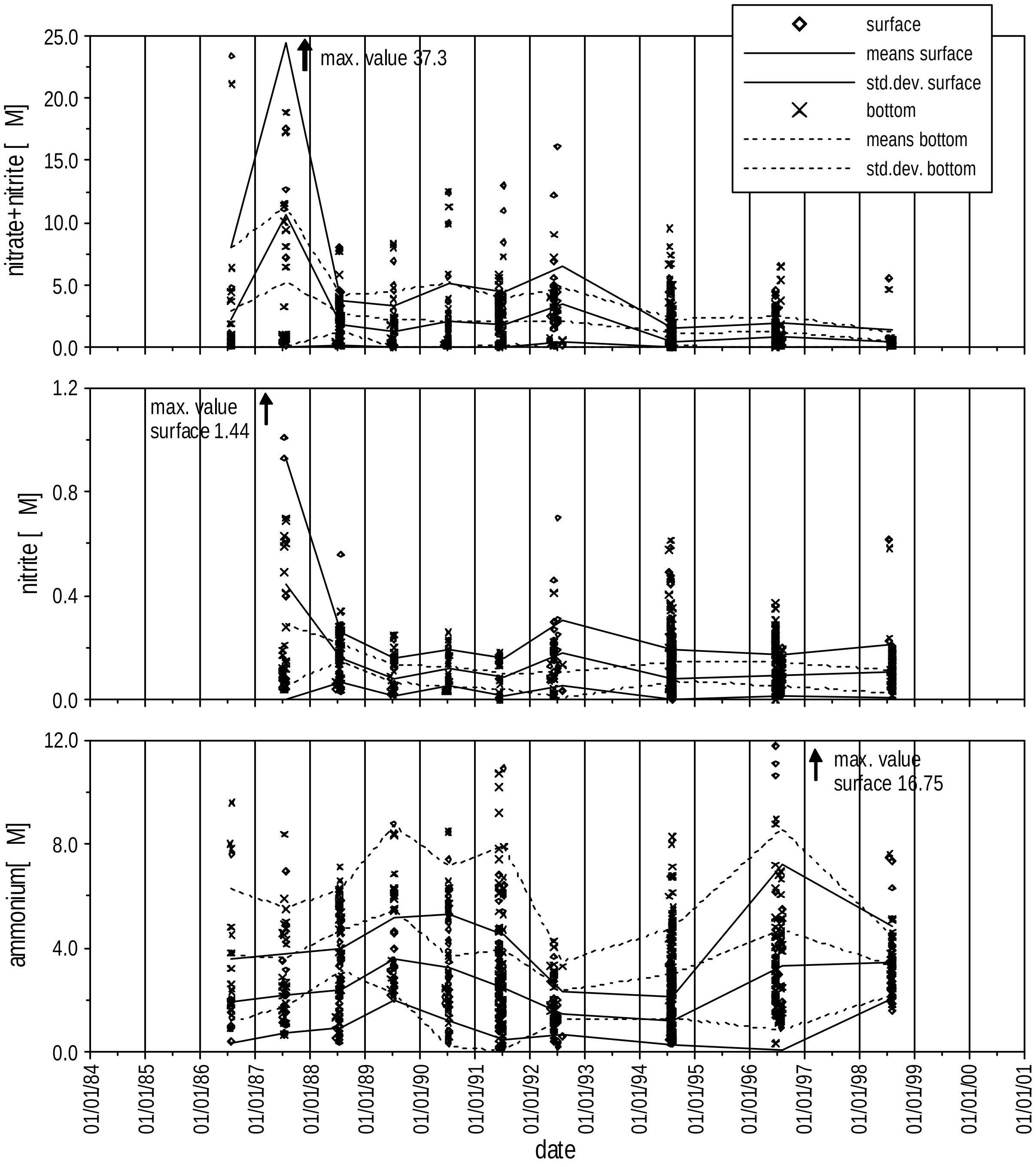


Fig. 2.3.6.2: Time series of summer concentrations in the total German Bight 1985 – 1998

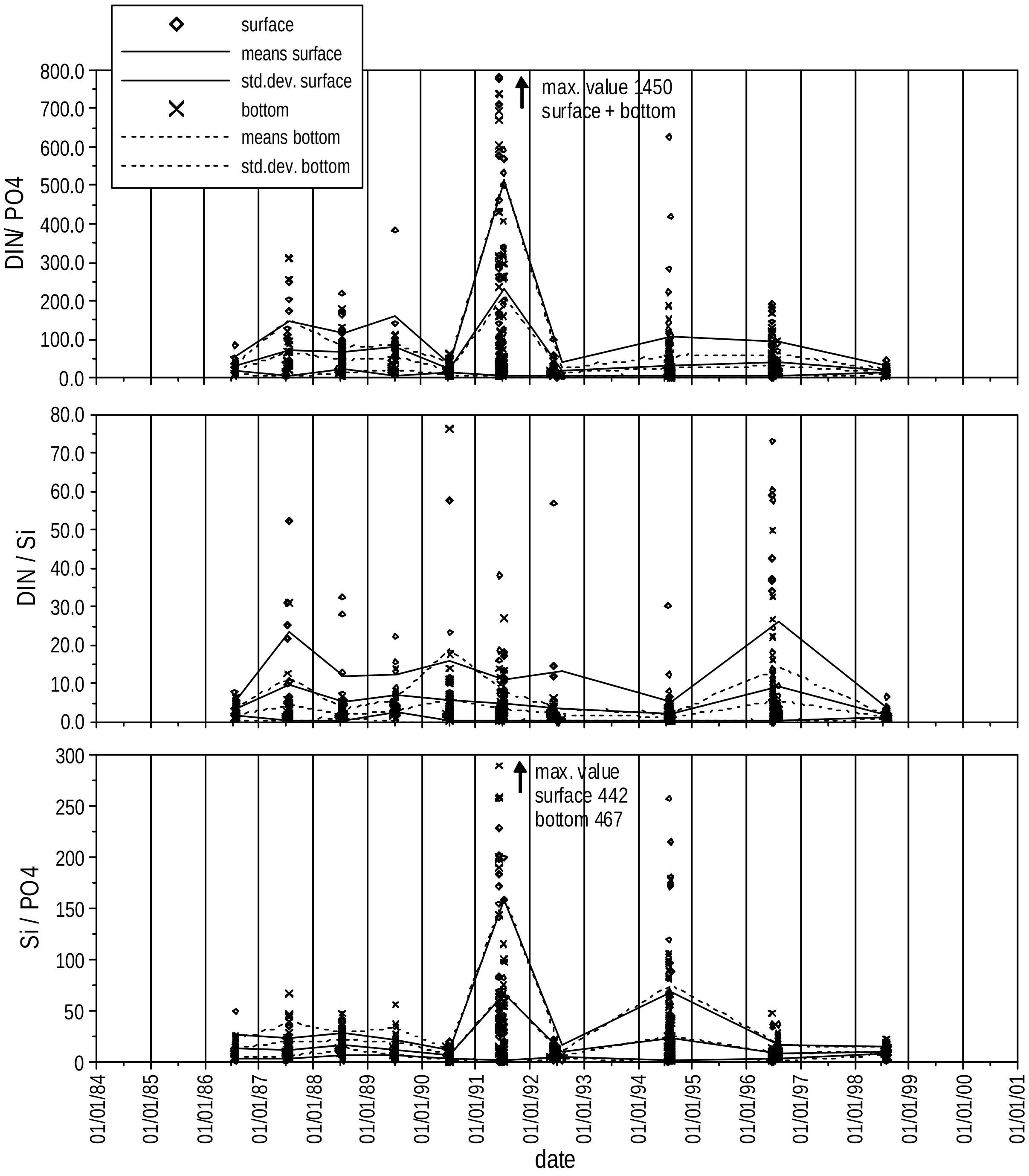


Fig. 2.3.6.3: Time series of nutrient ratios [M/M] in the total German Bight 1985 – 1998 during summer

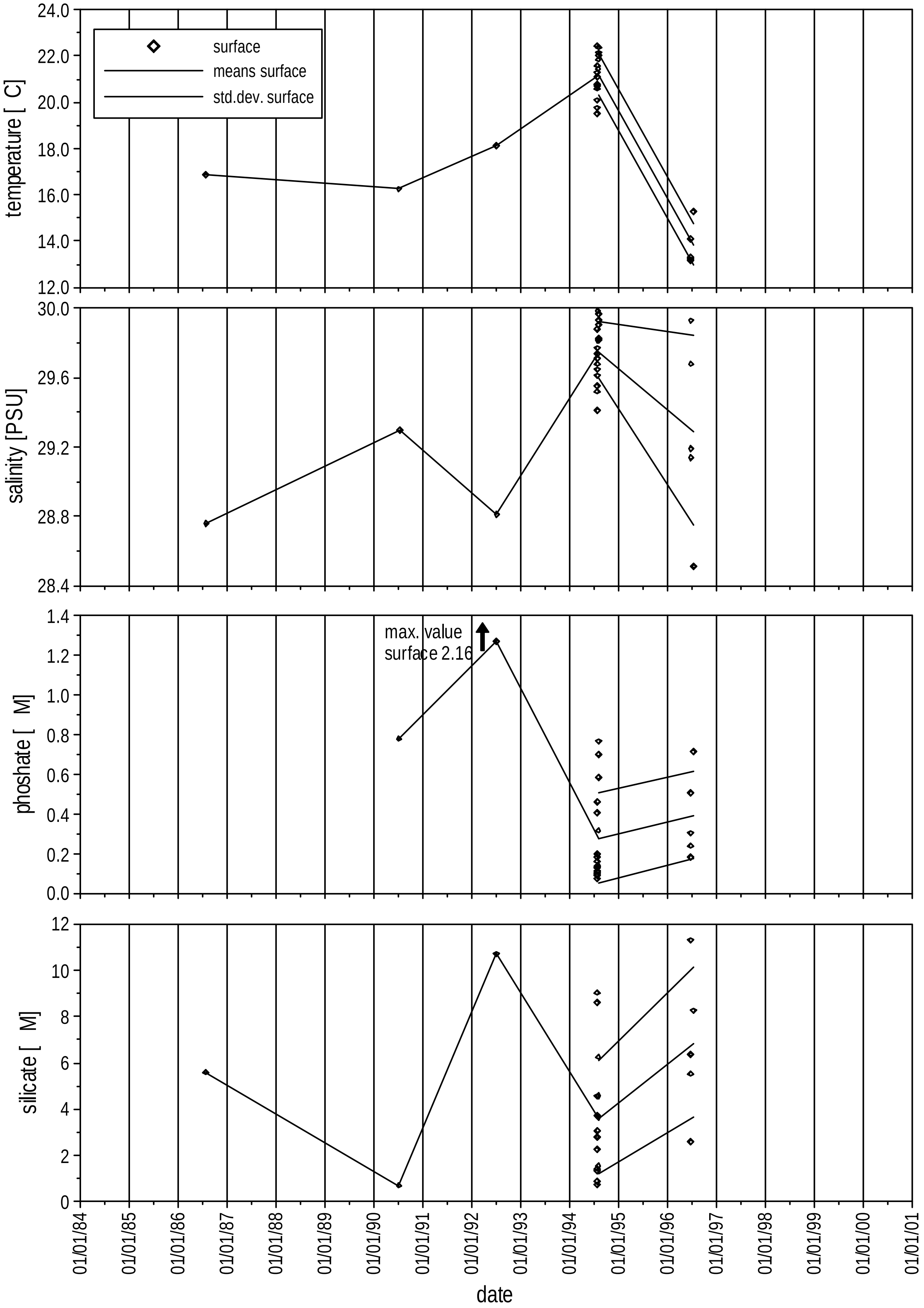


Fig. 2.3.6.4: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 21 – 30

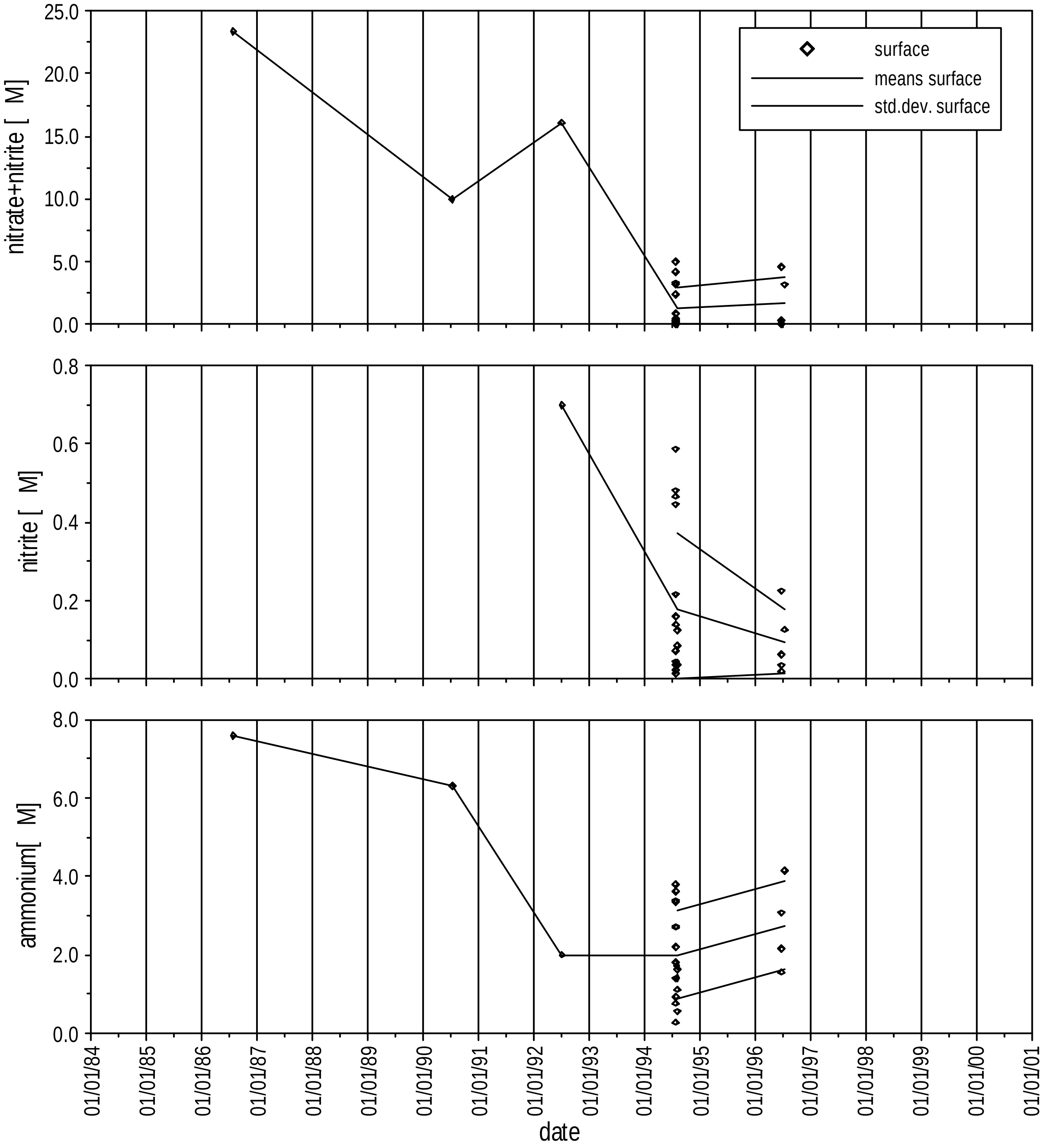


Fig. 2.3.6.5: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 21 – 30

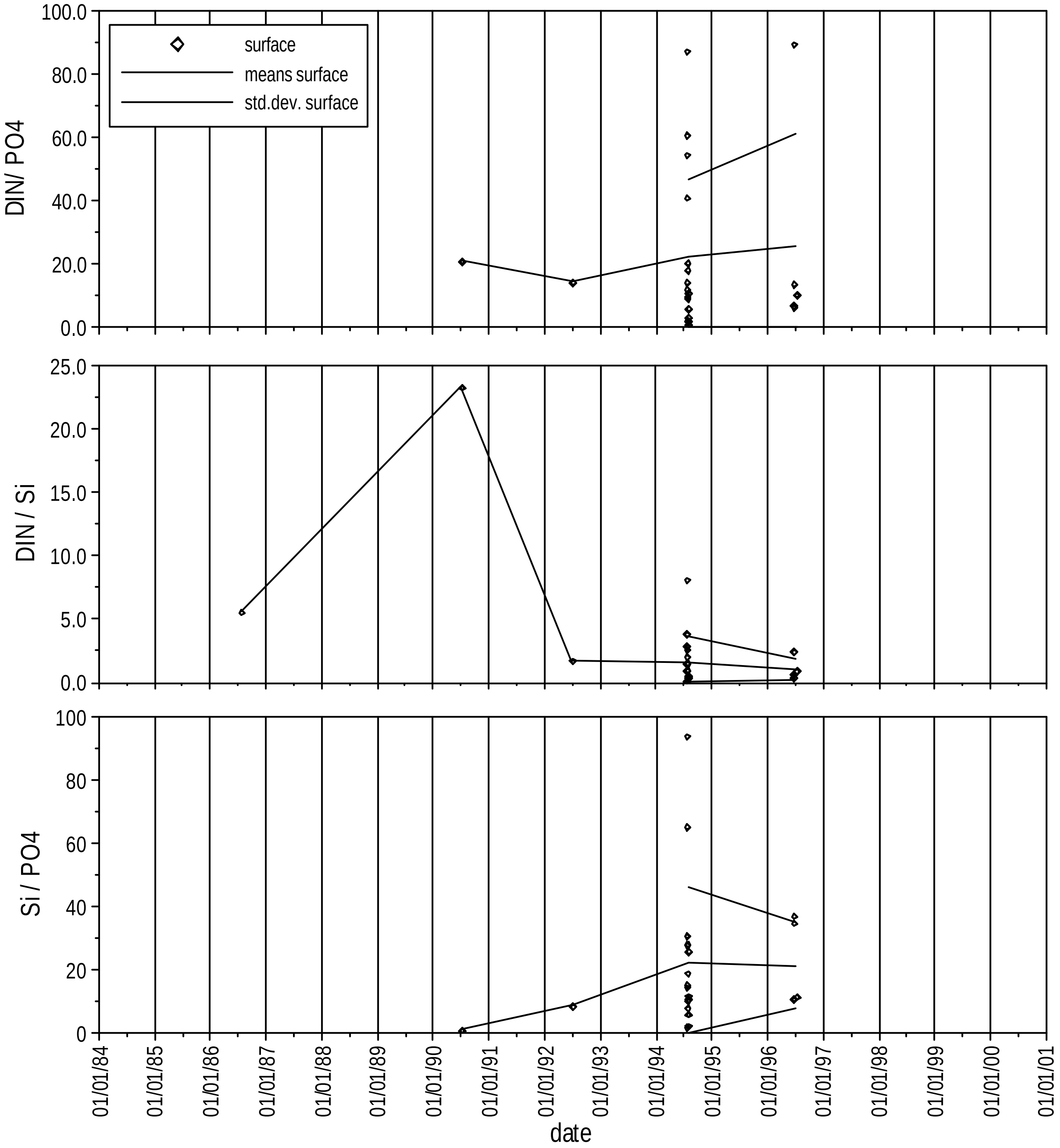


Fig. 2.3.6.6: Time series of nutrient ratios [M/M] in the German Bight 1985 – 1998 at the surface and at salinities of 21 – 30 during summer

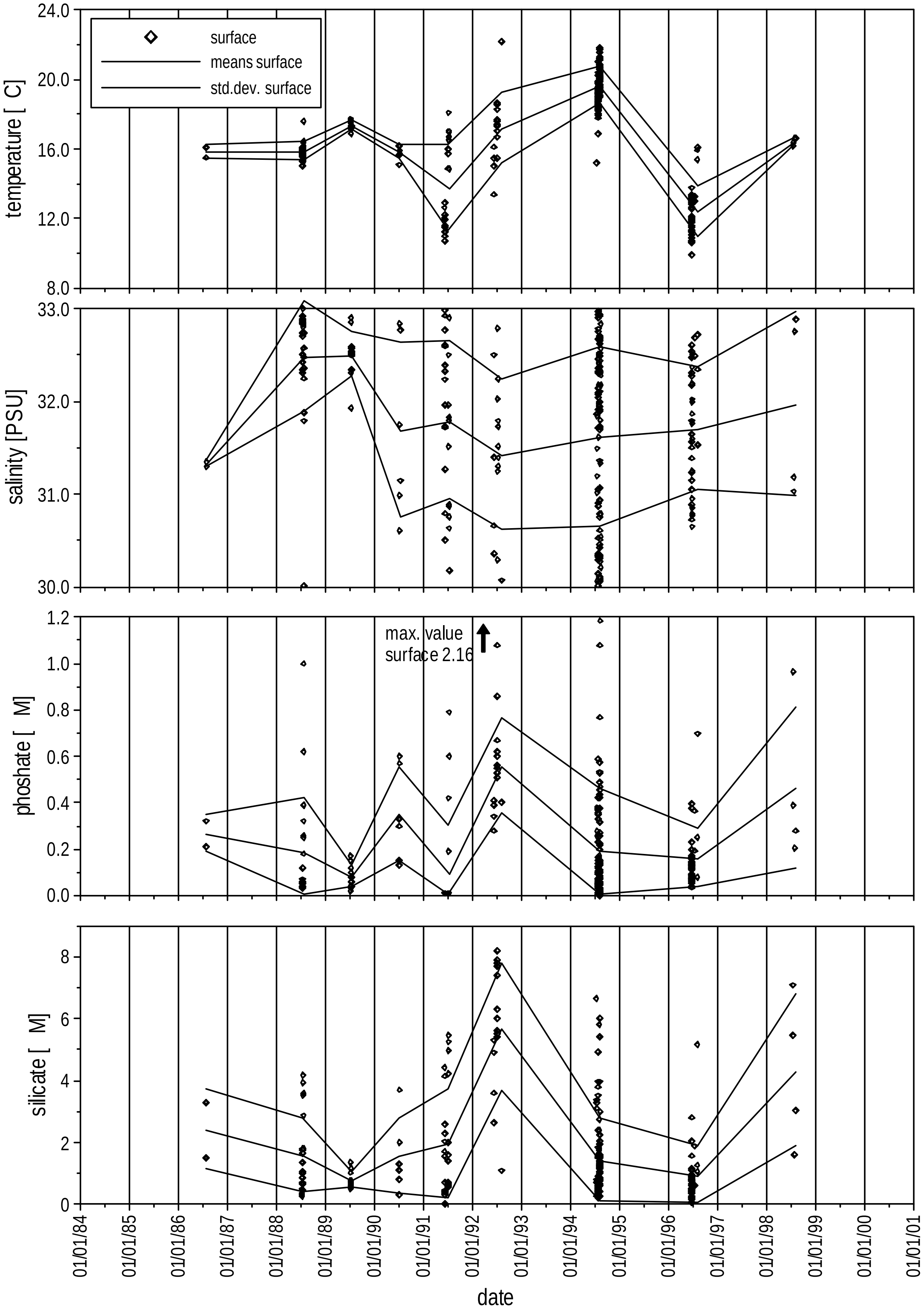


Fig. 2.3.6.7: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 30 – 33

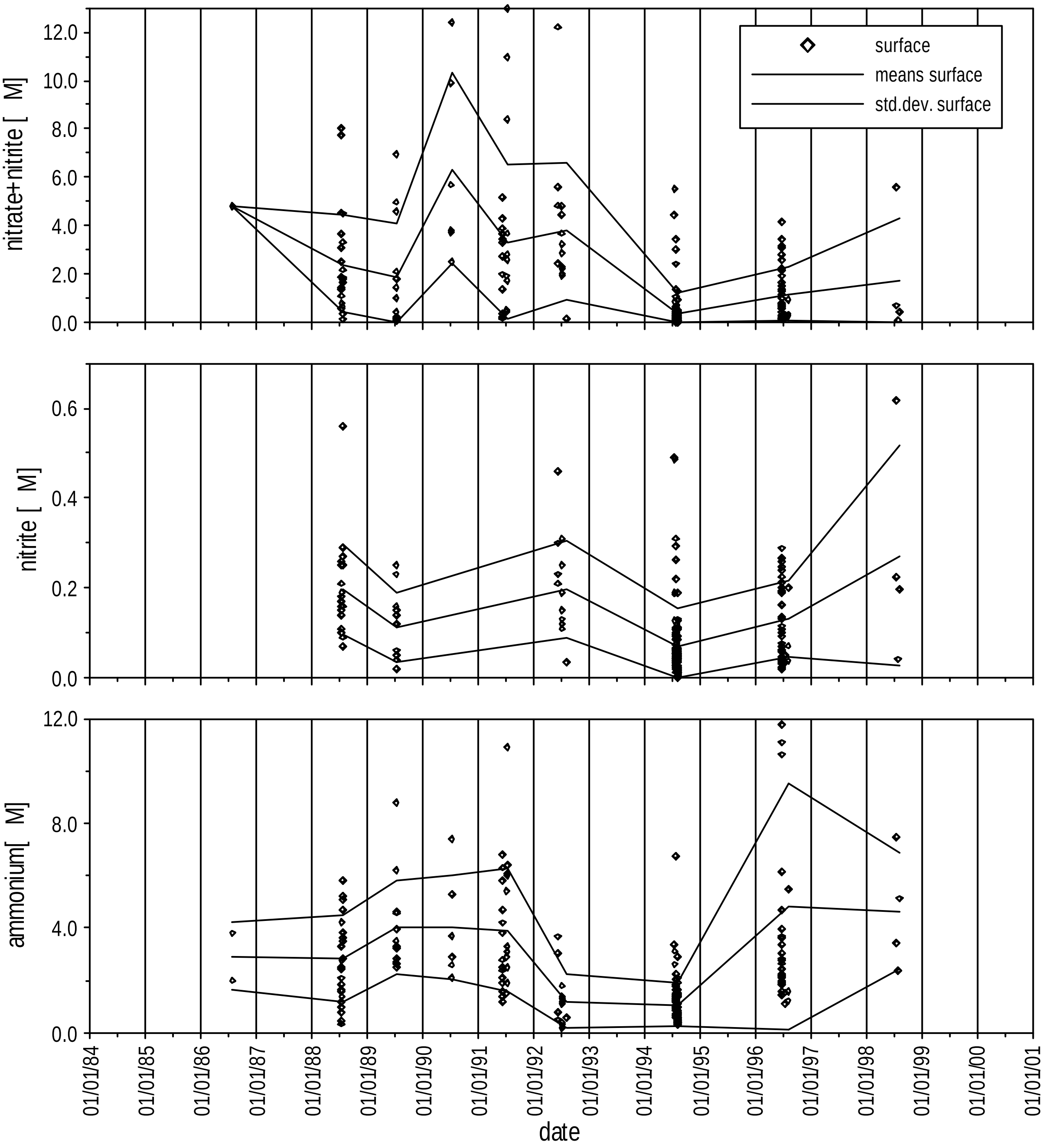


Fig. 2.3.6.8: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 30 – 33

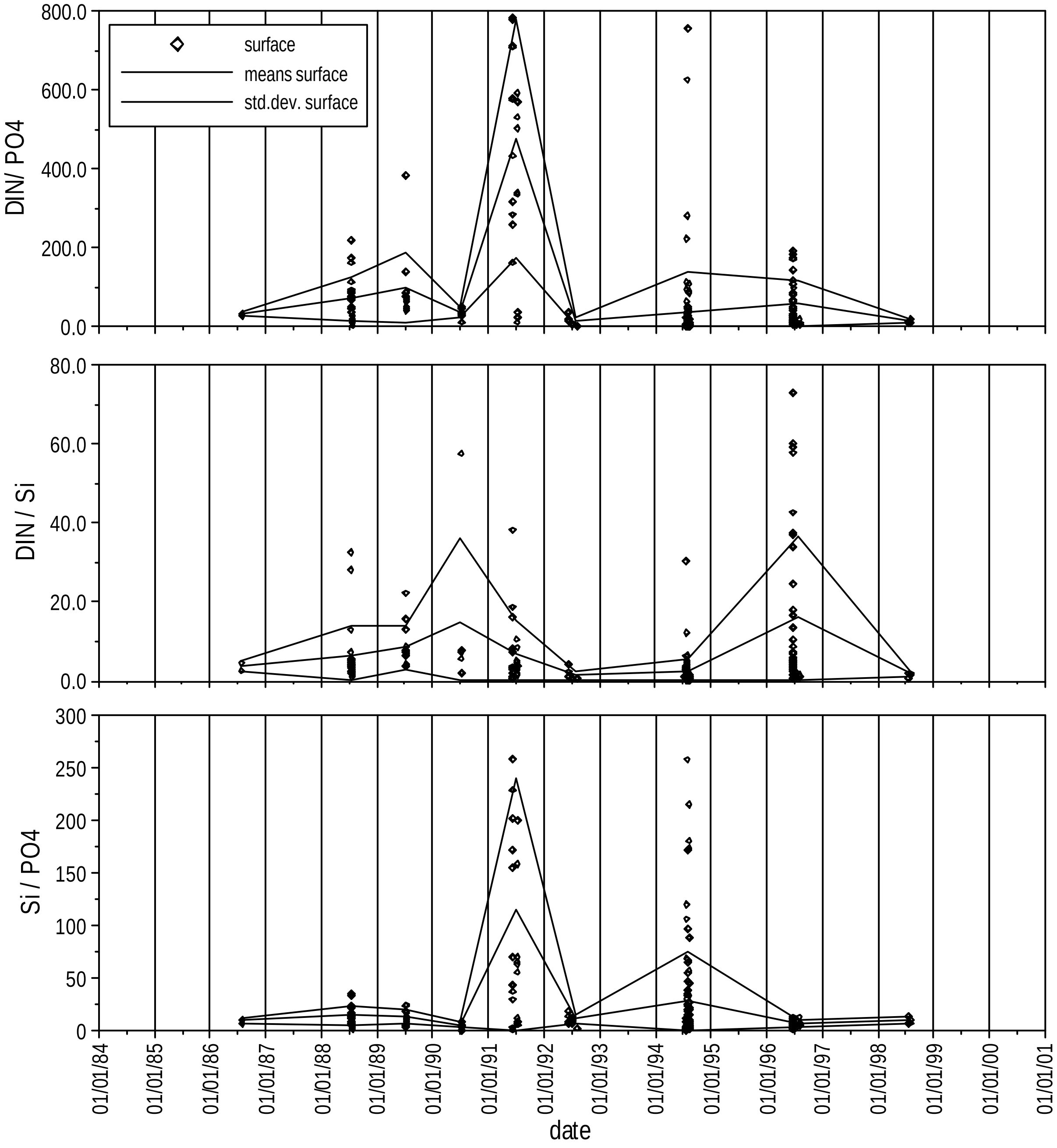


Fig. 2.3.6.9: Time series of nutrient ratios [M/M] in the German Bight 1985 – 1998 at the surface and at salinities of 30 – 33 during summer

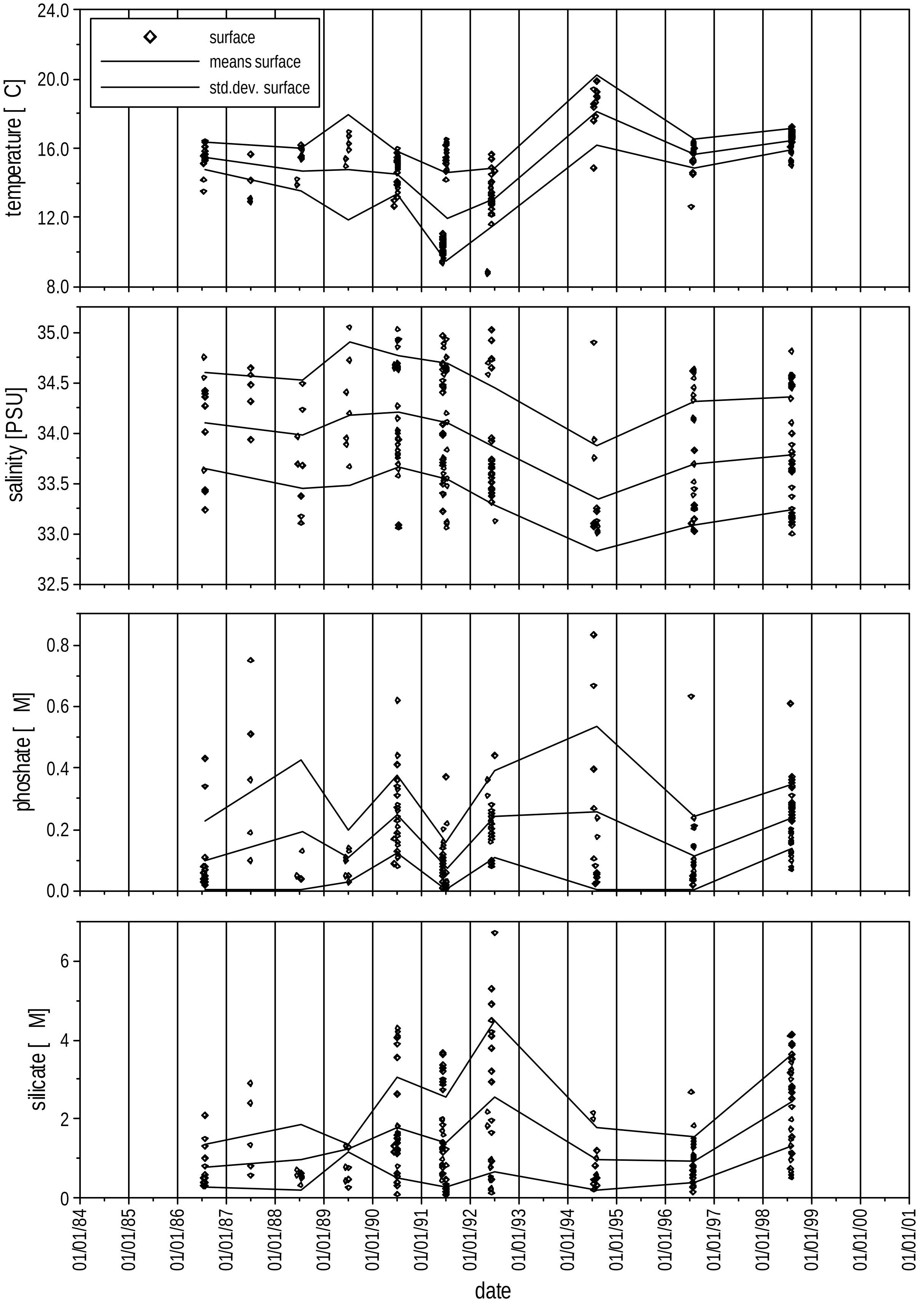


Fig. 2.3.6.10: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 33 – 35

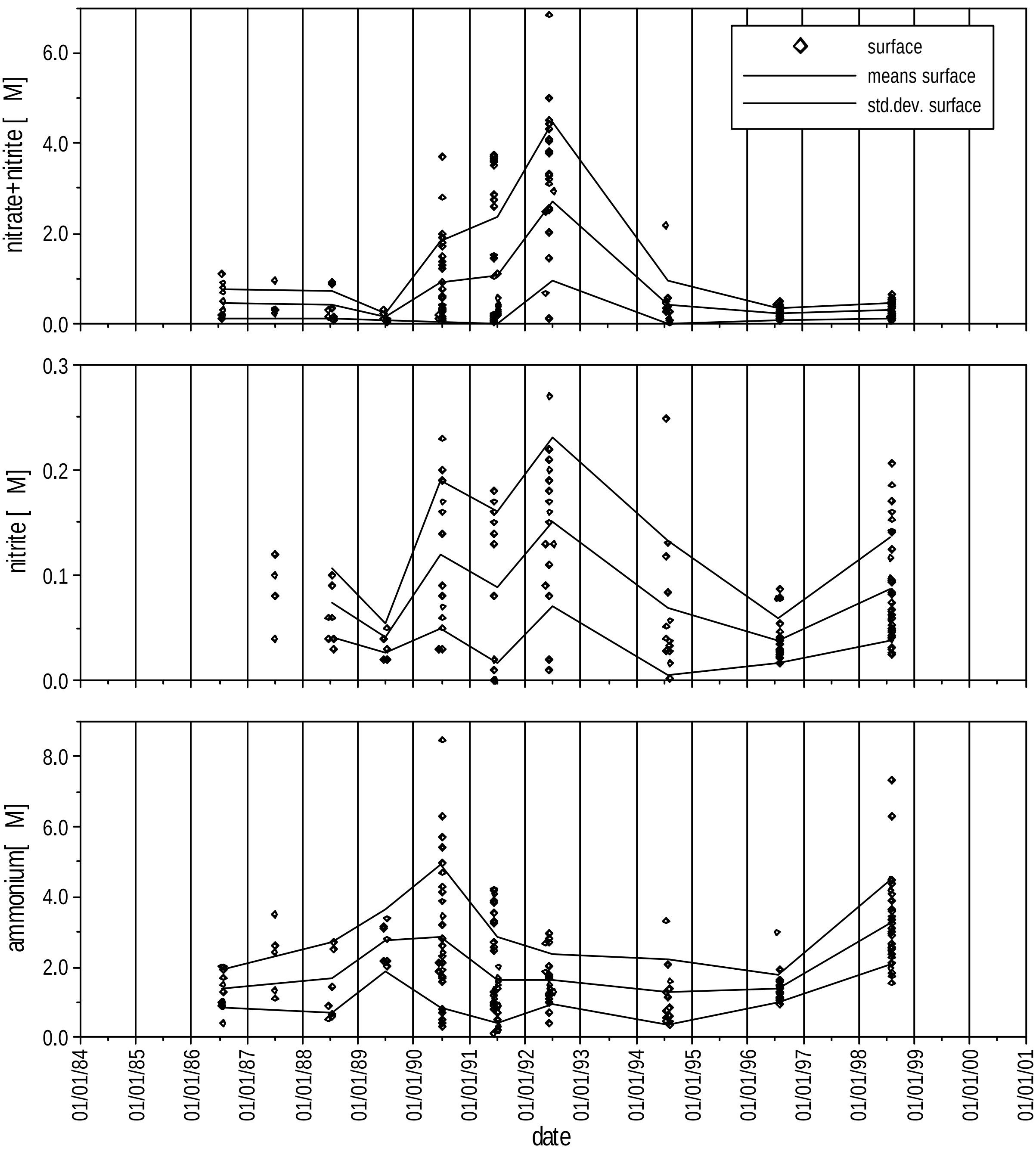


Fig. 2.3.6.11: Time series of summer concentrations in the German Bight 1985 – 1998 at the surface and at salinities of 33 – 35

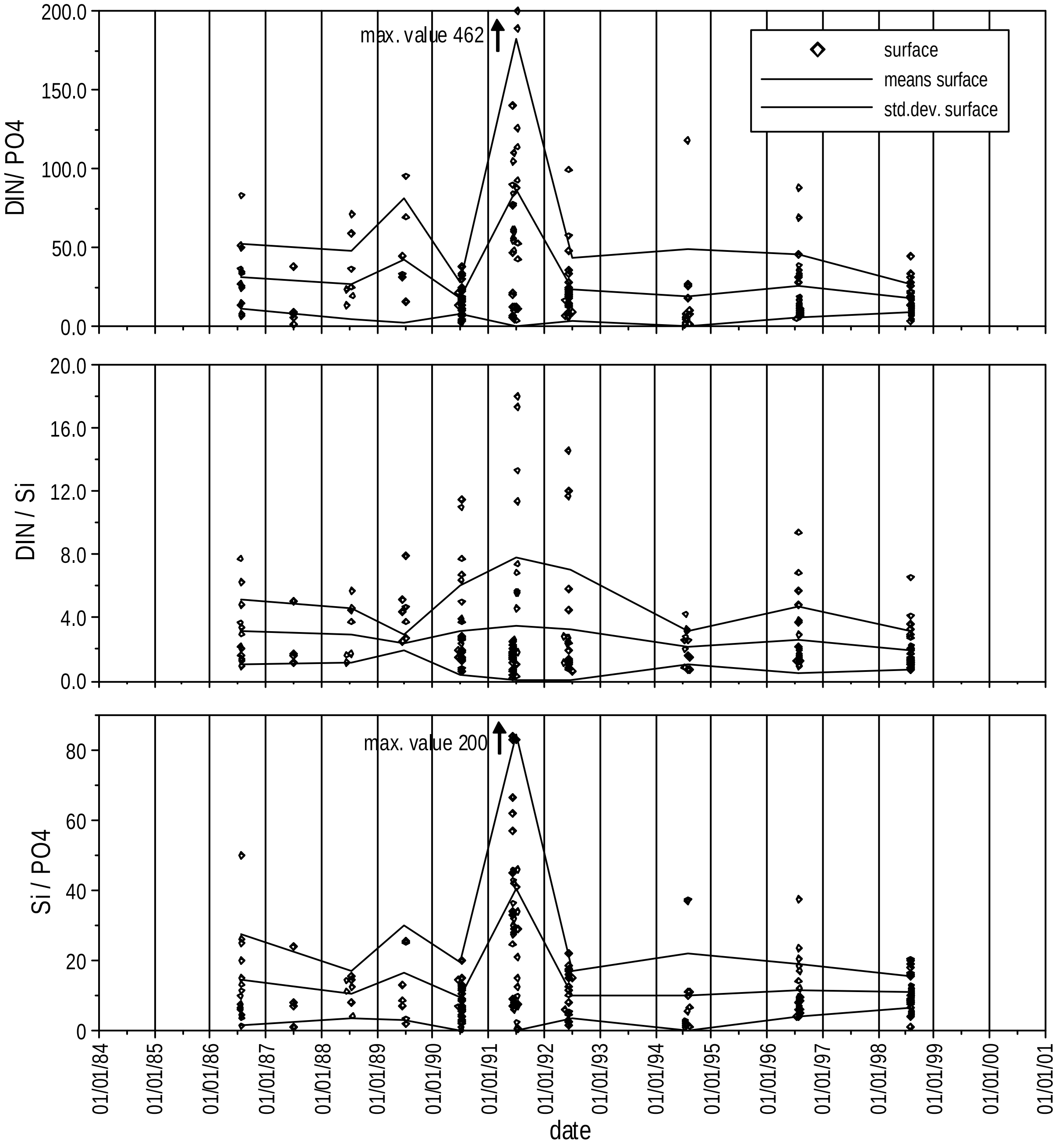


Fig. 2.3.6.12: Time series of nutrient ratios [M/M] in the German Bight 1985 – 1998 at the surface and at salinities of 33 – 35 during summer

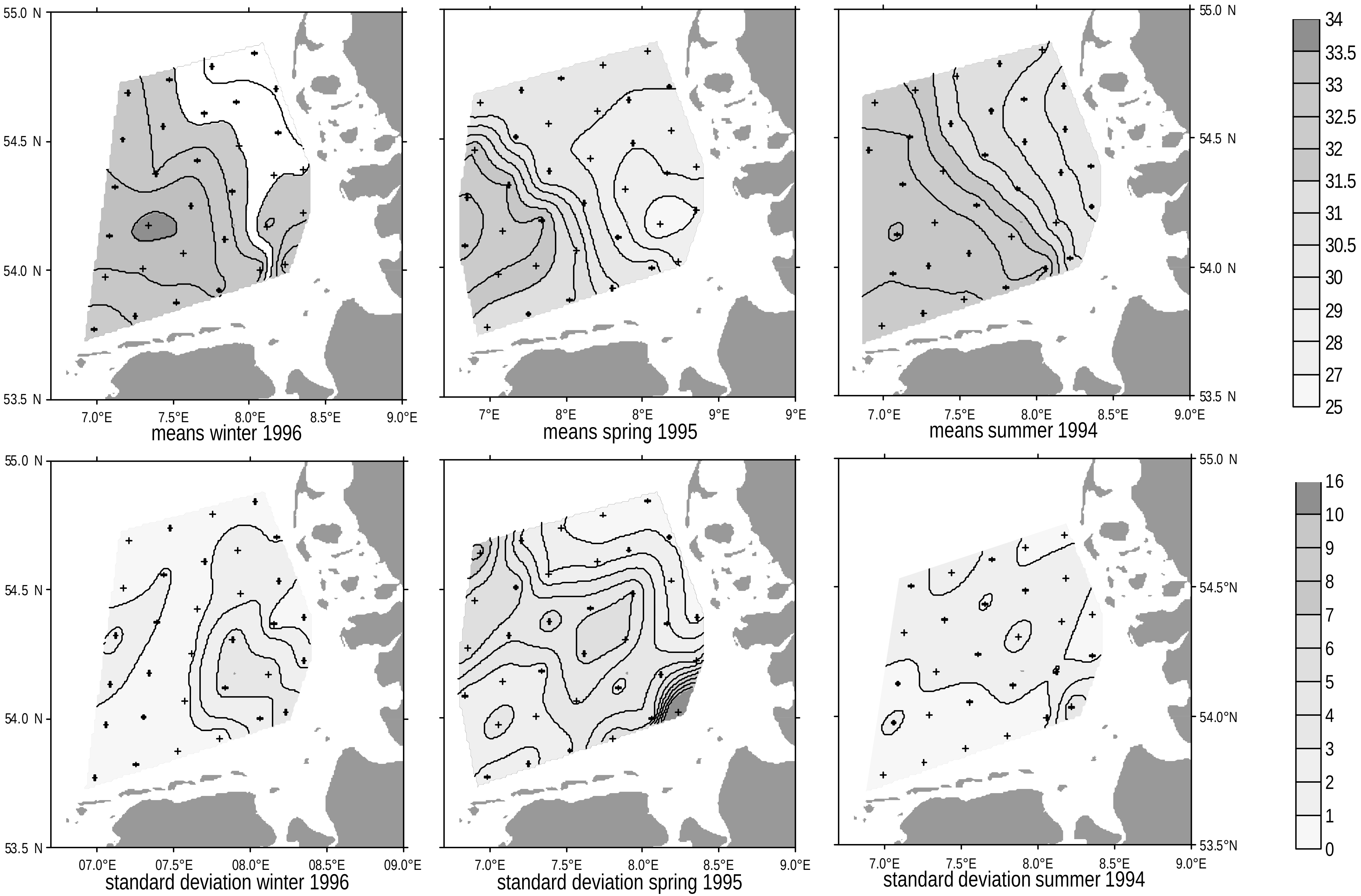
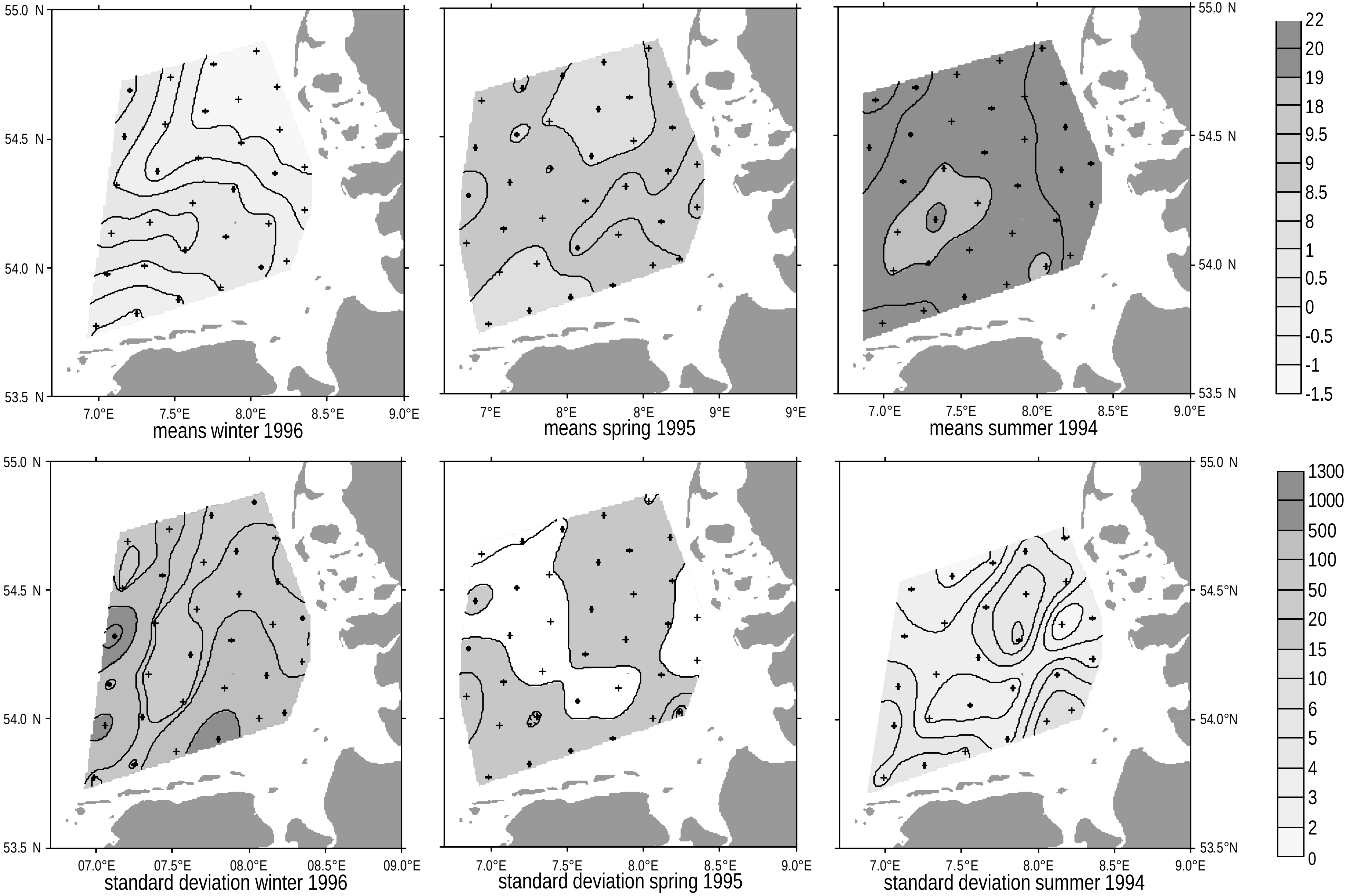


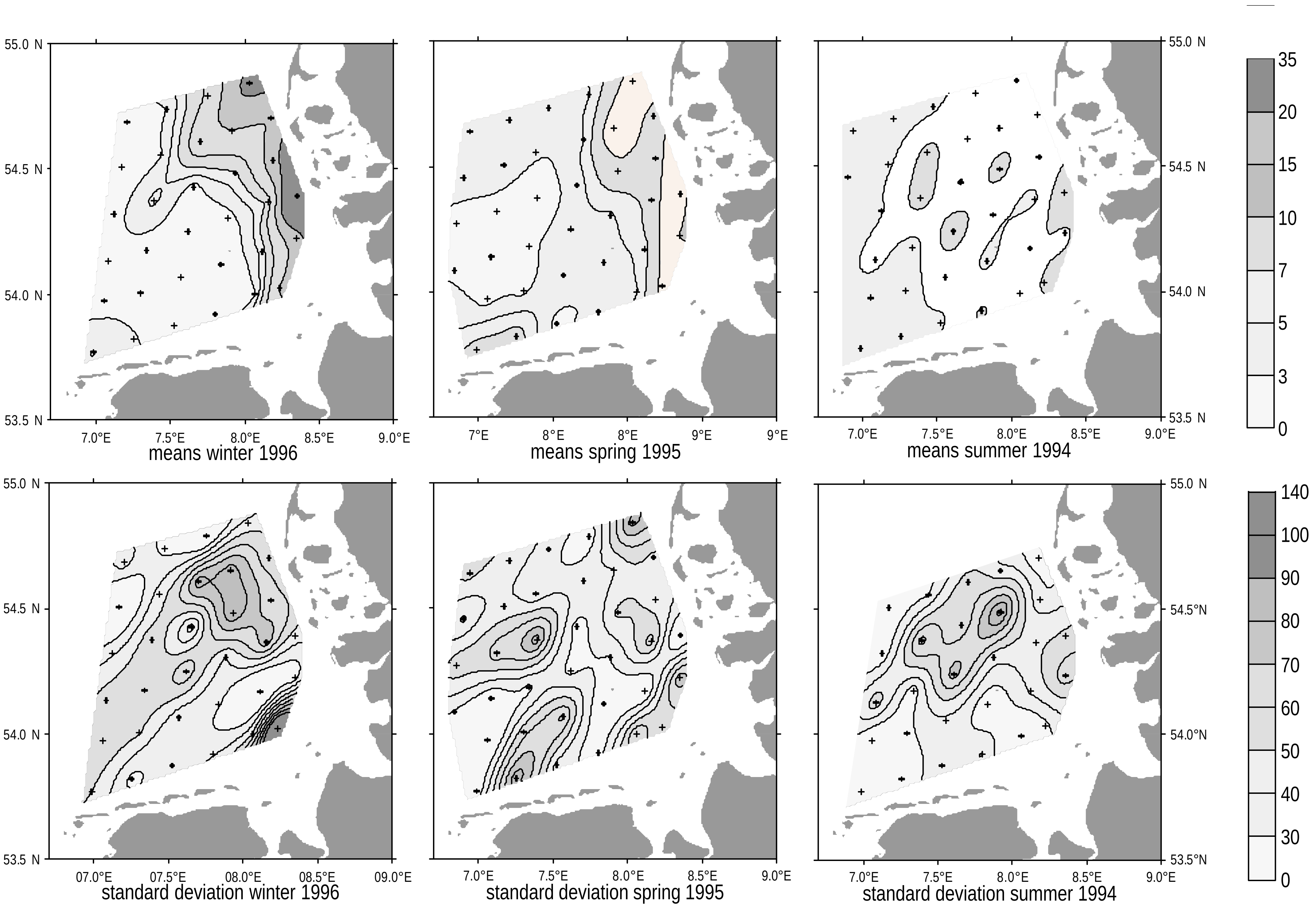
Fig. 2.3.7.1: Short time means and variability [%] (within about 2 weeks) of surface salinity in the German Bight during three selected seasons (see Tab. 2.3.7.1)



means winter 1996 — means spring 1995 — means summer 1994

standard deviation winter 1996 — standard deviation spring 1995 — standard deviation summer 1994

Fig. 2.3.7.2: Short time means [°C] and variability [%] of surface temperature in the German Bight during three selected seasons



means winter 1996

means spring 1995

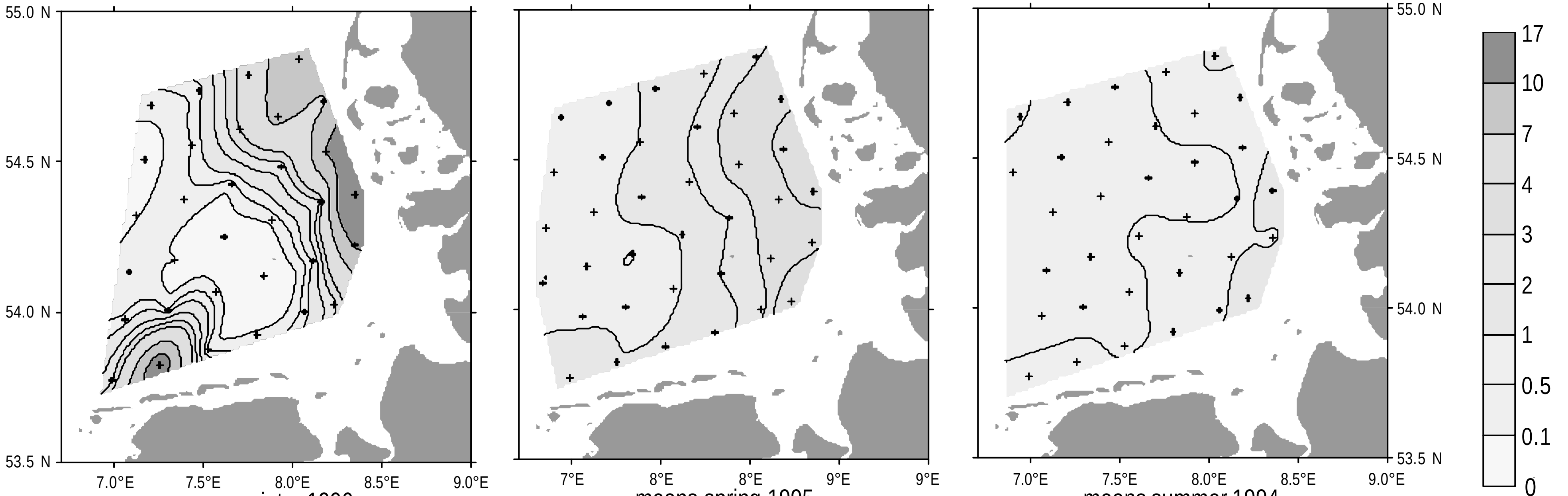
means summer 1994

standard deviation winter 1996

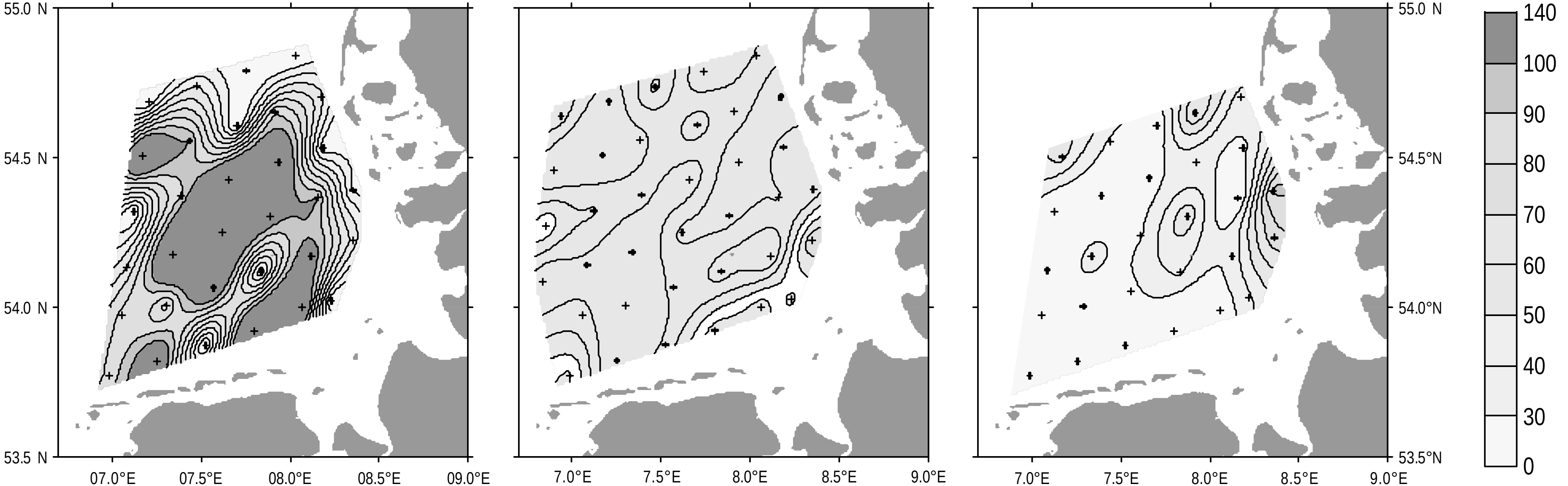
standard deviation spring 1995

standard deviation summer 1994

Fig. 2.3.7.3: Short time means [mg/L] and variability [%] of seston at the surface of the German Bight during three selected seasons



means winter 1996 means spring 1995 means summer 1994



standard deviation winter 1996 standard deviation spring 1995 standard deviation summer 1994

Fig. 2.3.7.4: Short time means [NTU] and variability [%] of turbidity at the surface of the German Bight during three selected seasons

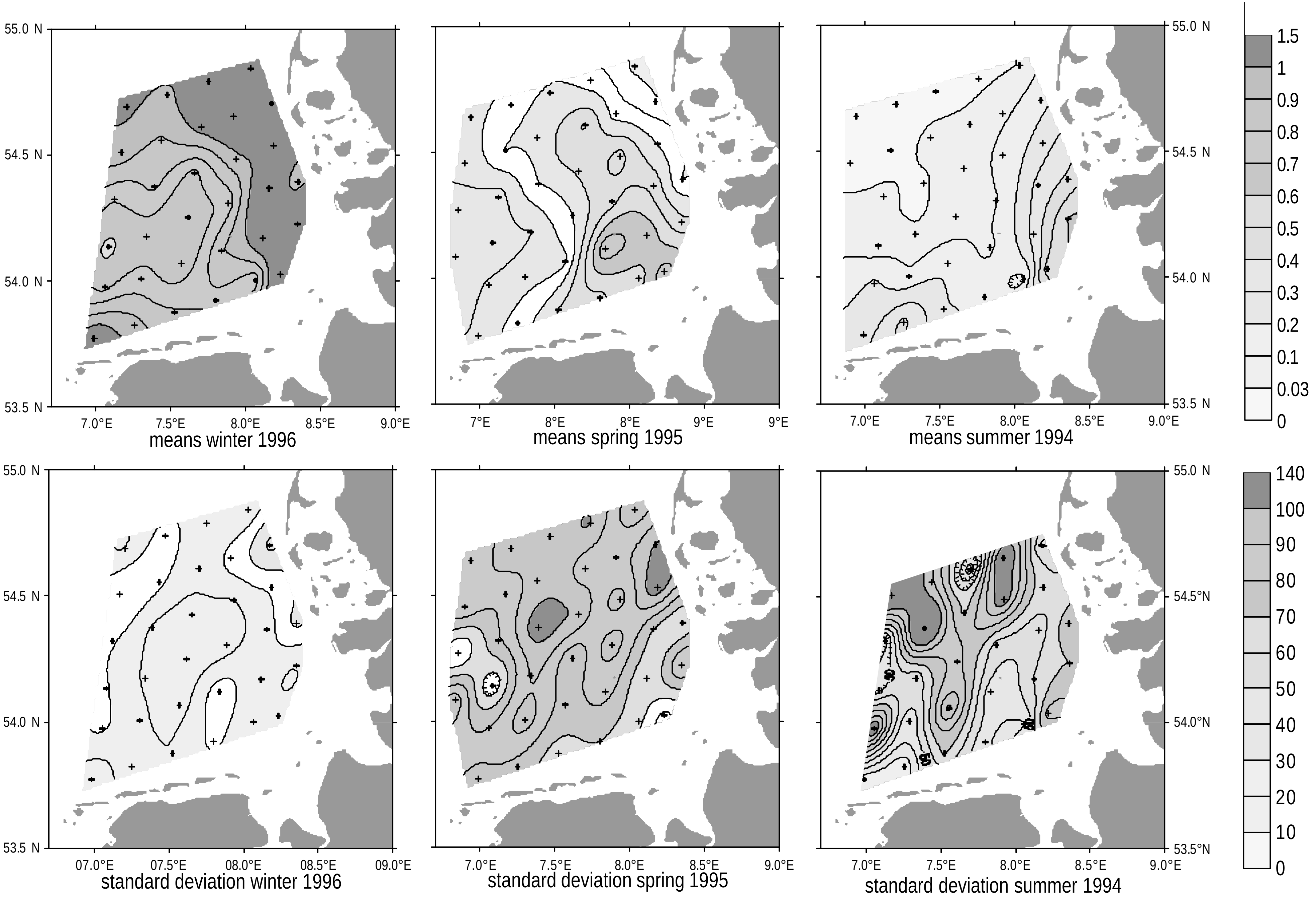
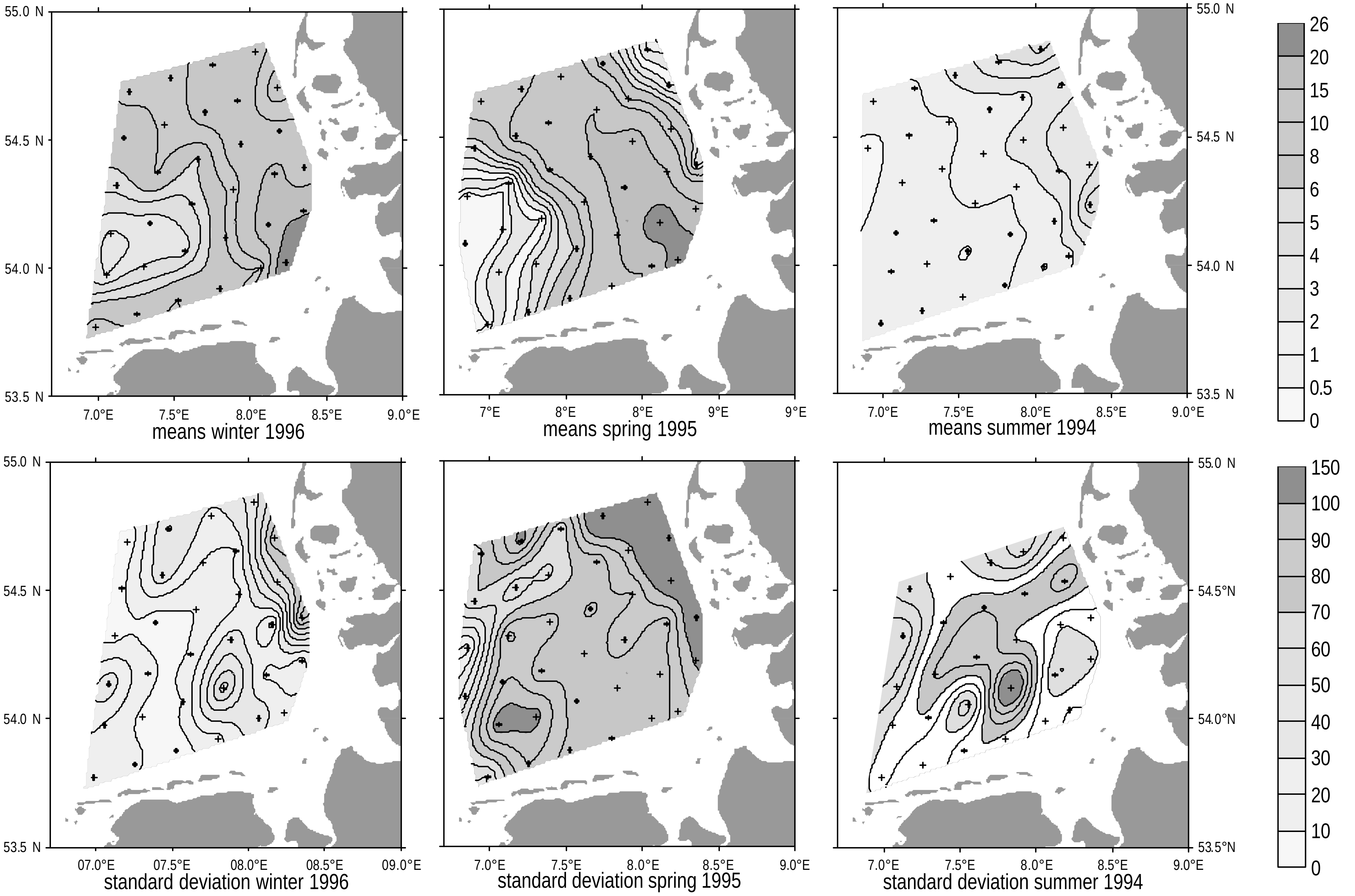


Fig. 2.3.7.5: Short time means [µM] and variability [%] of phosphate at the surface of the German Bight during three selected seasons



means winter 1996

means spring 1995

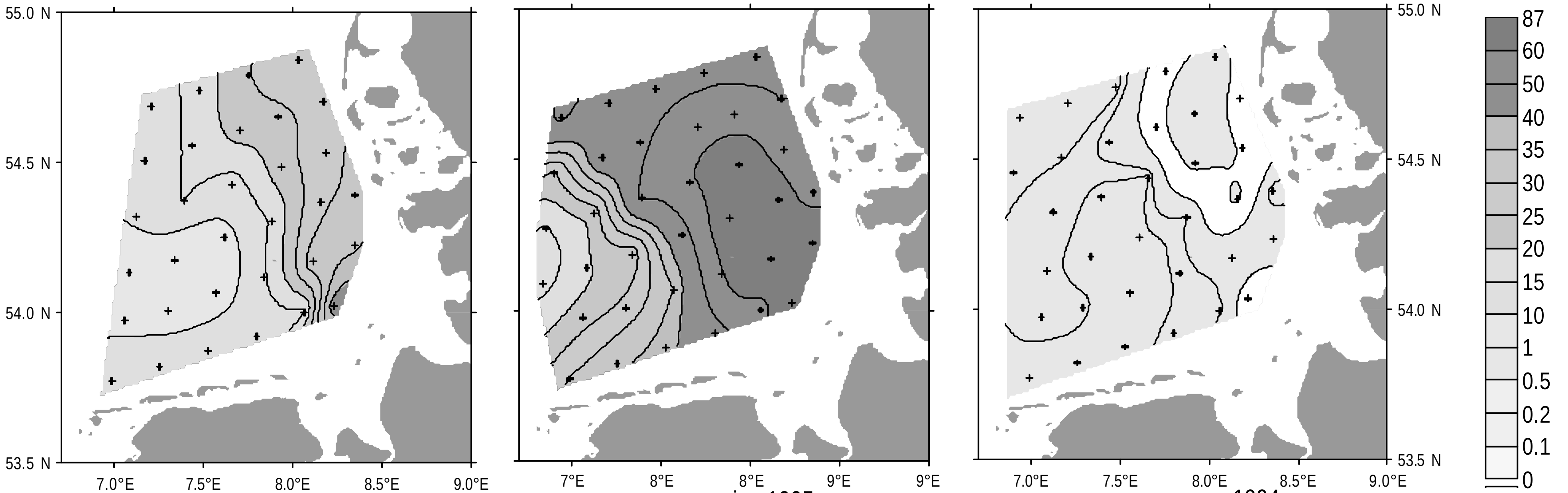
means summer 1994

standard deviation winter 1996

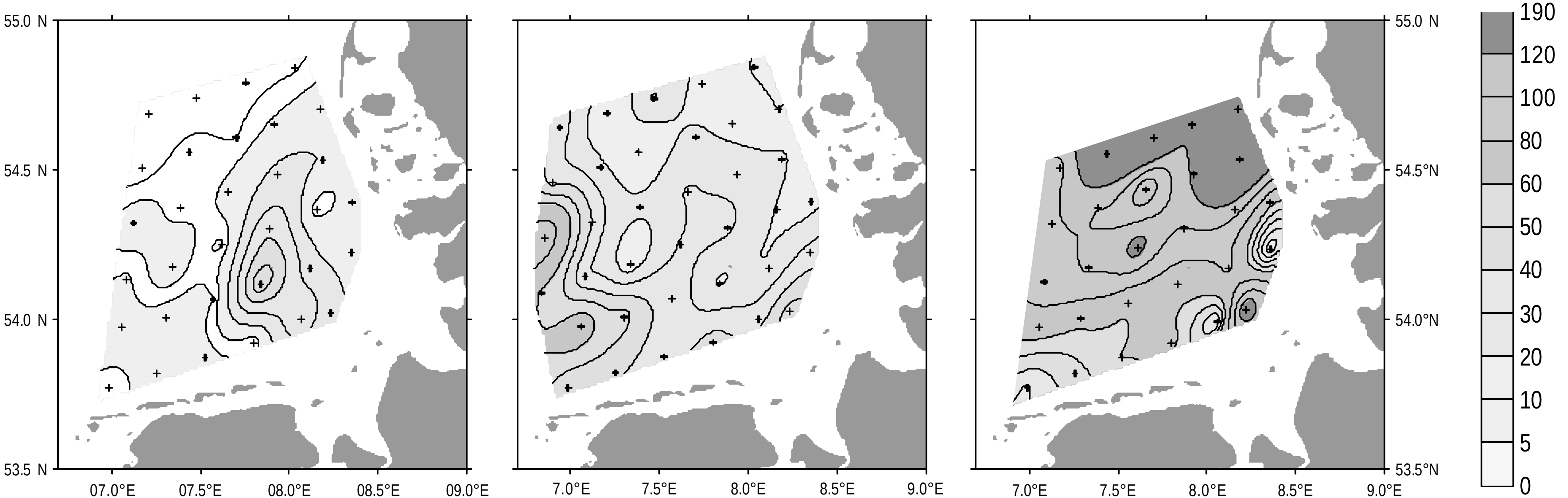
standard deviation spring 1995

standard deviation summer 1994

Fig. 2.3.7.6: Short time means [µM] and variability [%] of silicate at the surface of the German Bight during three selected seasons

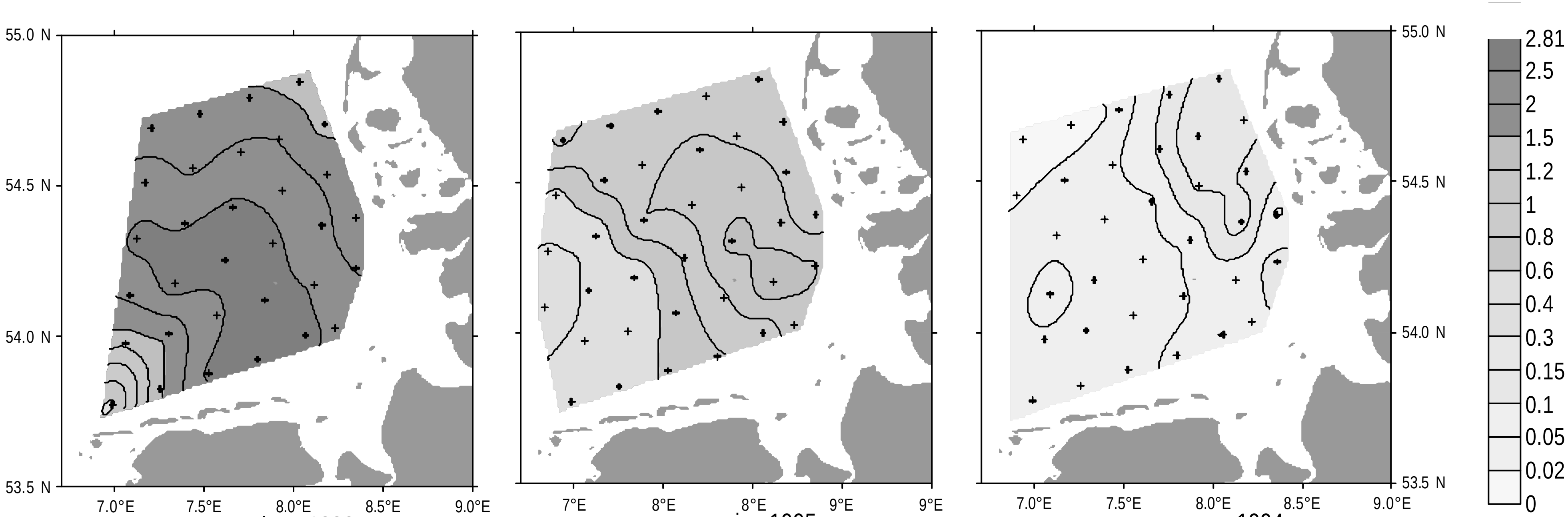


means winter 1996 means spring 1995 means summer 1994

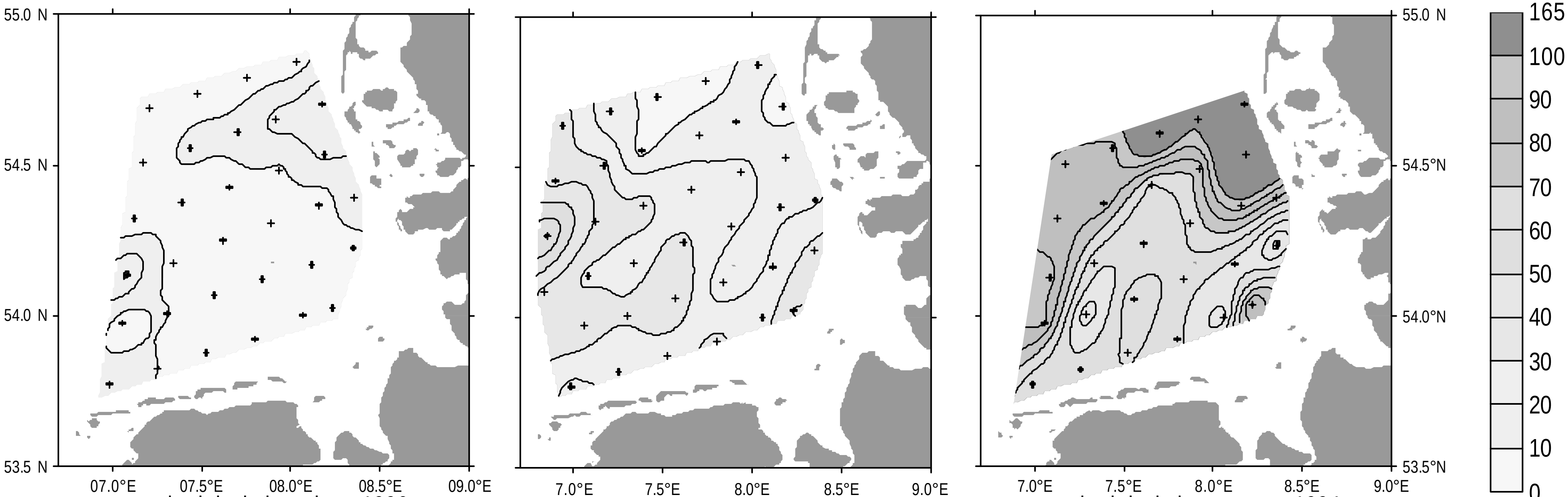


standard deviation winter 1996 standard deviation spring 1995 standard deviation summer 1994

Fig. 2.3.7.7: Short time means [µM] and variability [%] of nitrate at the surface of the German Bight during three selected seasons

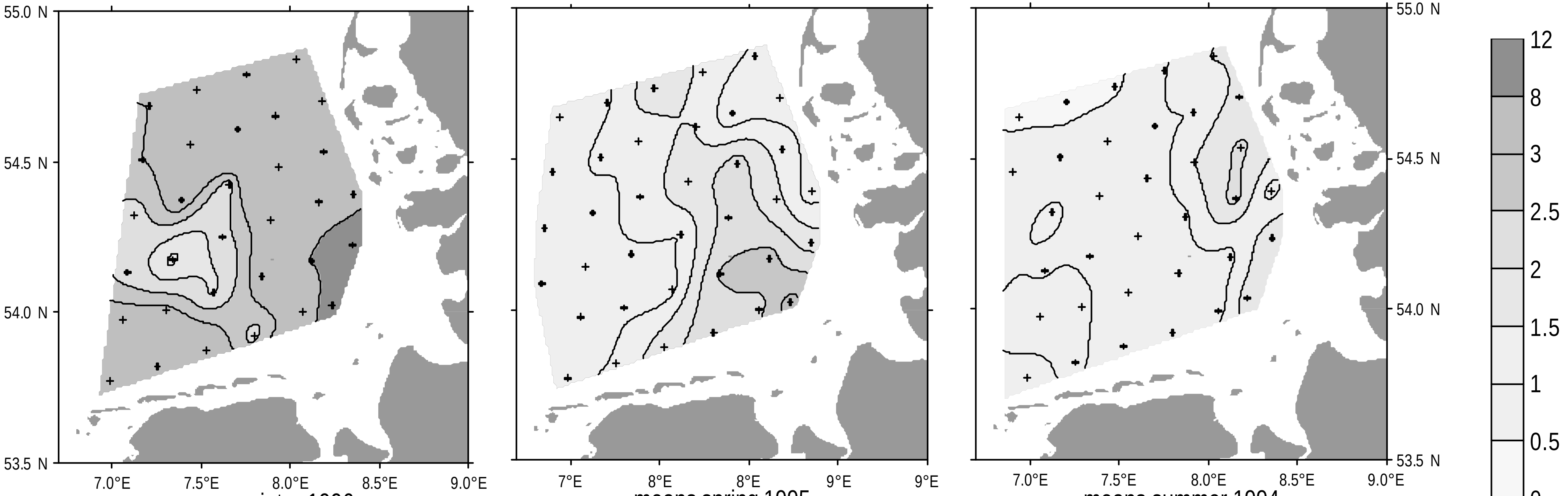


means winter 1996 means spring 1995 means summer 1994

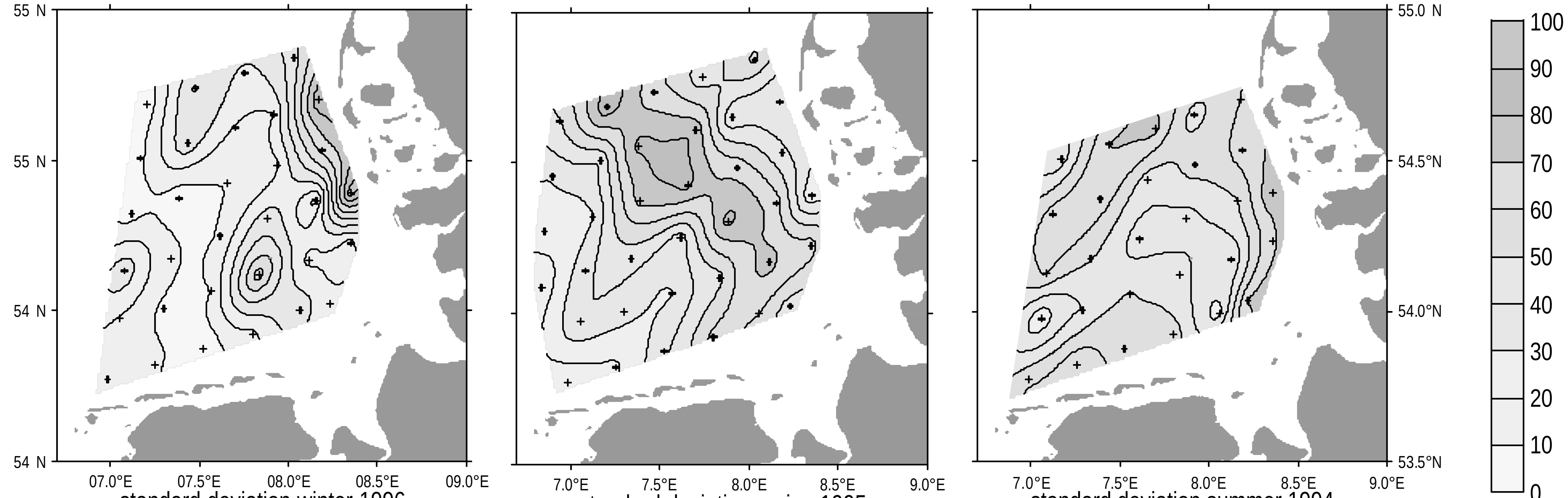


standard deviation winter 1996 standard deviation spring 1995 standard deviation summer 1994

Fig. 2.3.7.8: Short time means [µM] and variability [%] of nitrite at the surface of the German Bight during three selected seasons



means winter 1996 means spring 1995 means summer 1994



standard deviation winter 1996 standard deviation spring 1995 standard deviation summer 1994

Fig. 2.3.7.9: Short time means [µM] and variability [%] of ammonium at the surface of the German Bight during three selected seasons

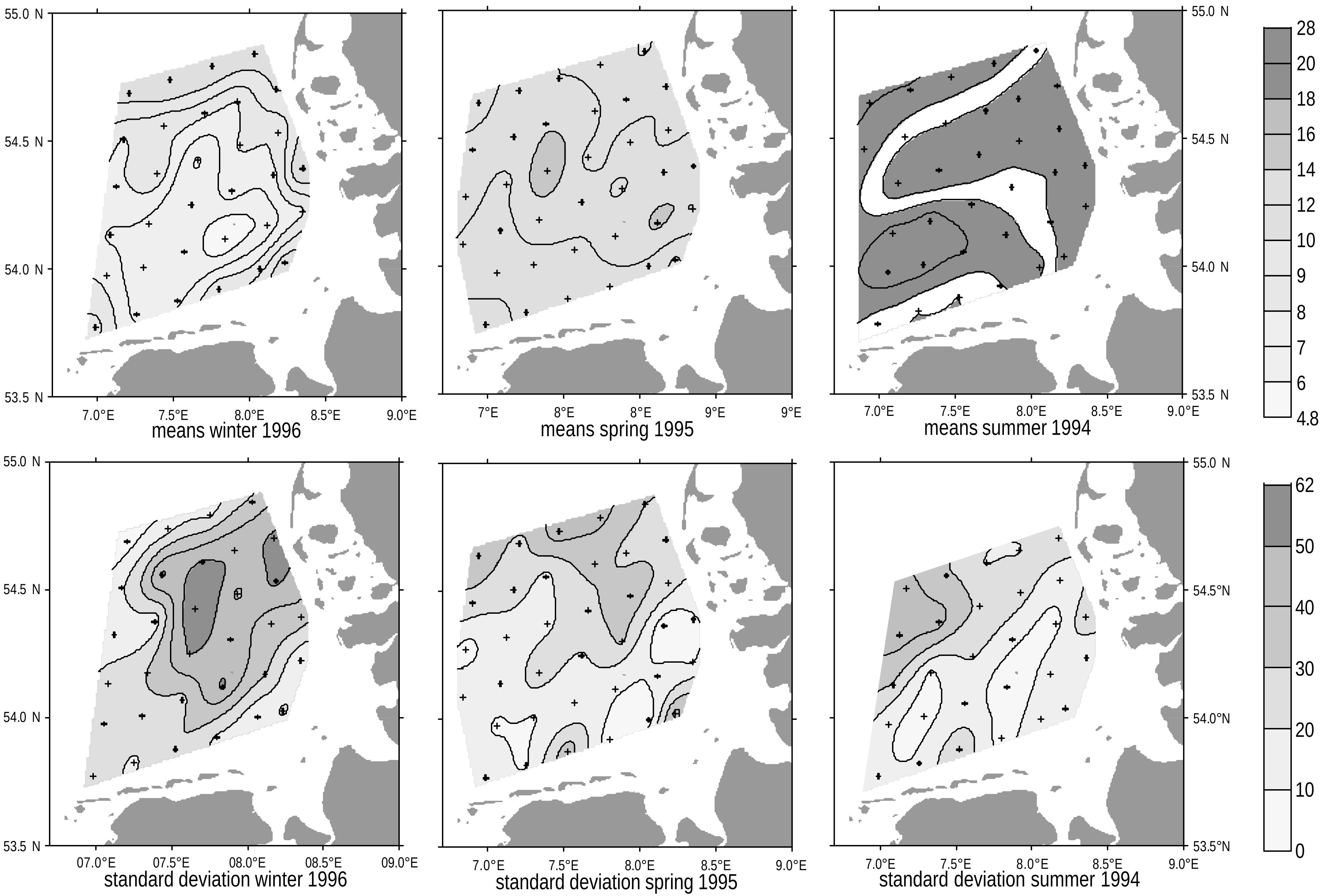
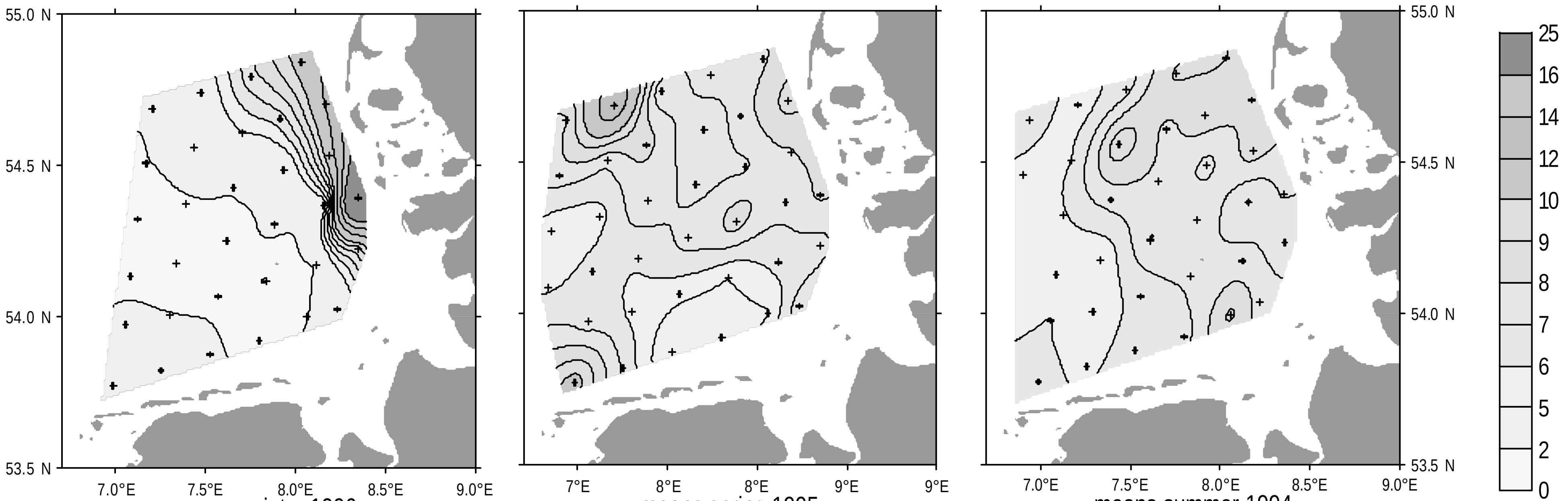


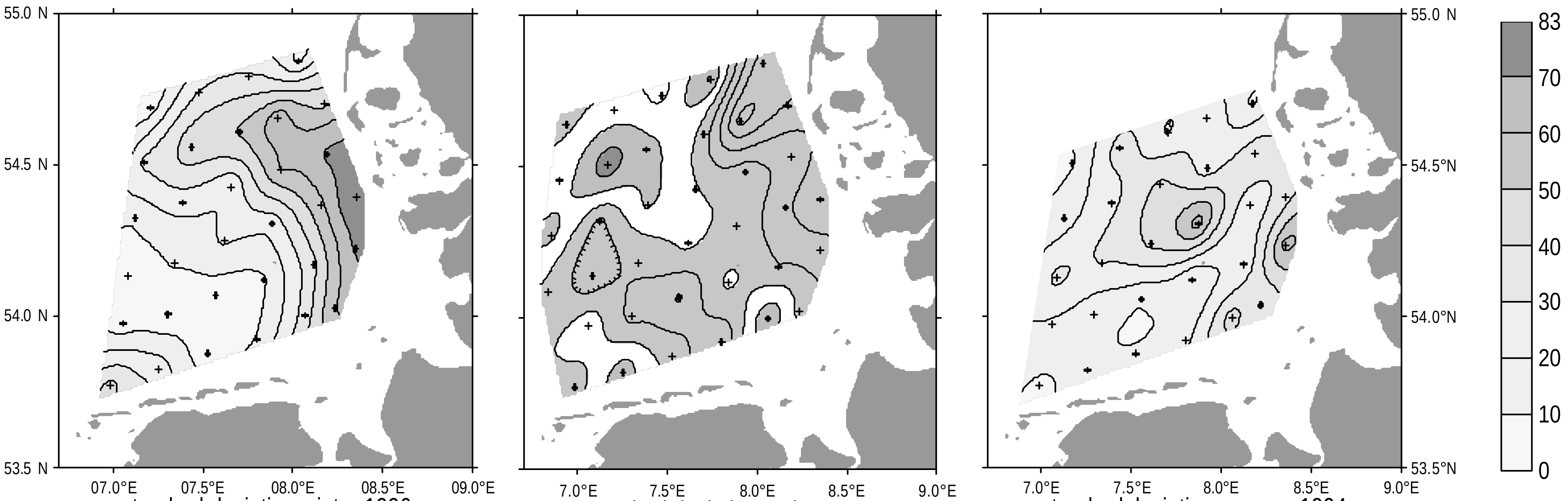
Fig. 2.3.7.10: Short time means [µM] and variability [%] of diss. organic nitrogen at the surface of the German Bight during three selected seasons



means winter 1996

means spring 1995

means summer 1994



standard deviation winter 1996

standard deviation spring 1995

standard deviation summer 1994

Fig. 2.3.7.11: Short time means [µM] and variability [%] of particulate nitrogen at the surface of the German Bight during three selected seasons

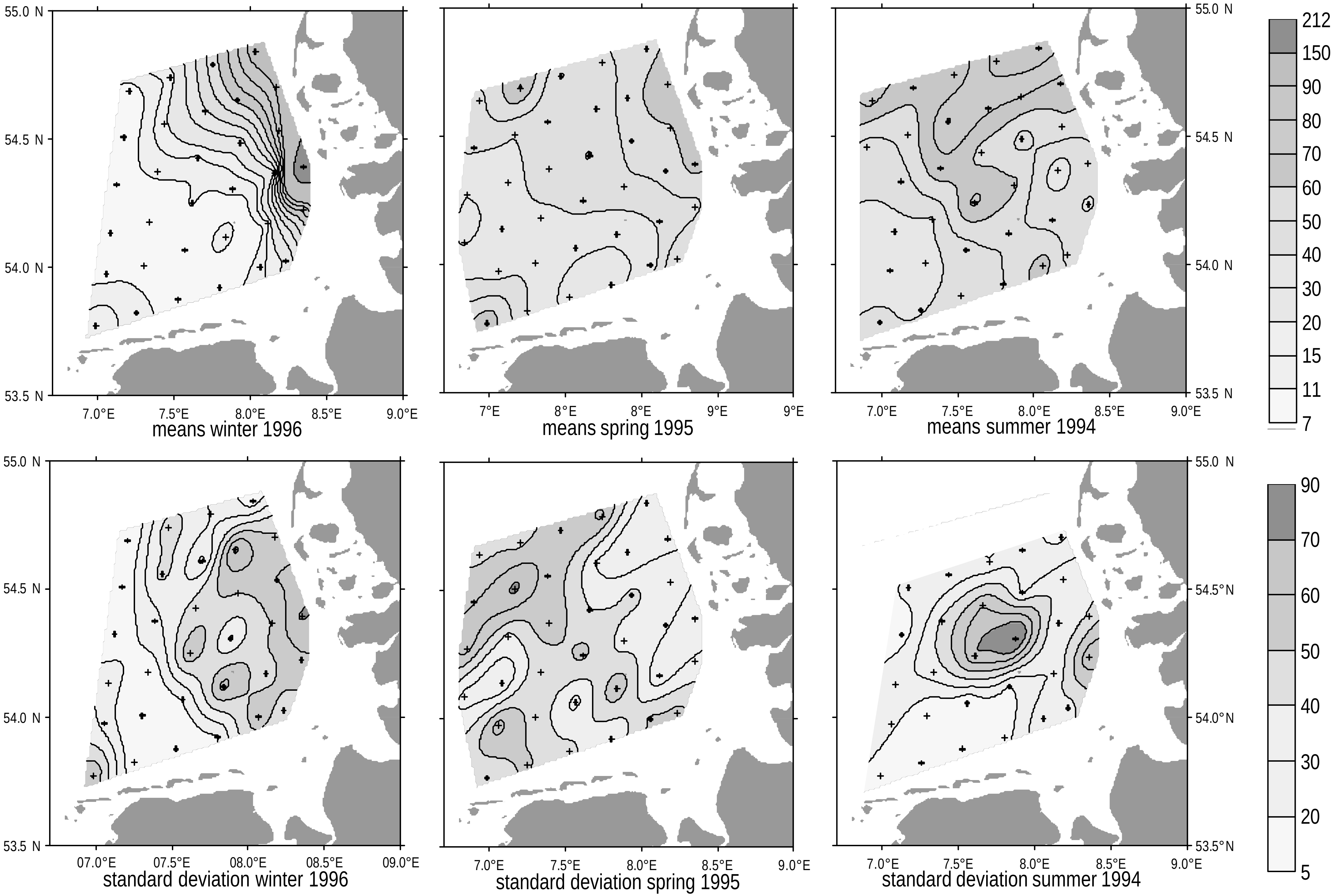
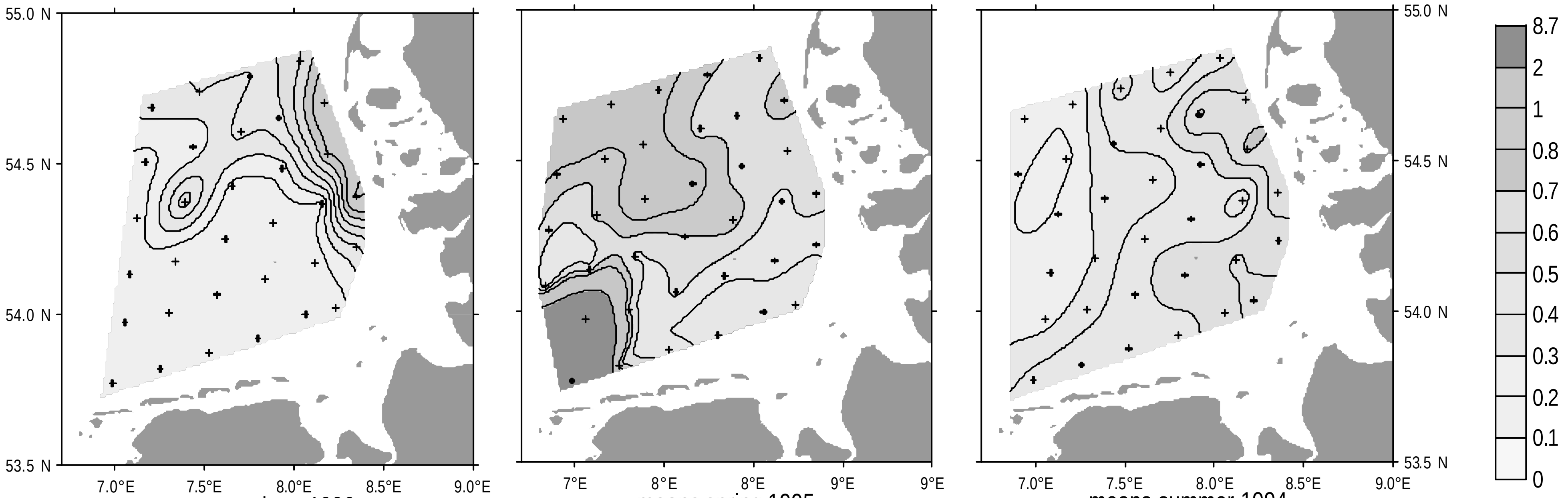
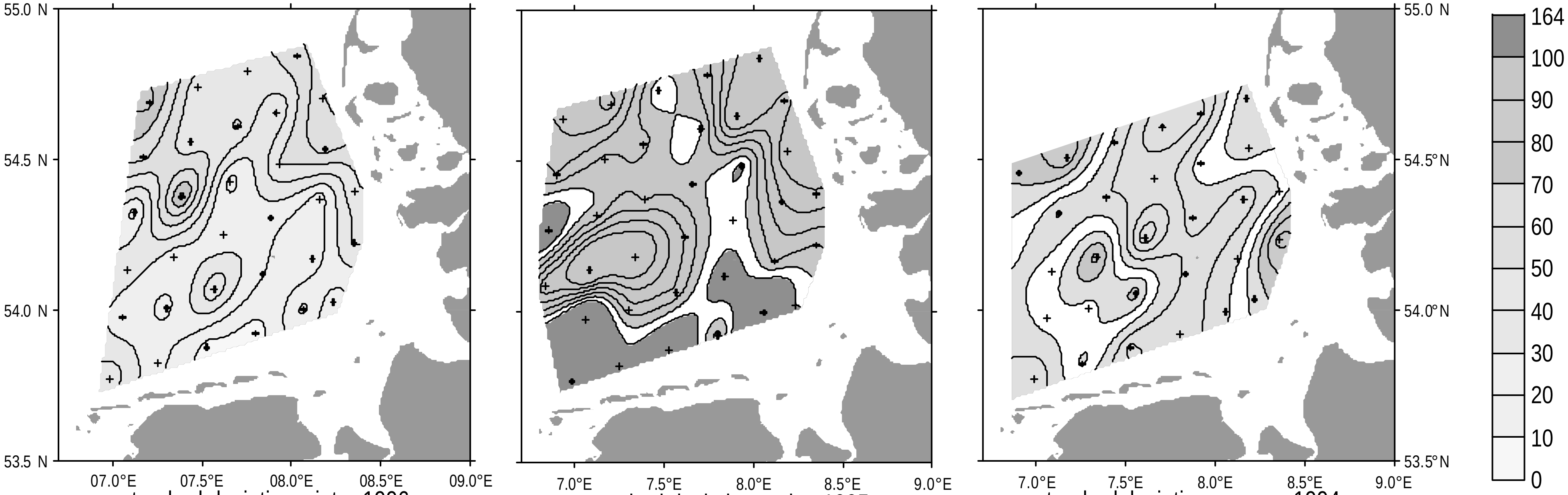


Fig. 2.3.7.12: Short time means [µM] and variability [%] of part. organic carbon at the surface of the German Bight during three selected seasons

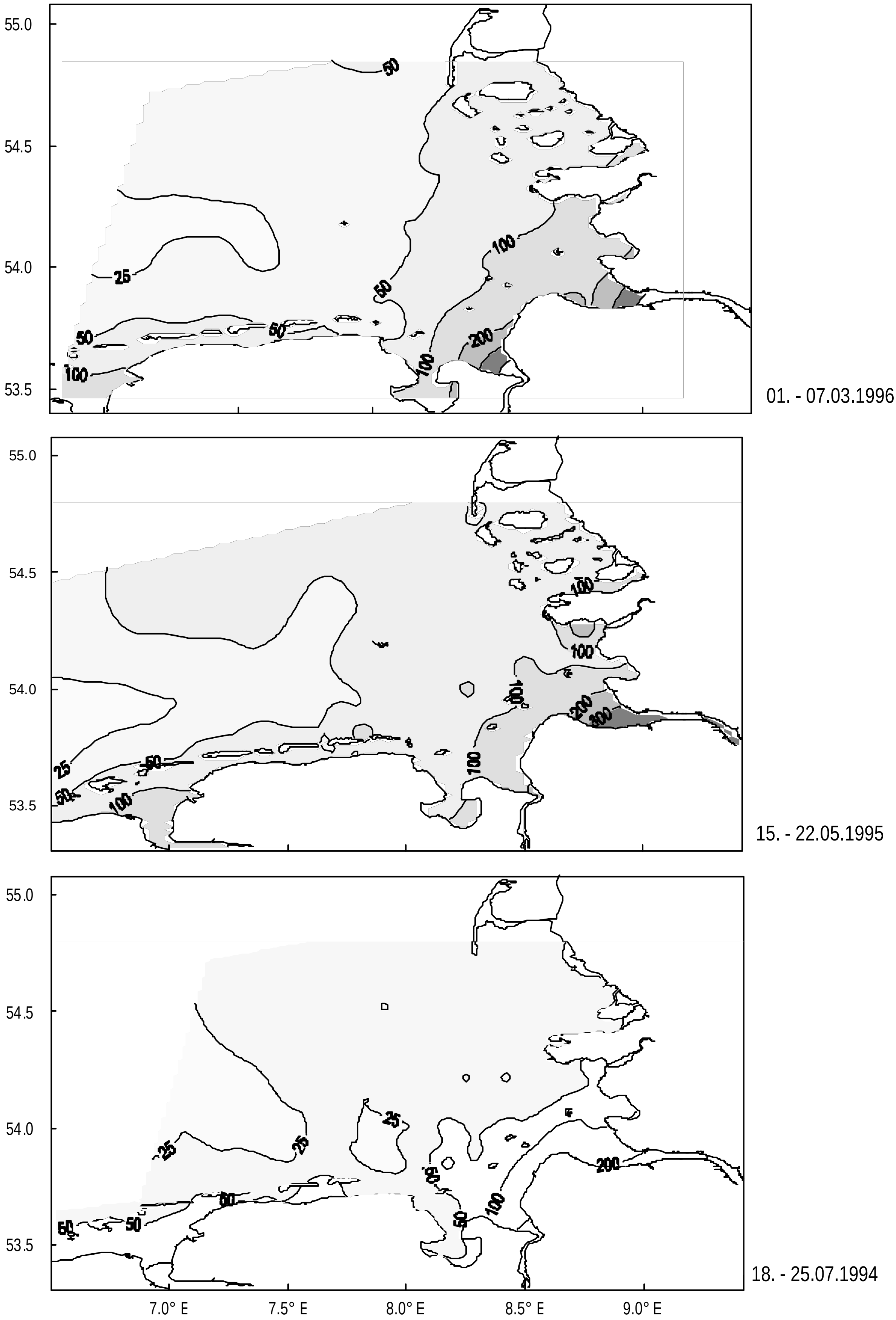


means winter 1996 means spring 1995 means summer 1994



standard deviation winter 1996 standard deviation spring 1995 standard deviation summer 1994

Fig. 2.3.7.13: Short time means [Turner units] and variability [%] of fluorescence at the surface of the German Bight during three selected seasons



01. - 07.03.1996

15. - 22.05.1995

18. - 25.07.1994

Fig. 2.3.8.1: Seasonal gradients of total N (µM) at the surface of the German Bight

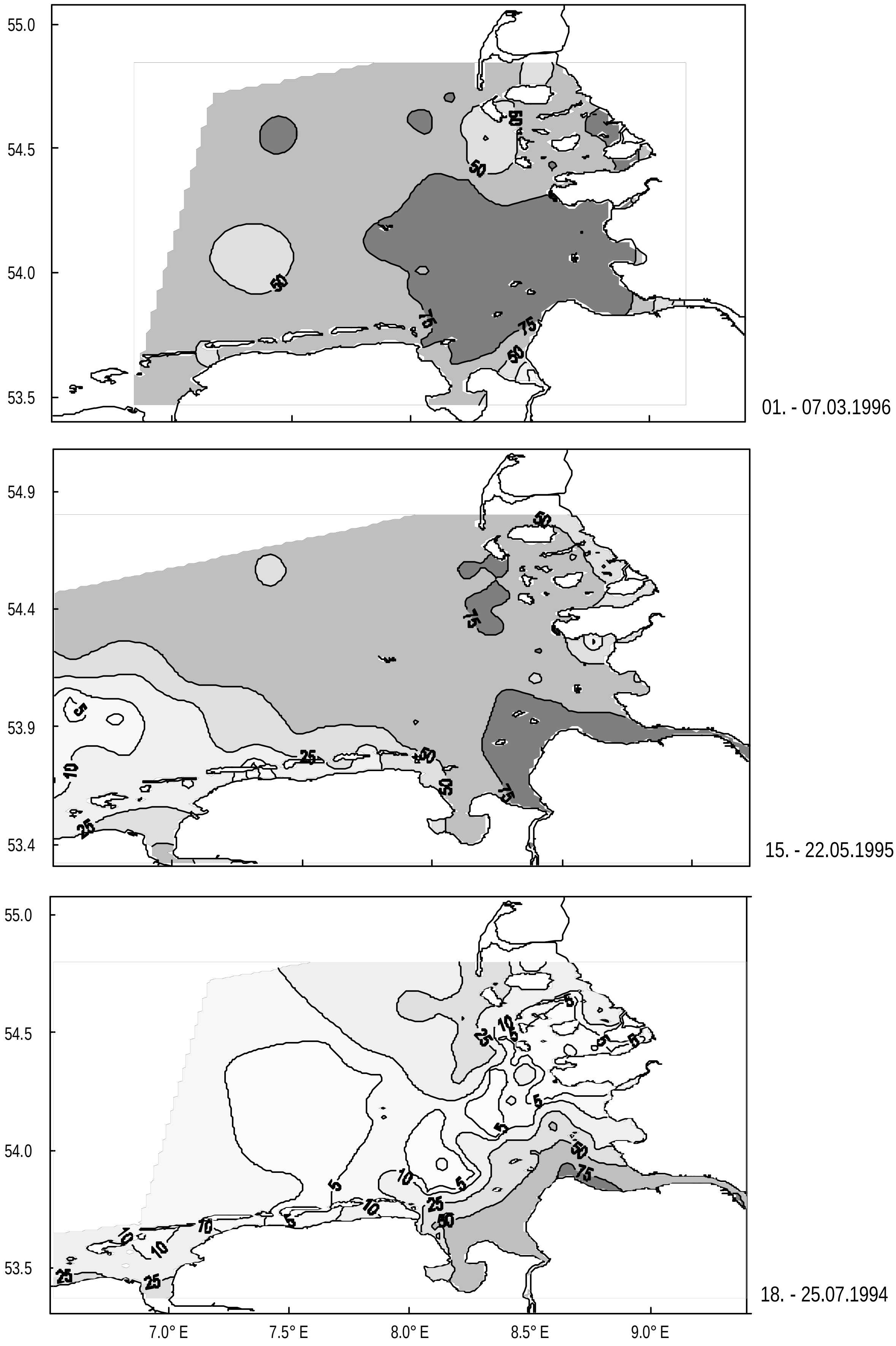


Fig. 2.3.8.2: Seasonal phase distribution of nitrogen: DIN (% of TN) at the surface of the German Bight

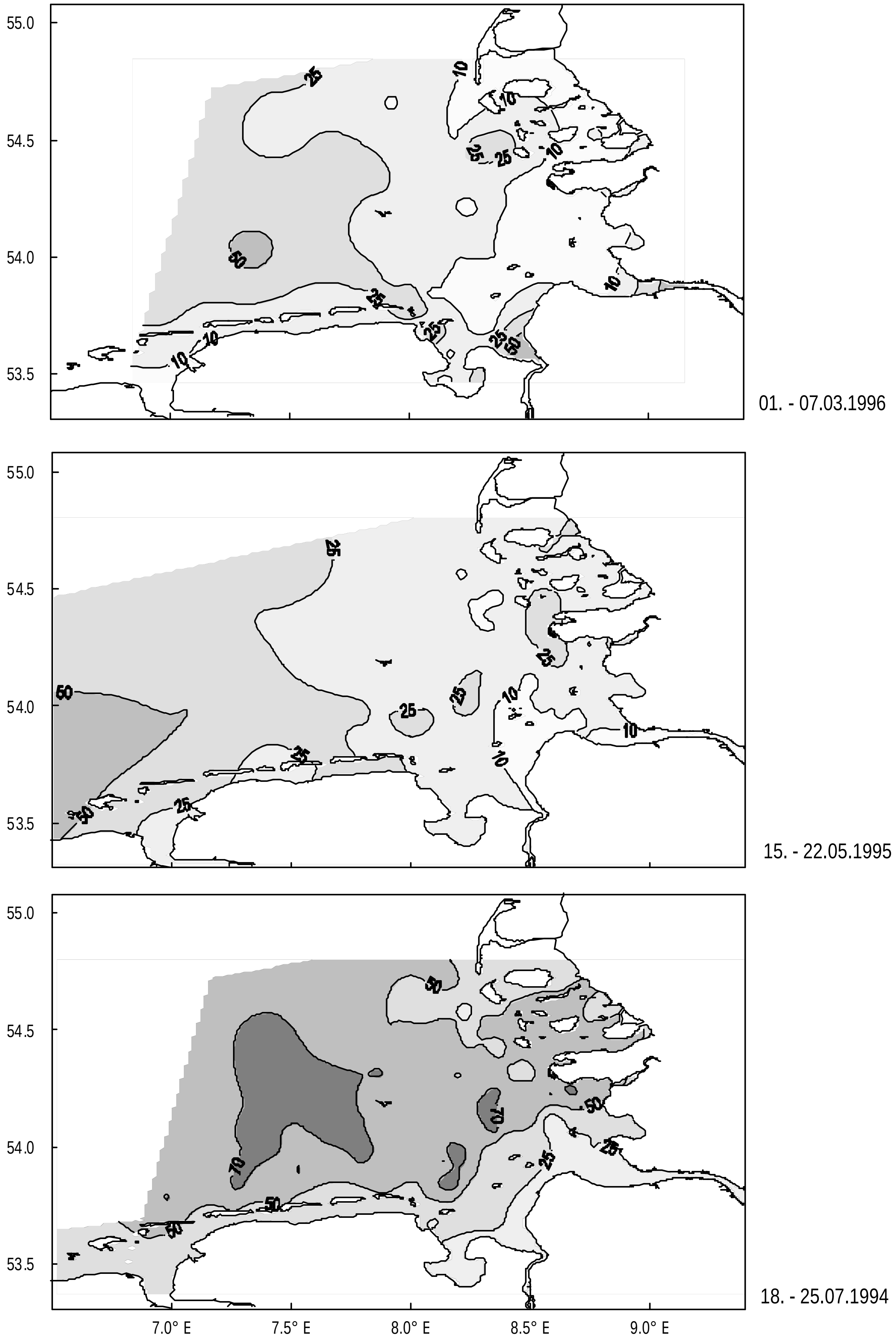


Fig. 2.3.8.3: Seasonal phase distribution of nitrogen: DON (% of TN) at the surface of the German Bight

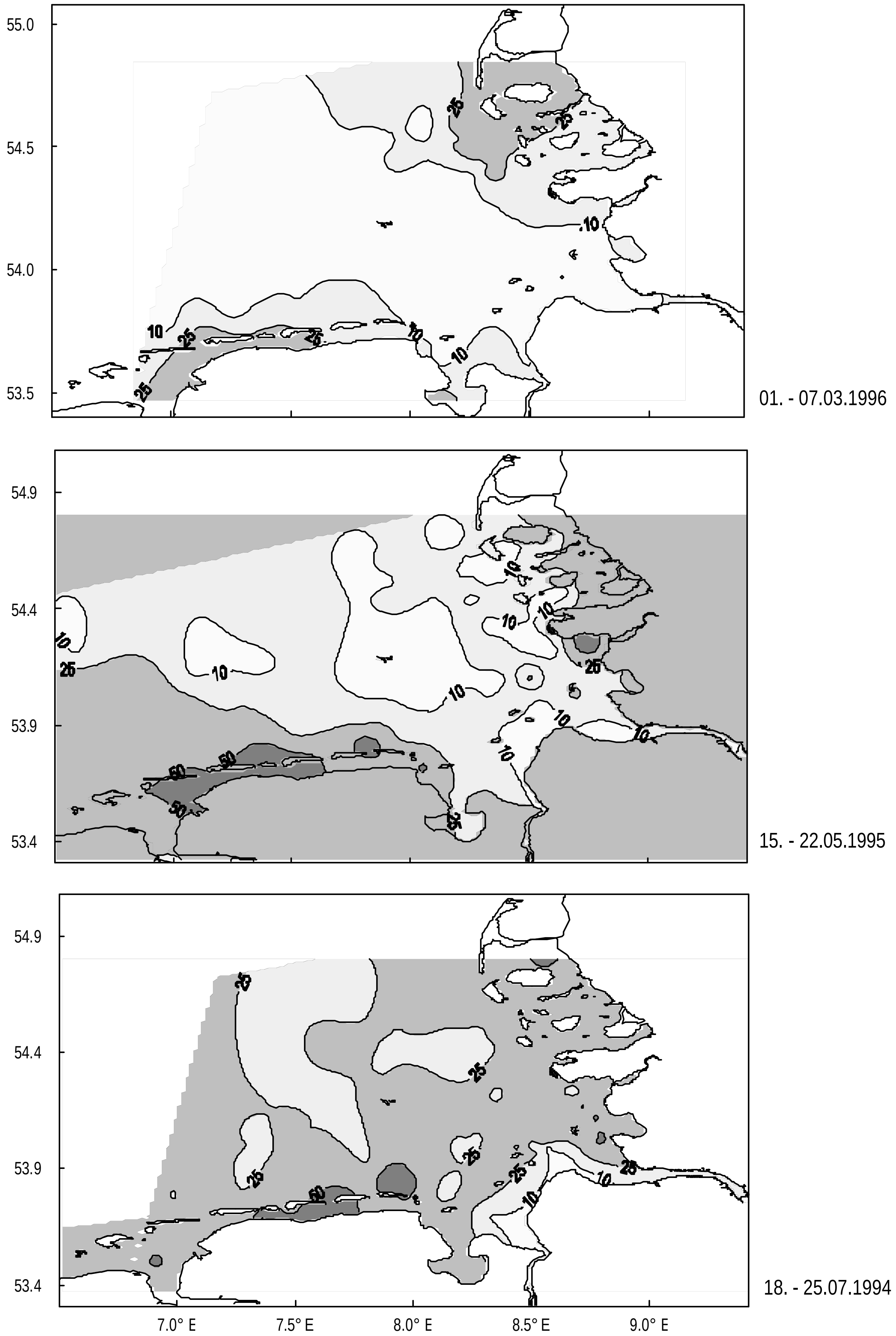


Fig. 2.3.8.4: Seasonal phase distribution of nitrogen: PN (% of TN) at the surface of the German Bight

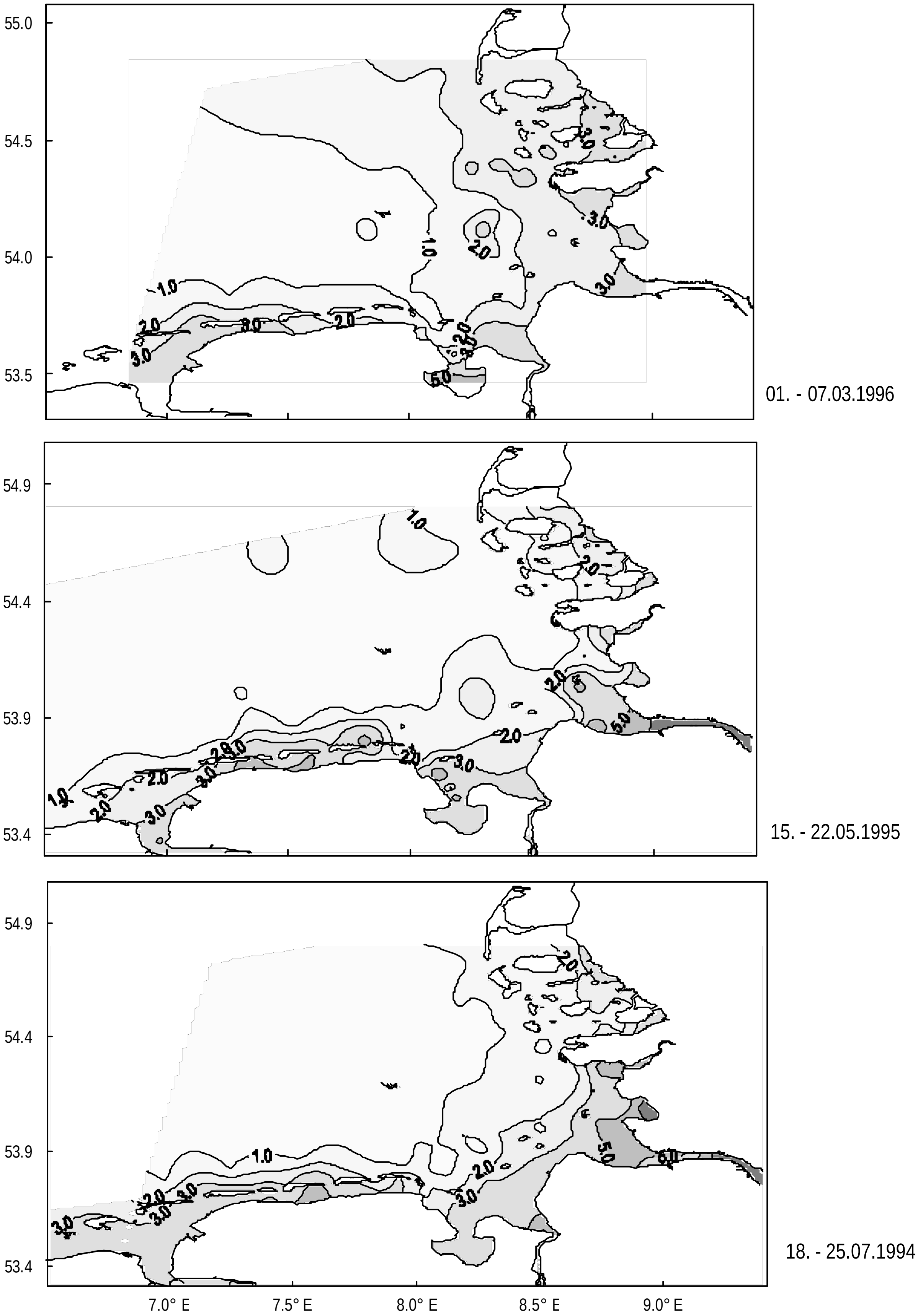


Fig. 2.3.8.5: Seasonal gradients of total P (µM) at the surface of the German Bight

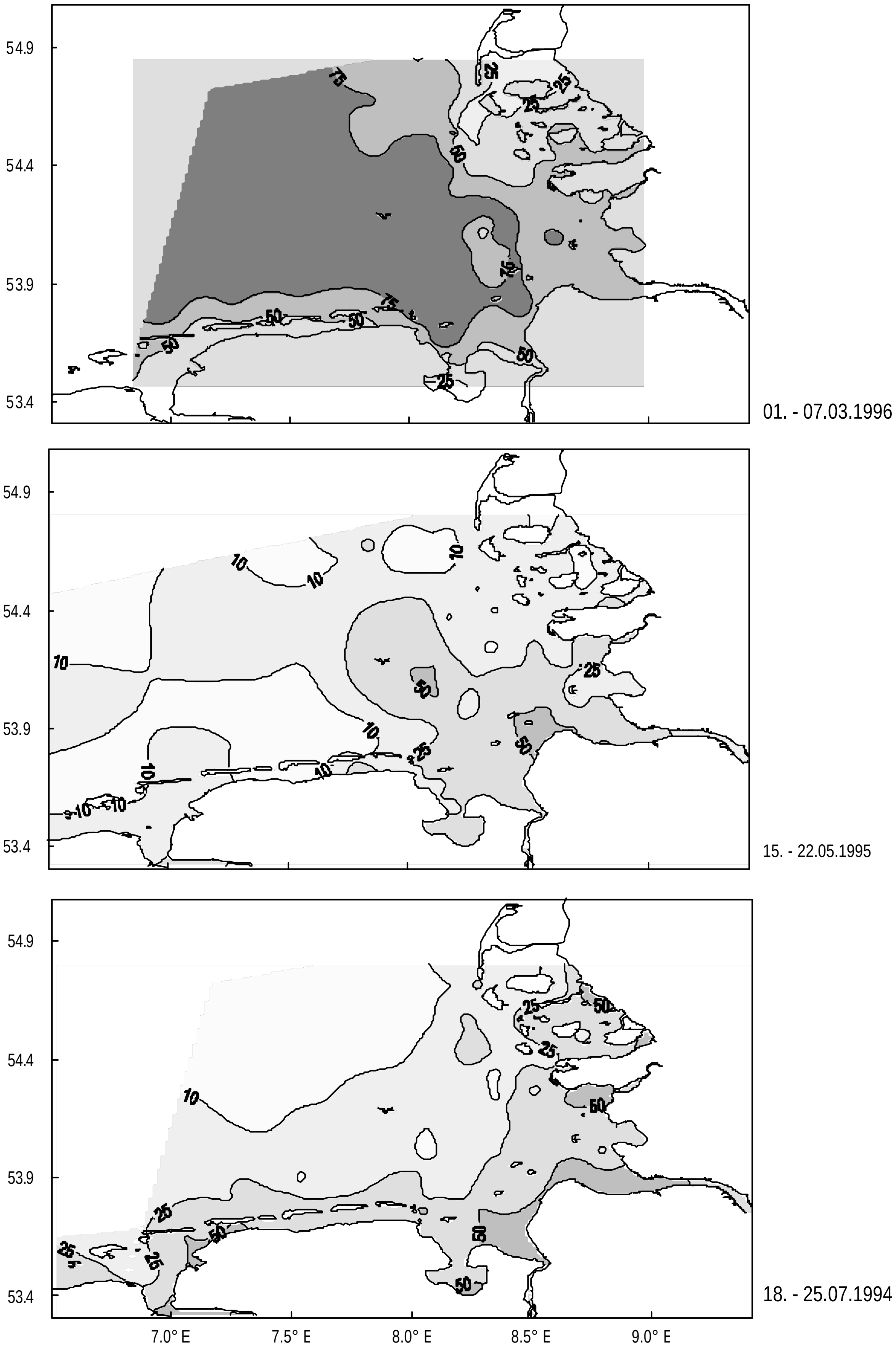


Fig. 2.3.8.6: Seasonal phase distribution of phosphorus: DIP (% of TP) at the surface of the German Bight

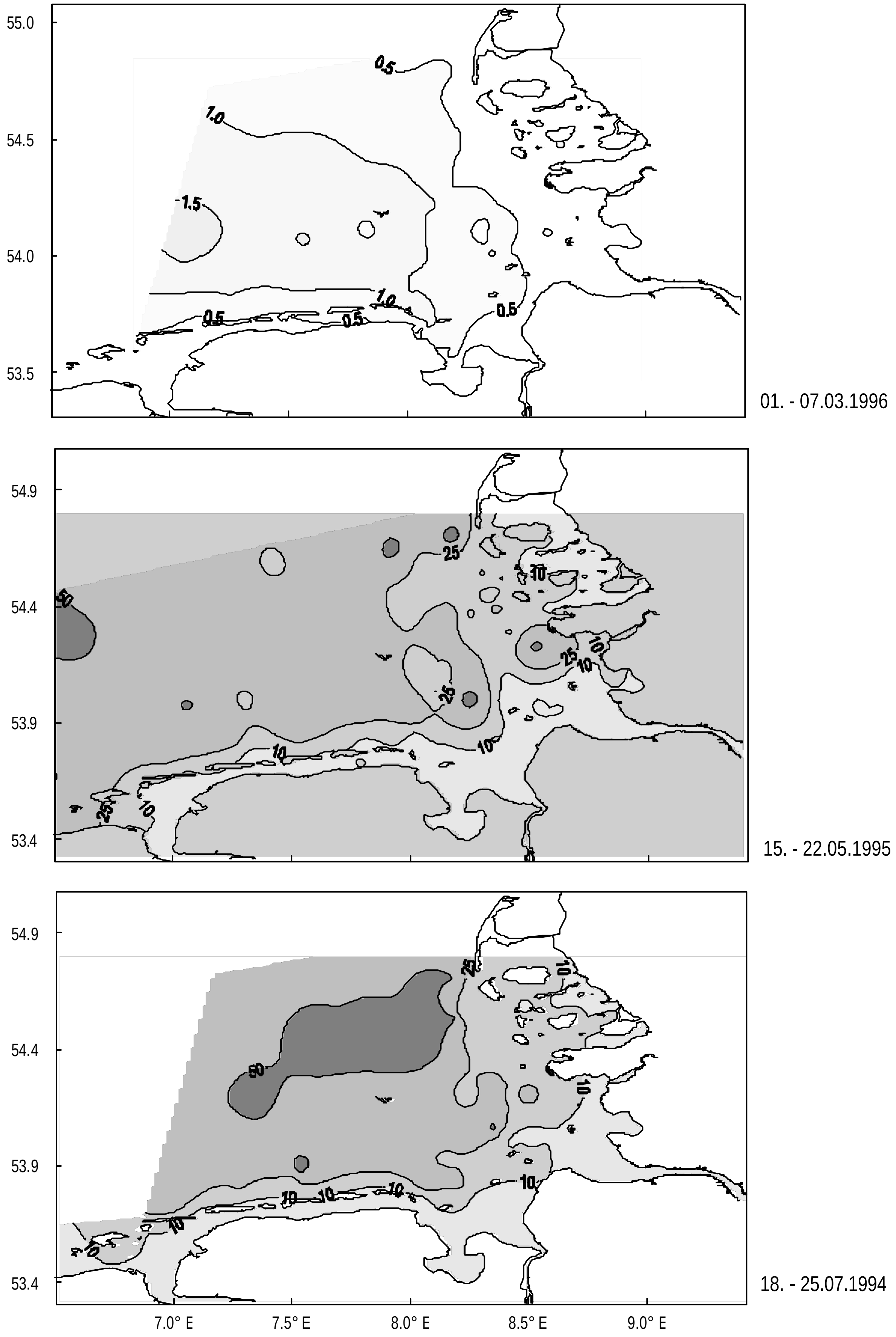
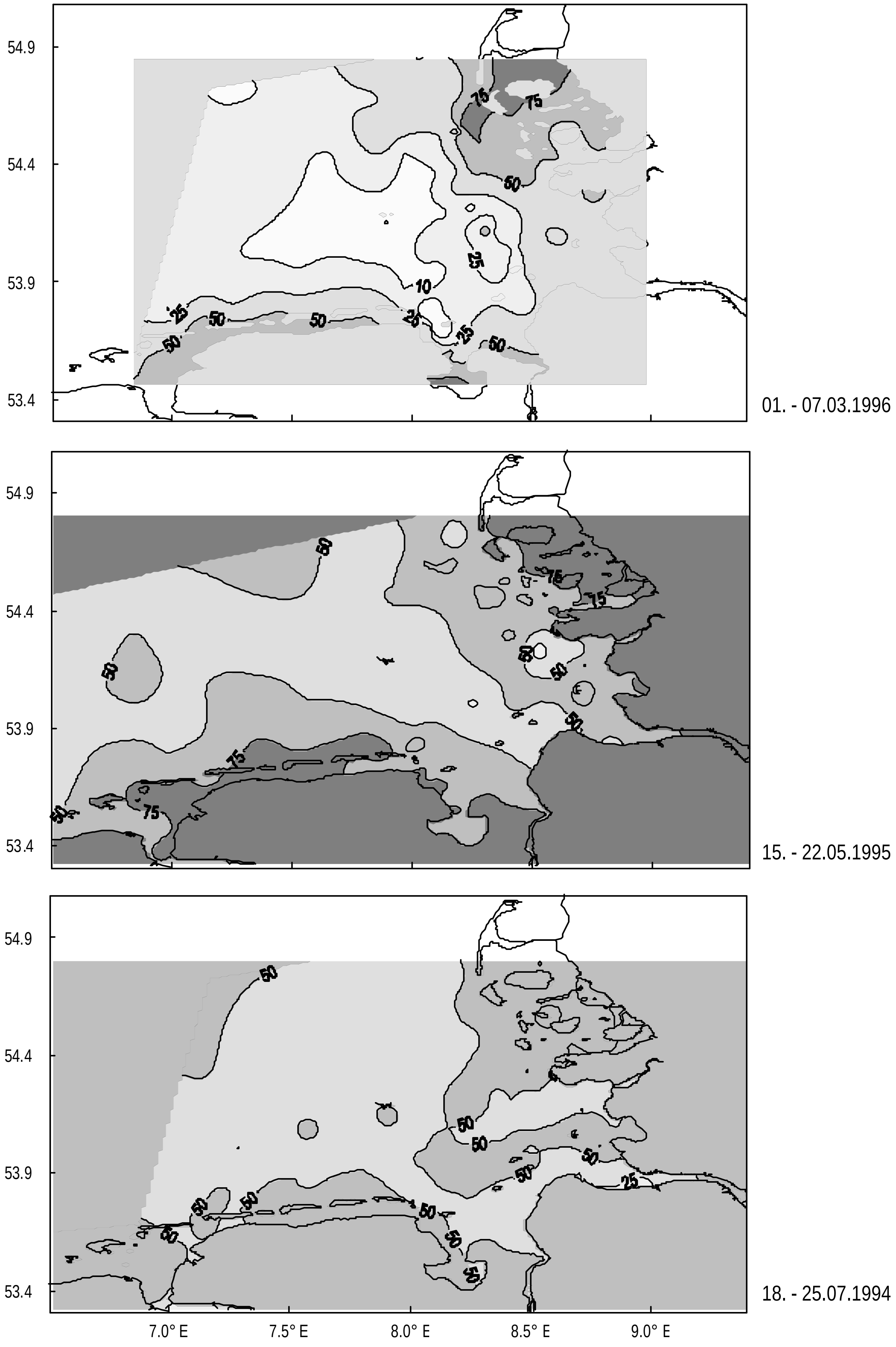


Fig. 2.3.8.7: Seasonal phase distribution of phosphorus: DOP (% of TP) at the surface of the German Bight



01. - 07.03.1996

15. - 22.05.1995

18. - 25.07.1994

Fig. 2.3.8.8: Seasonal phase distribution of phosphorus: PP (% of TP) at the surface of the German Bight

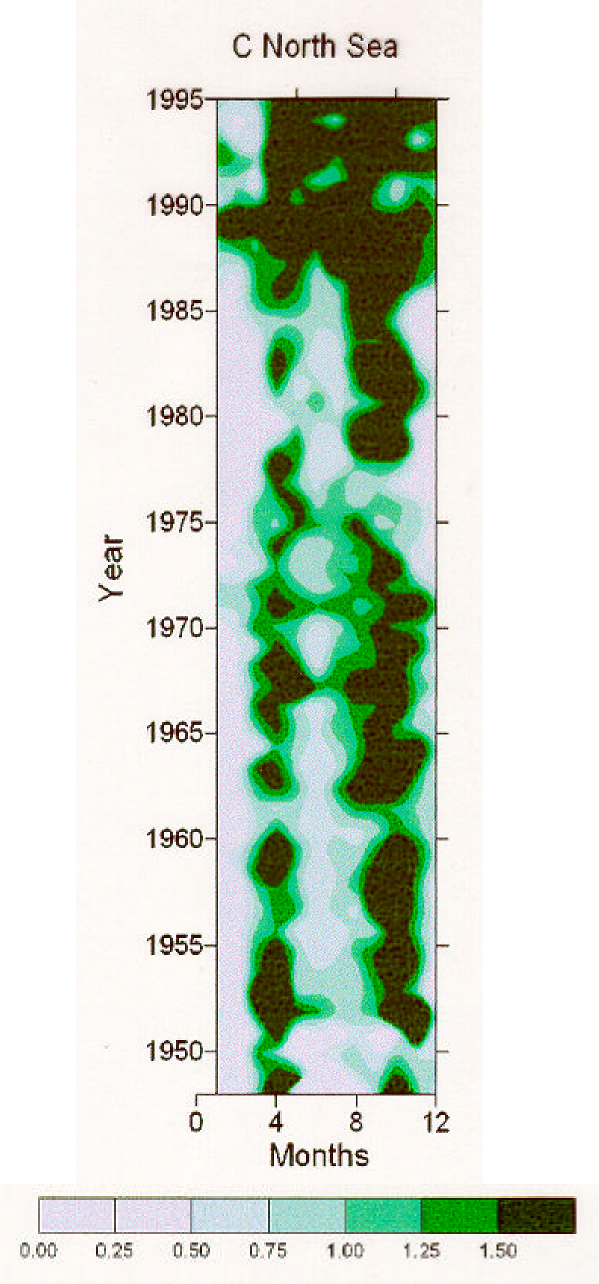


Fig. 2.4.1.1: Aggregated 'Greenness' - data from different CPR - surveys in the North Sea (Reid et al. 1998).

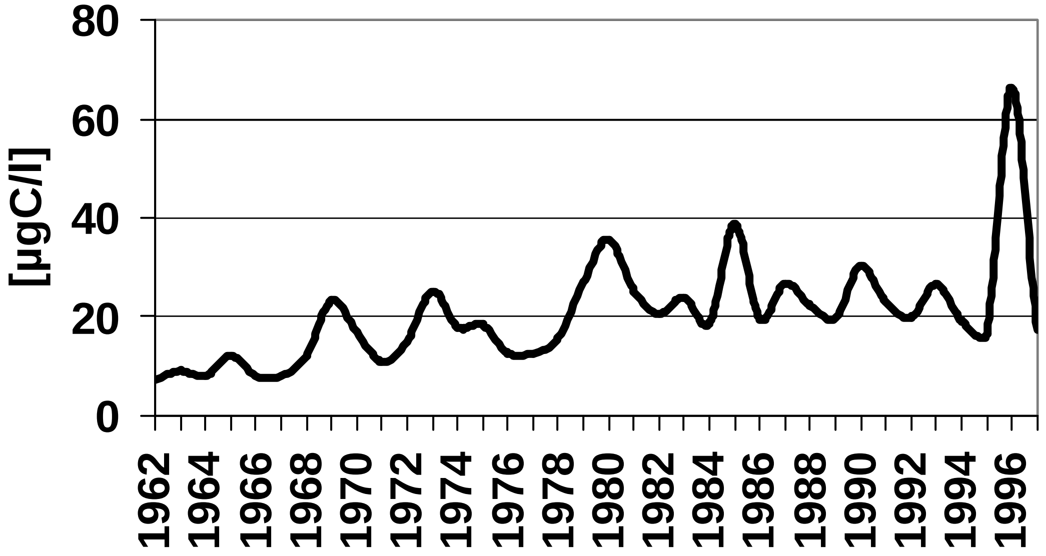


Fig. 2.4.2.1: Annual median phytoplankton carbon (PPC) concentration at Helgoland Roads (modified after Hickel 1998).

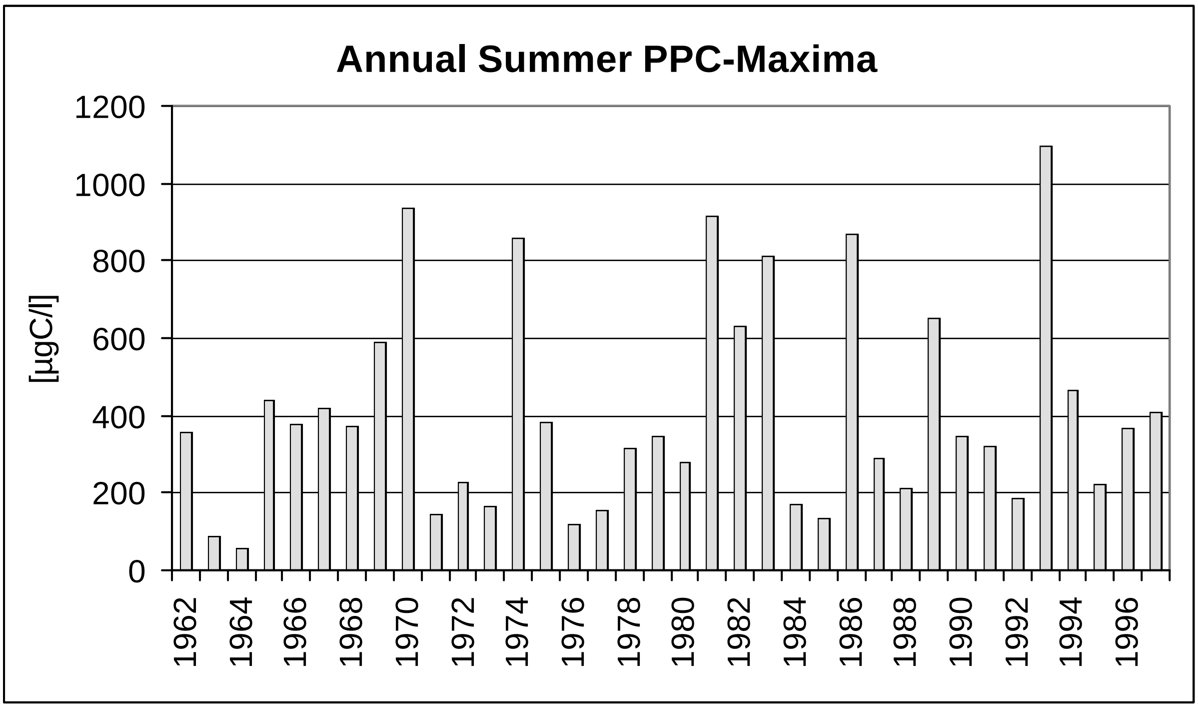


Fig. 2.4.2.2: Annual phytoplankton carbon (PPC) summer maxima (July - September) at Helgoland Roads (data kindly provided by BAH/AWI).

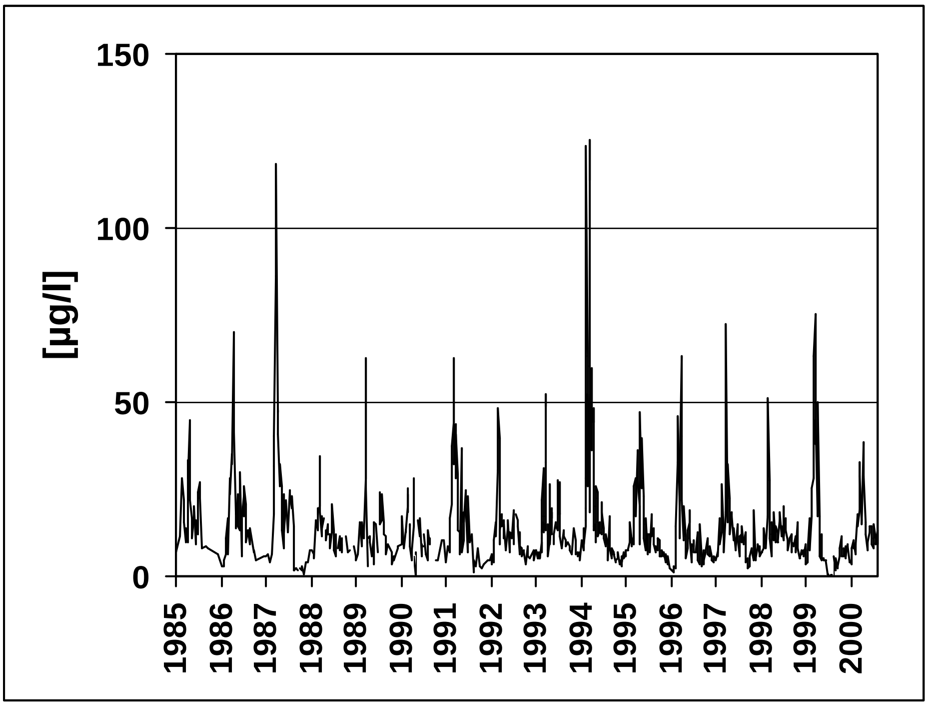


Fig. 2.4.3.1: Annual cycle of total chlorophyll concentration (active + inactive) at the permanent station Norderney (data kindly provided by NLÖ).

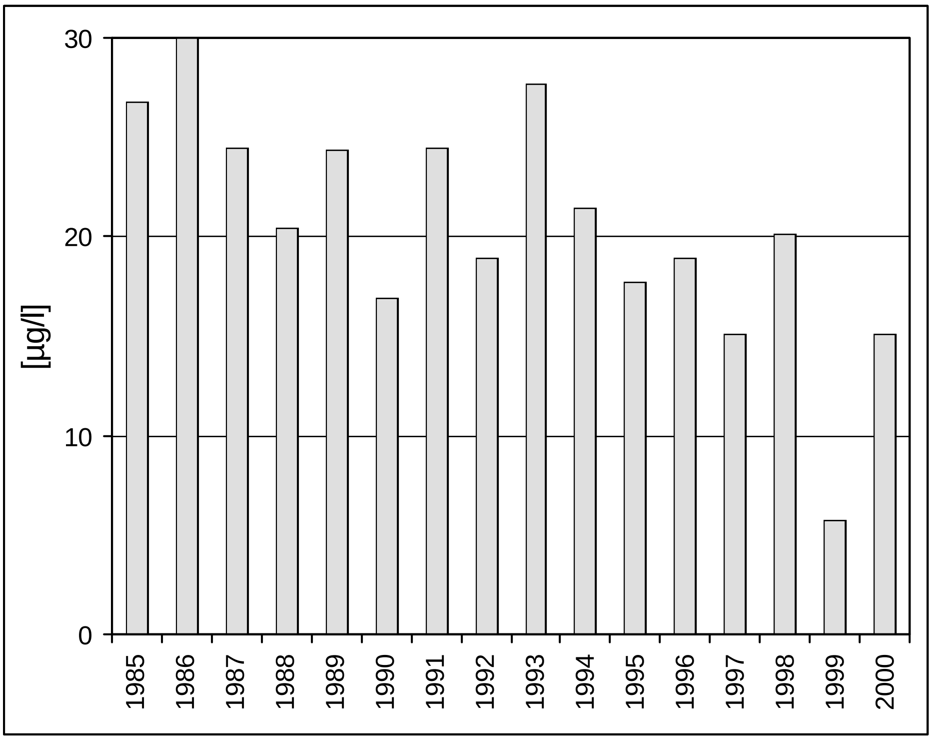


Fig. 2.4.3.2: Maximum total chlorophyll concentration (active + inactive) at the permanent station Norderney in summer (July - September)
(data kindly provided by NLÖ).

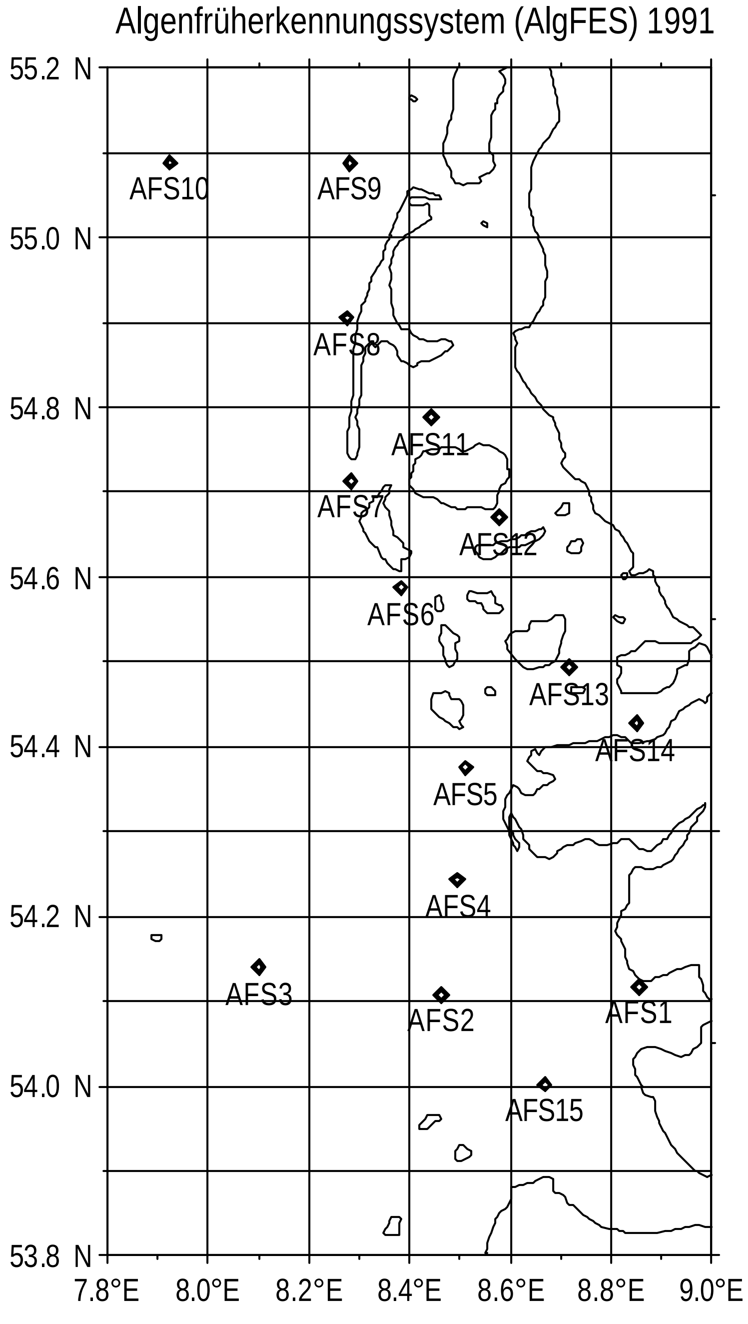


Fig. 2.4.3.3: Station grid of the AlgFes monitoring programme (Göbel 1991).

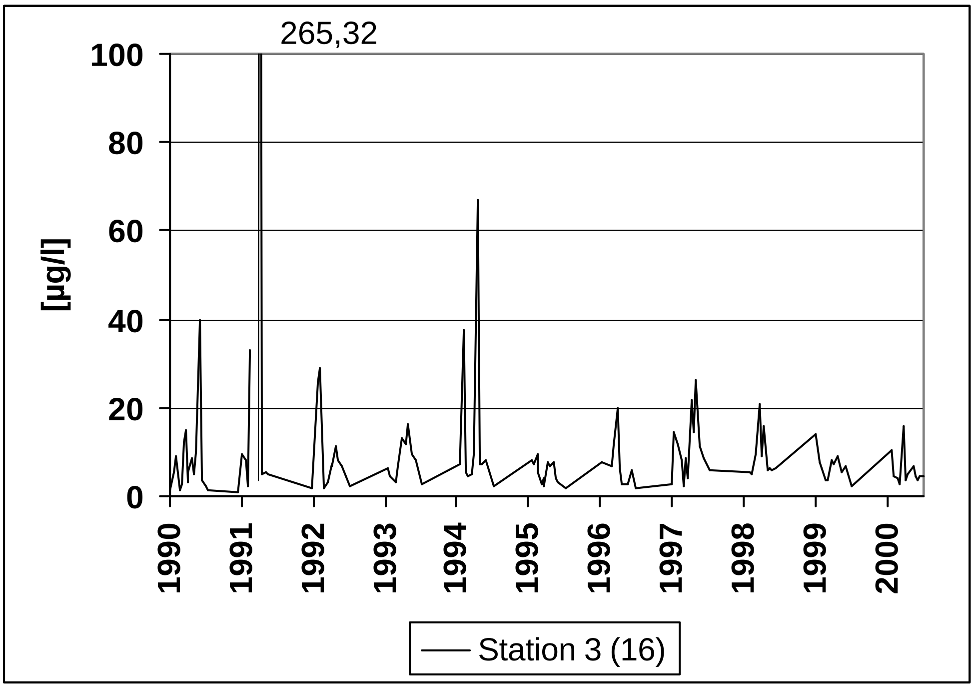


Fig. 2.4.3.4: Annual cycle of the chlorophyll-a concentration at AlgFes monitoring station 3 (later station 16) east of the island of Helgoland (data kindly provided by LANU).

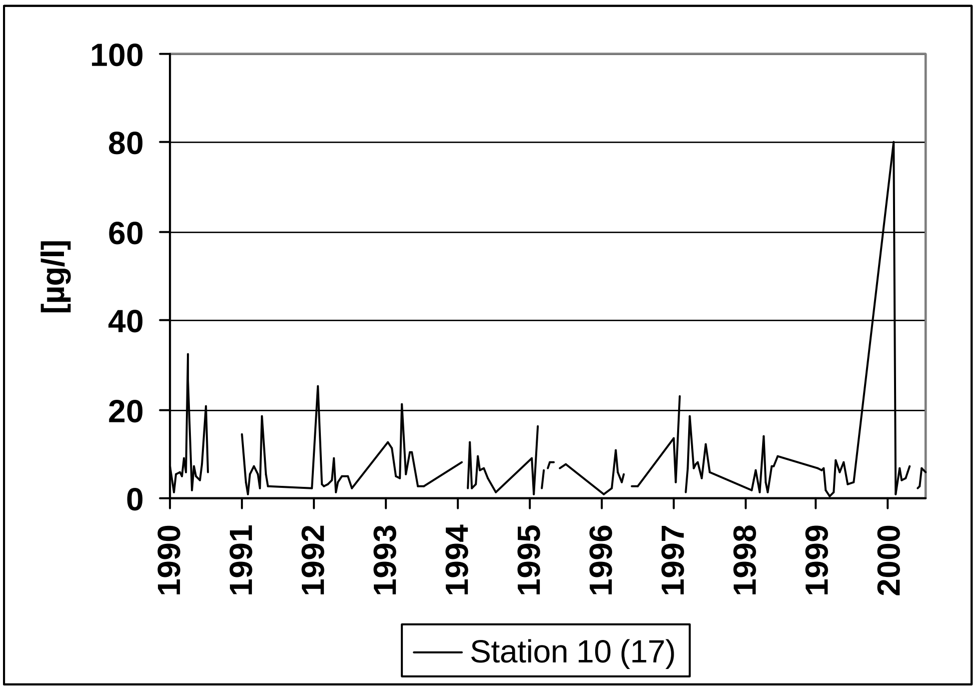


Fig. 2.4.3.5: Annual cycle of the chlorophyll-a concentration at AlgFes monitoring station 10 (later station 17) west of the island of Sylt (data kindly provided by LANU).

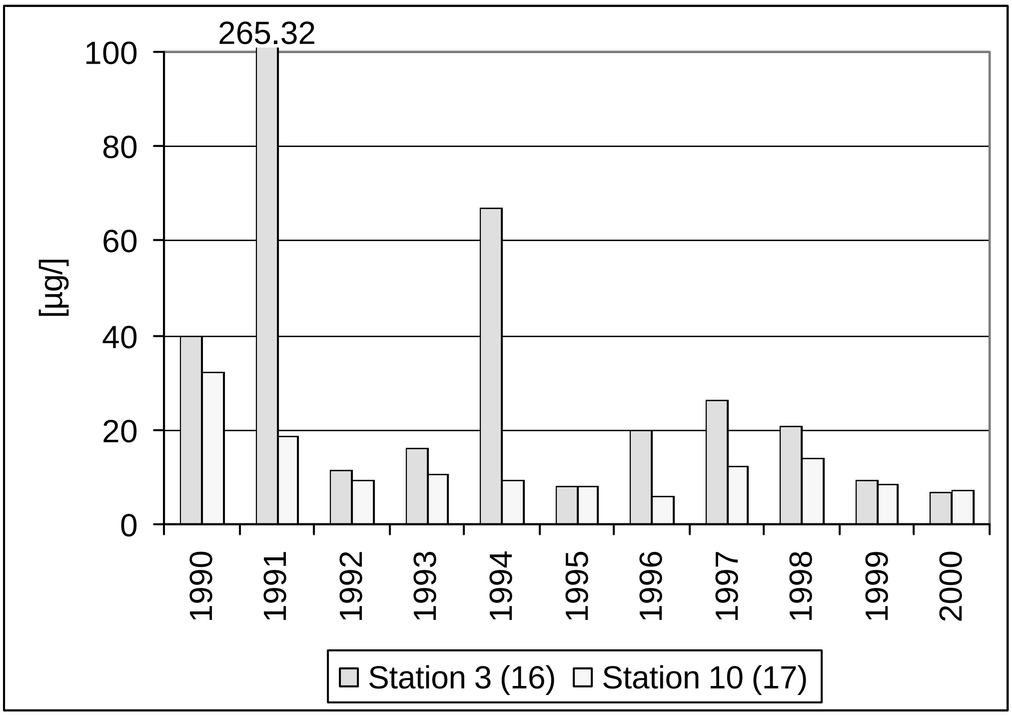


Fig. 2.4.3.6: Maxima in chlorophyll-a concentrations at two offshore monitoring stations in summer (July - September) (data kindly provided by LANU).

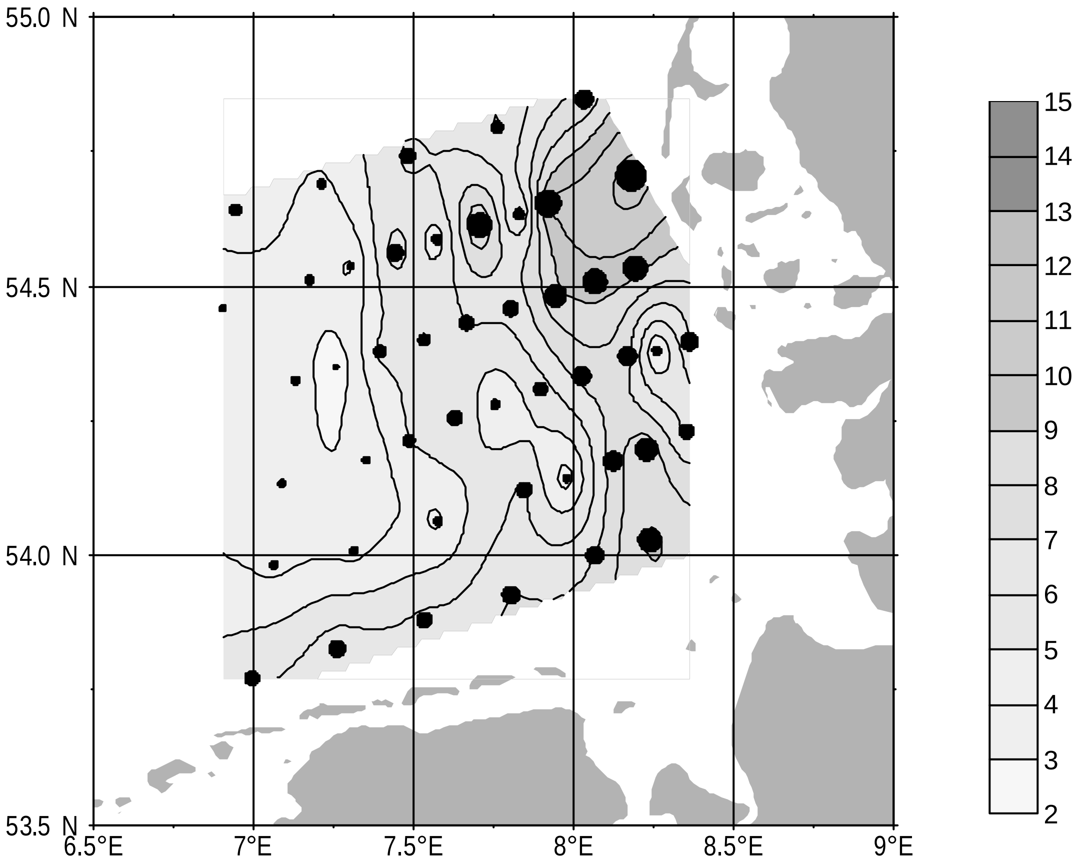


Fig. 2.4.3.7: Average surface chlorophyll-a concentration [µg/l] in the inner German Bight in summer 1994 (July - August) (data from KUSTOS).

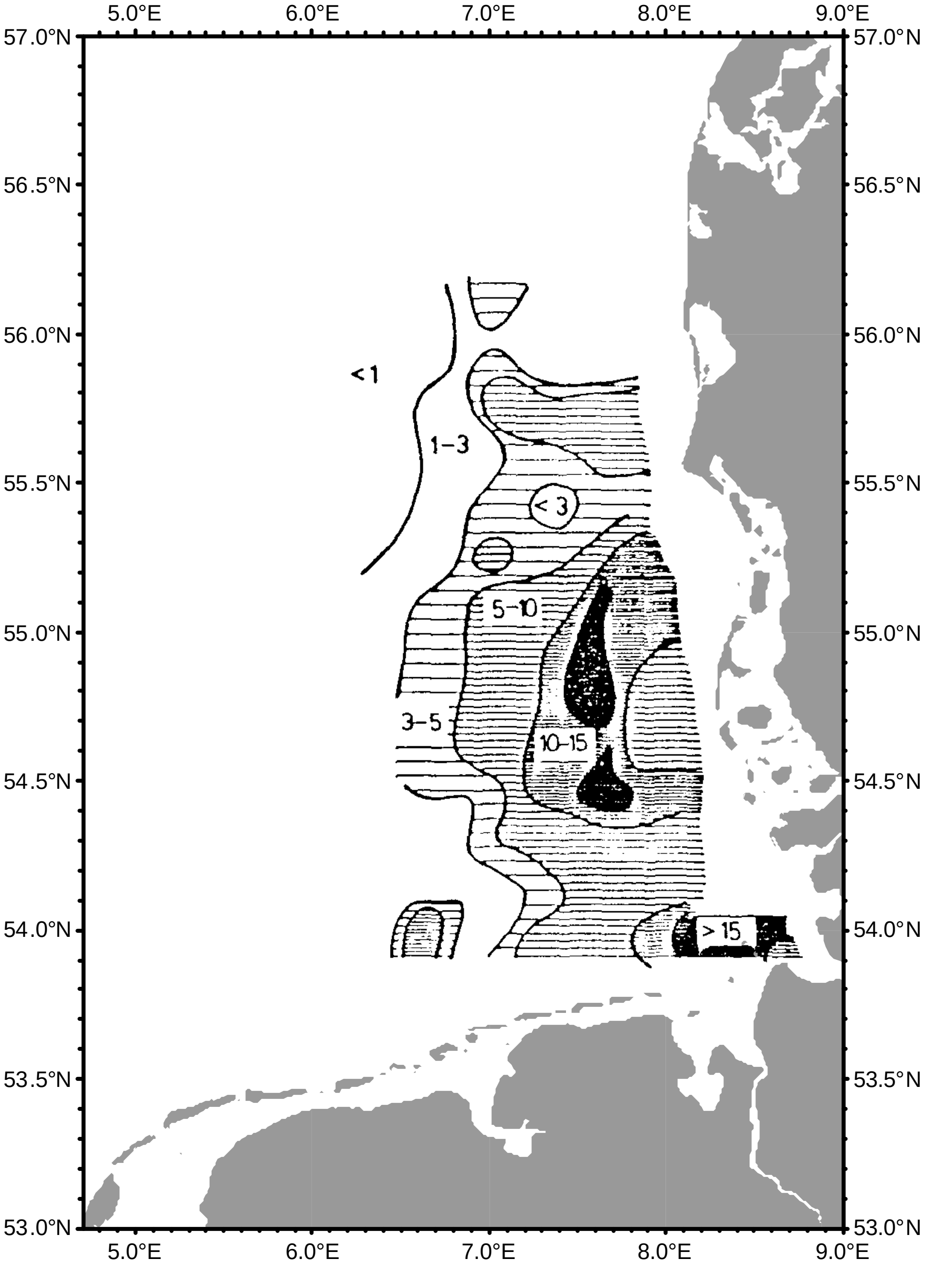


Fig. 2.4.3.8: Chlorophyll-a concentration [µg/l] in the bottom water in August 1986 [redrawn from Hickel et al. 1989].

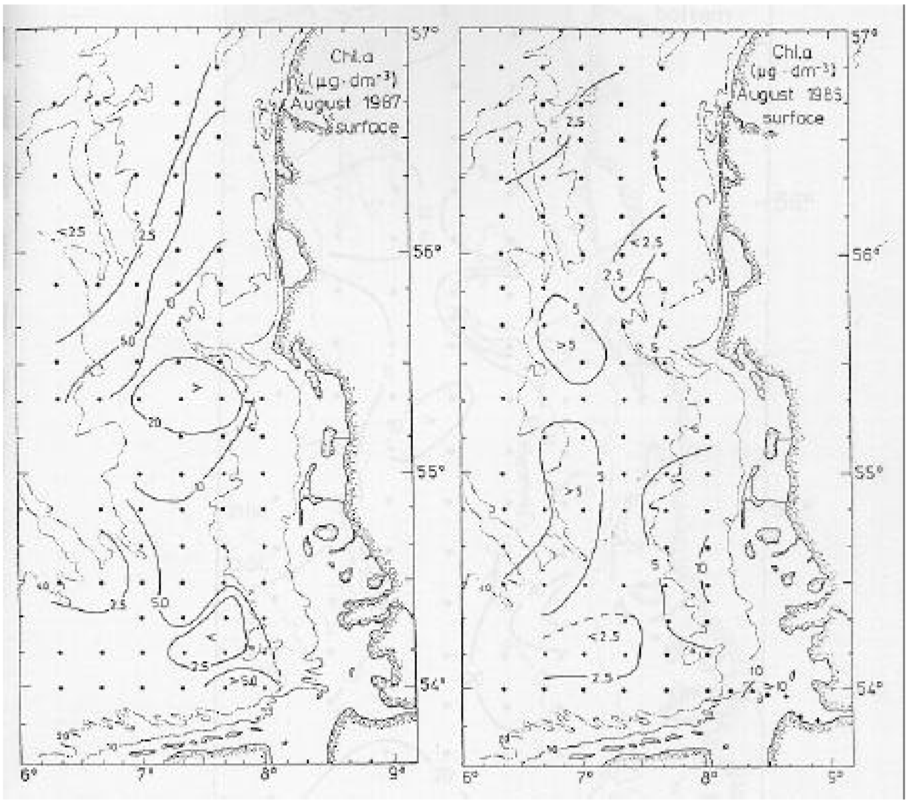


Fig. 2.4.3.9: Chlorophyll-a concentration [µg/l] in the surface water of the south - eastern German Bight in August 1987 and August 1988 (von Westernhagen et al. 1989).

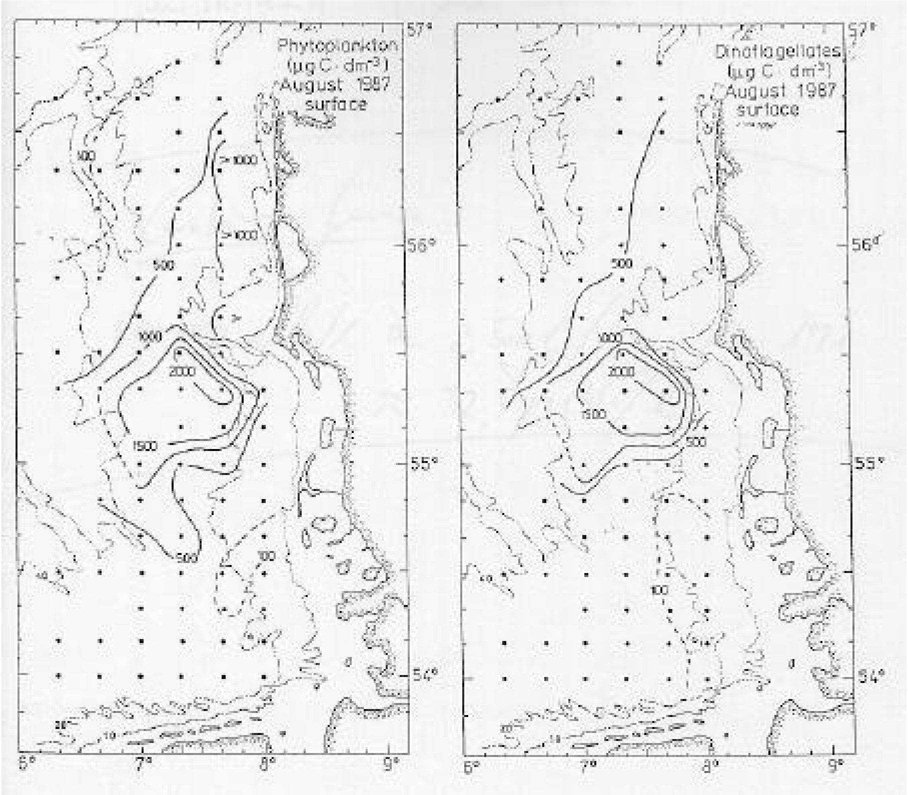
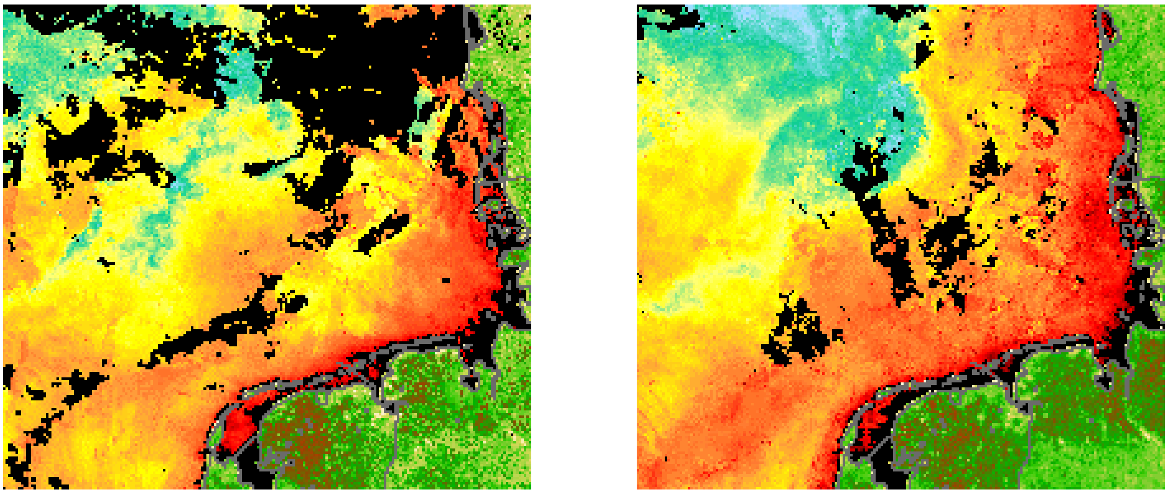
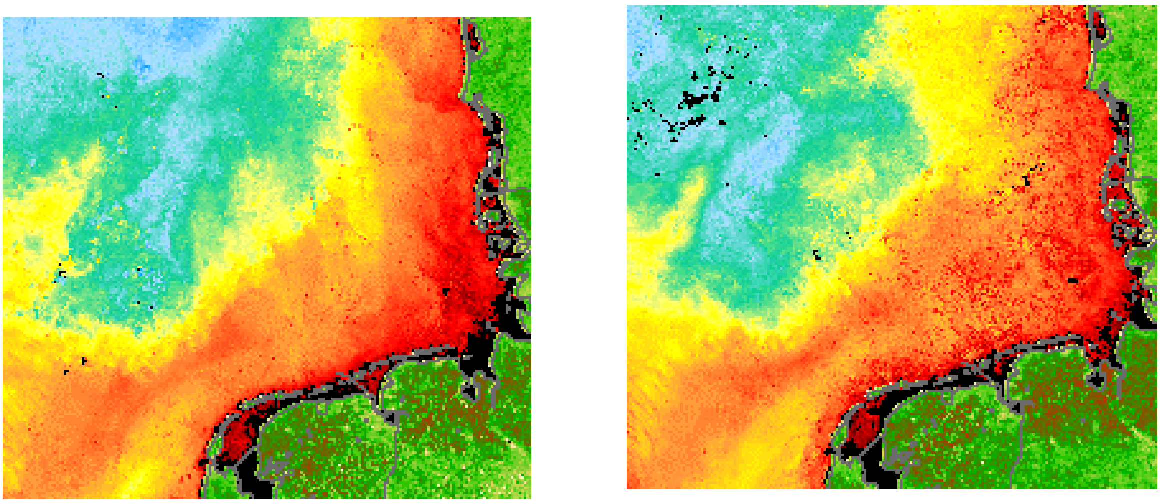


Fig. 2.4.3.10: Phytoplankton biomass (total and dinoflagellates fraction) in the surface water of the north - eastern German Bight [µg C/L] in August 1987 (von Westernhagen et al. 1989).



a. September 1997

b. August 1998



c. August 1999

d. August 2000

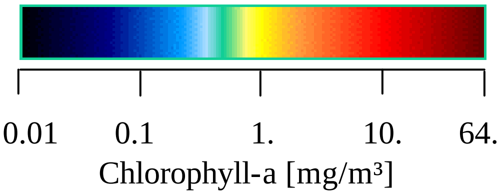


Fig. 2.4.3.11: Composites of monthly averaged surface chlorophyll-a concentrations in the German Bight in summer 1997 - 2000 (black = no data) (SeaWifs images).

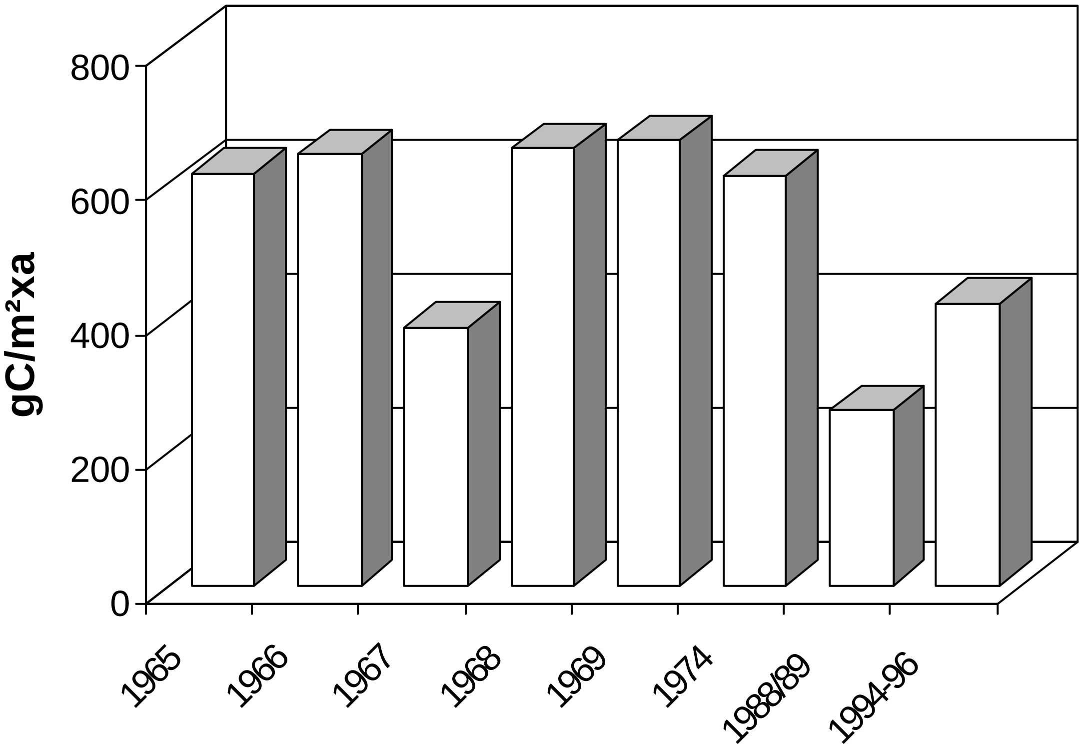


Fig. 2.4.4.1: Estimation of the annual primary production in the German Bight
(after Hagmeier 1967,1969,1970 (pers. comm.);
Joint and Pomroy 1993; Rick et al. 1998).

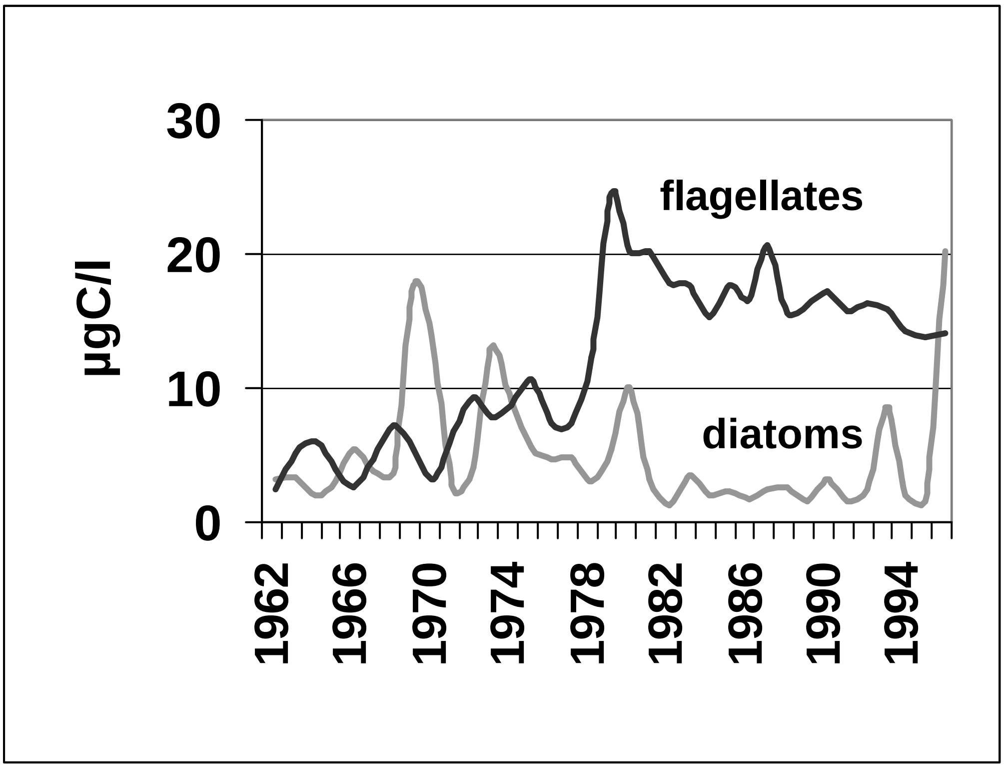


Fig. 2.5.1.1: Annual medians of the PPC concentrations at Helgoland Roads divided into functional groups (modified after Hickel 1998).

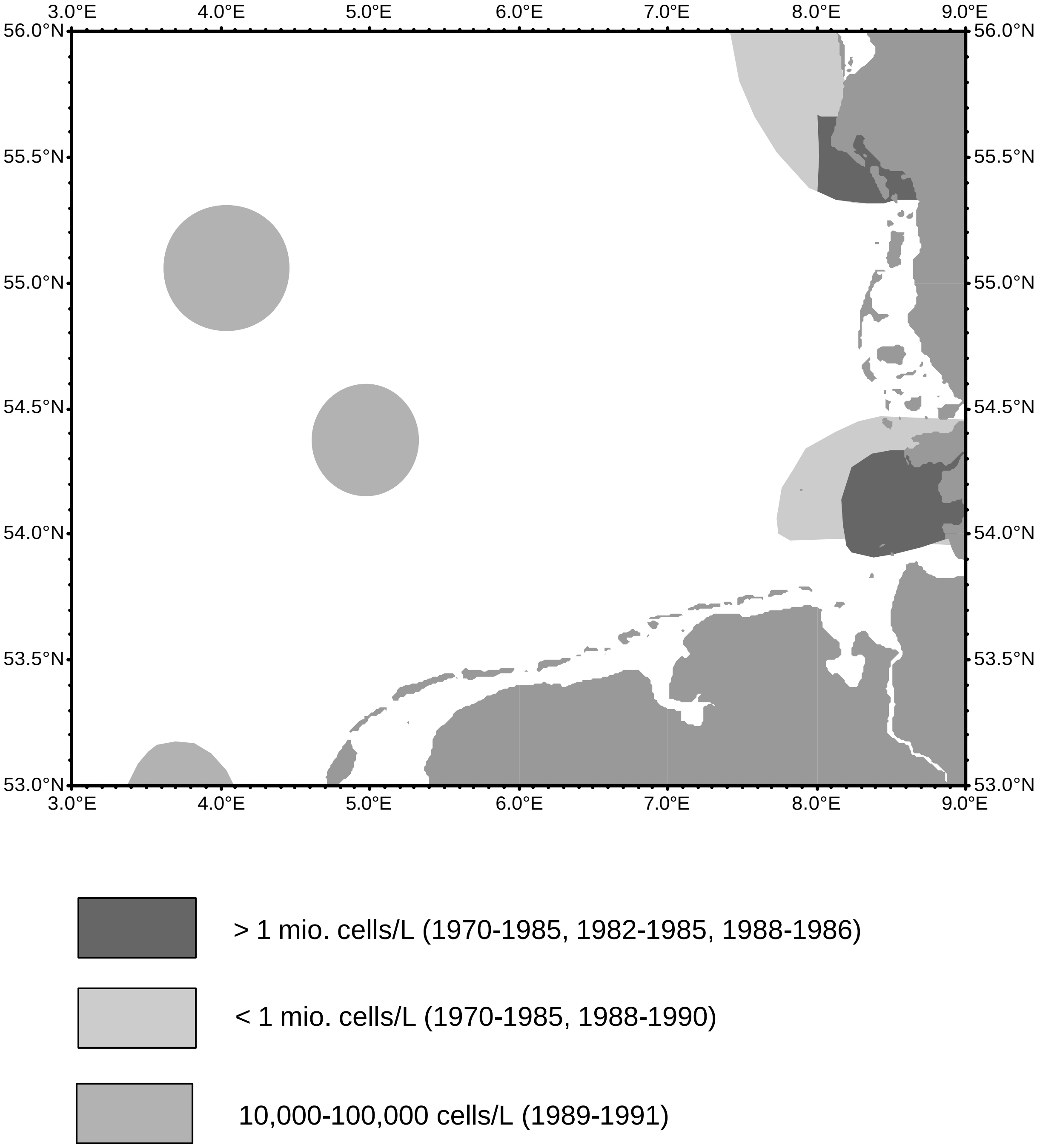
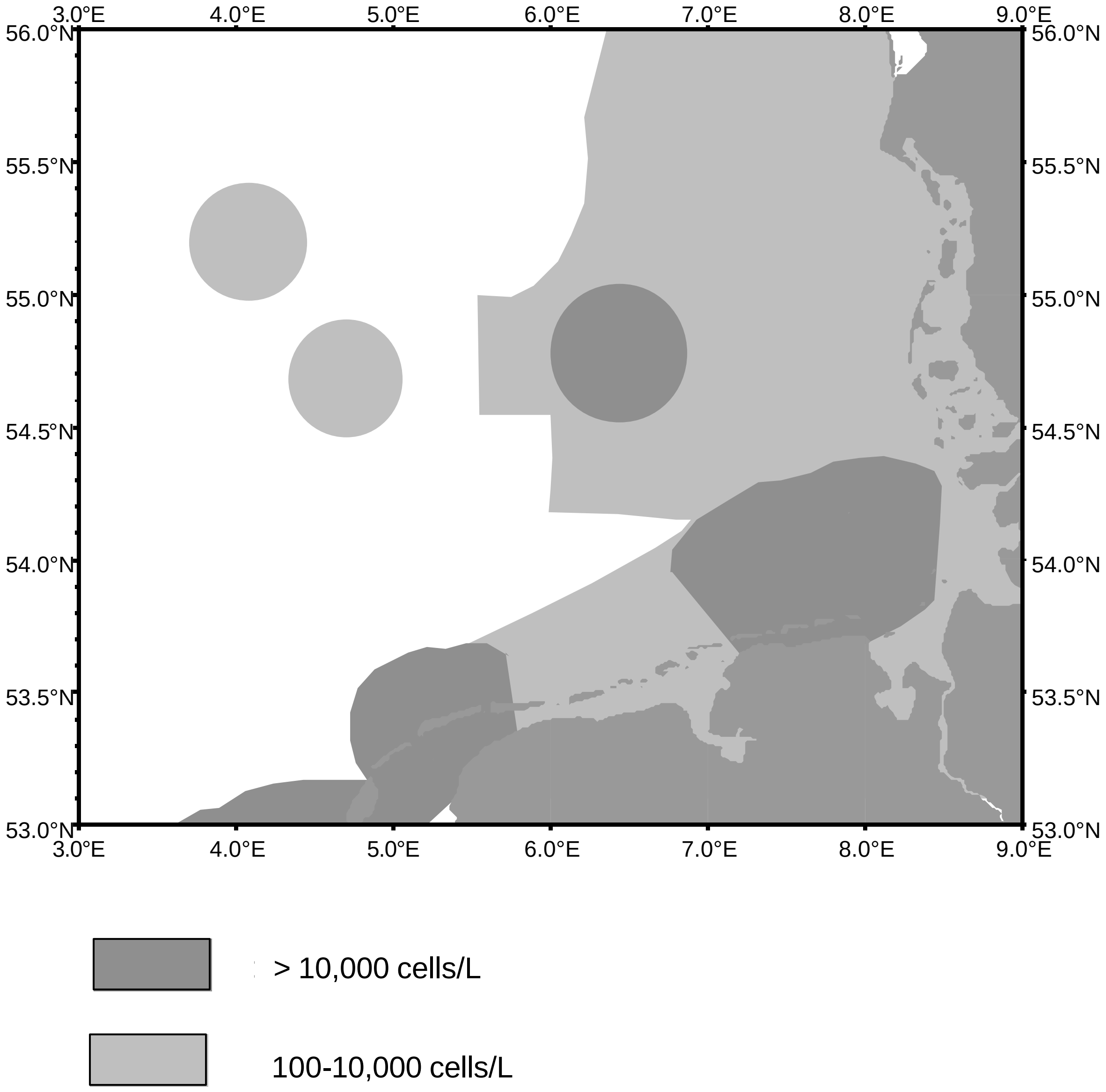


Fig. 2.5.2.1: Distribution of *Gymnodinium mikimotoi* in autumn during various years (redrawn from Zevenboom 1994)



data from: 1973-1984; 1985-1989; 1976-1990; 1983 survey and 1987-1990

Fig. 2.5.2.2: Distribution of *Dinophysis acuminata* in late summer – autumn during various years [redrawn from Zevenboom 1994]

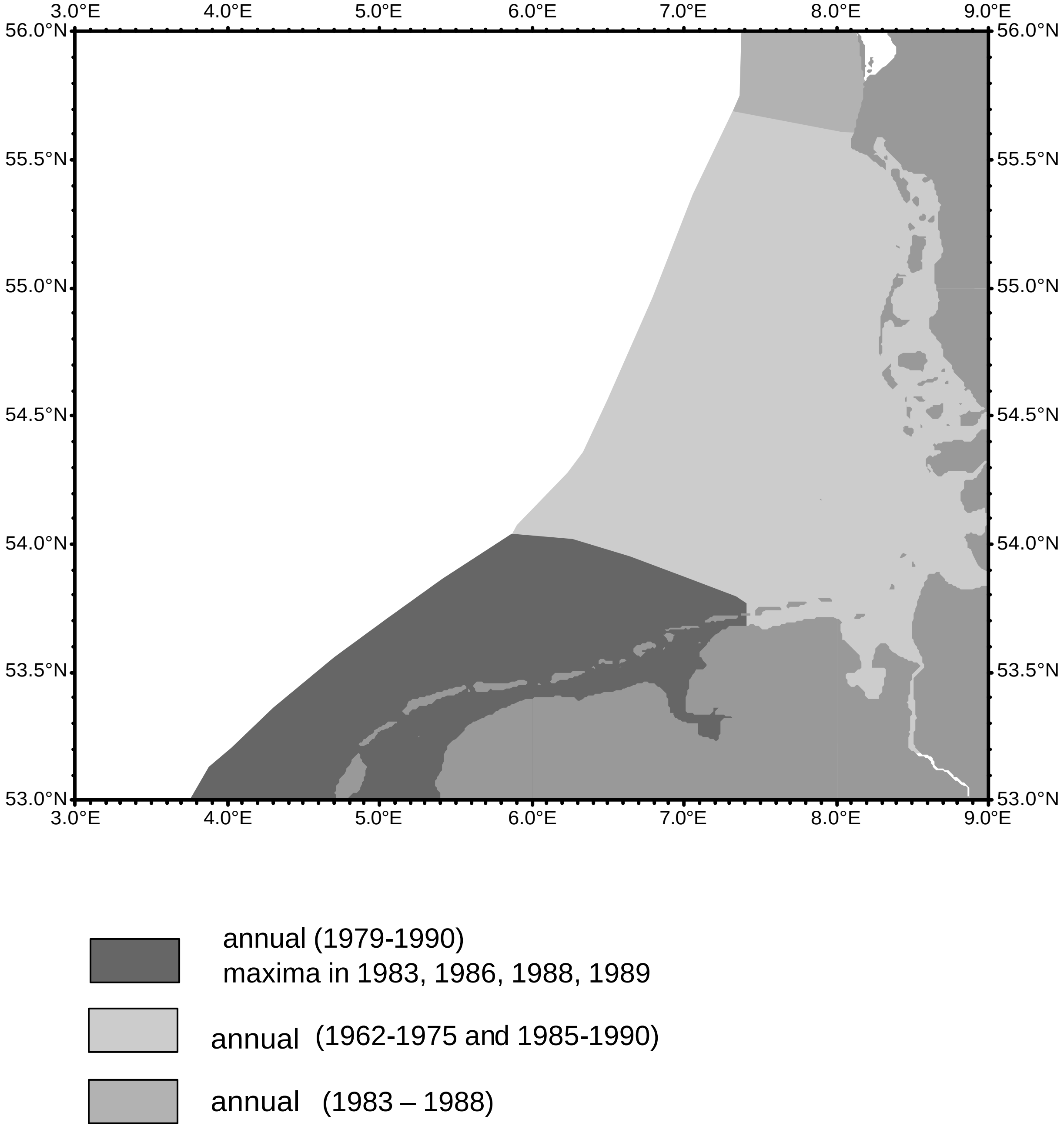


Fig. 2.5.2.3: Distribution of *Noctiluca scintillans* in spring – summer during various years [redrawn from Zevenboom 1994]

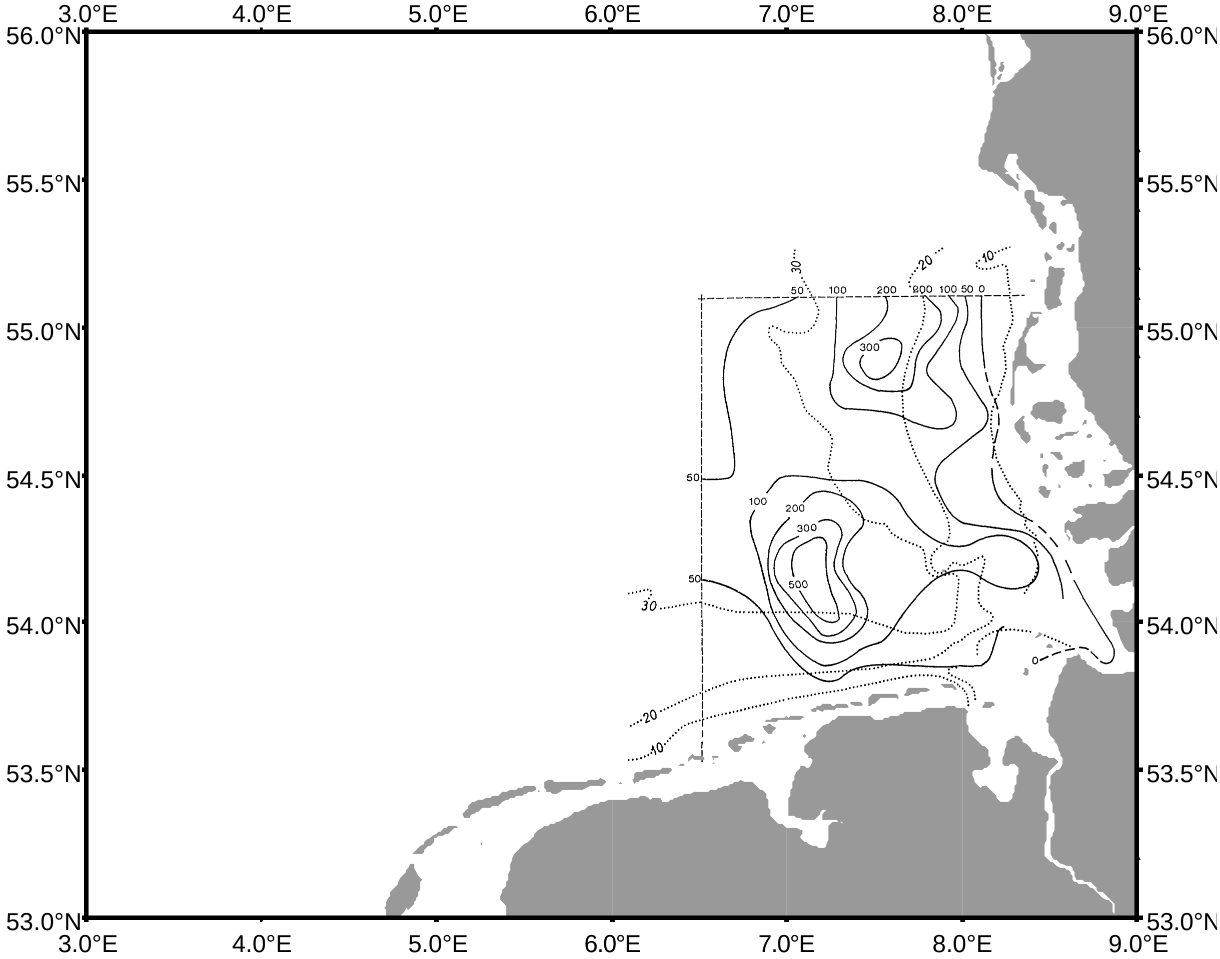


Fig. 2.5.2.4: *Ceratium furca* (x1000 cells/L) in the upper 10 m of the watercolumn of the German Bight (17. - 26. 8.1981).
Dotted line = depth contour; broken line = area of investigation.
[redrawn from Hickel 1983].

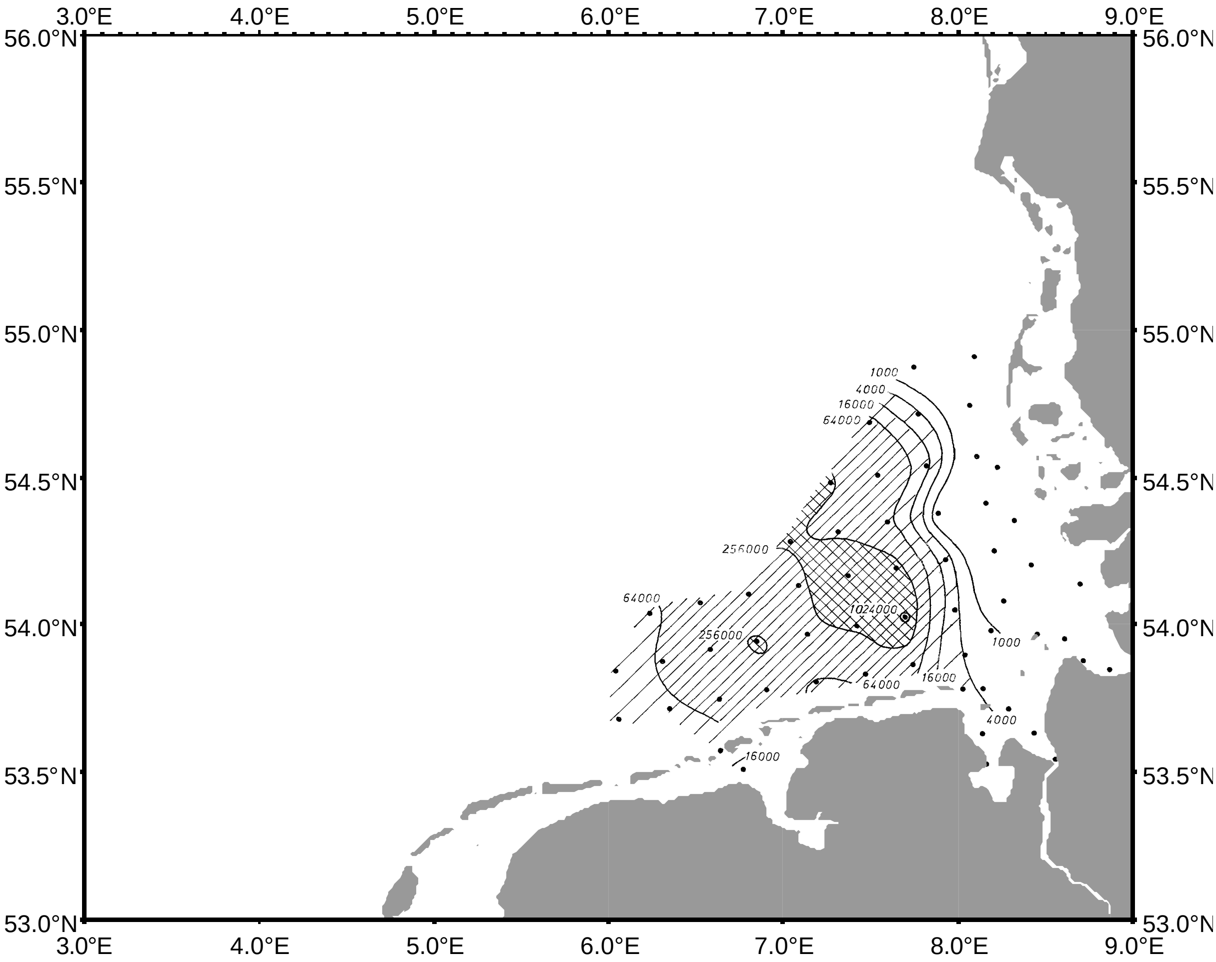


Fig. 2.5.2.5: Distribution of *Ceratium fusus* (cells/L) at the surface of the German Bight in August 1984 (31.7. - 4.8.84). [redrawn from Hagmeier 1985]

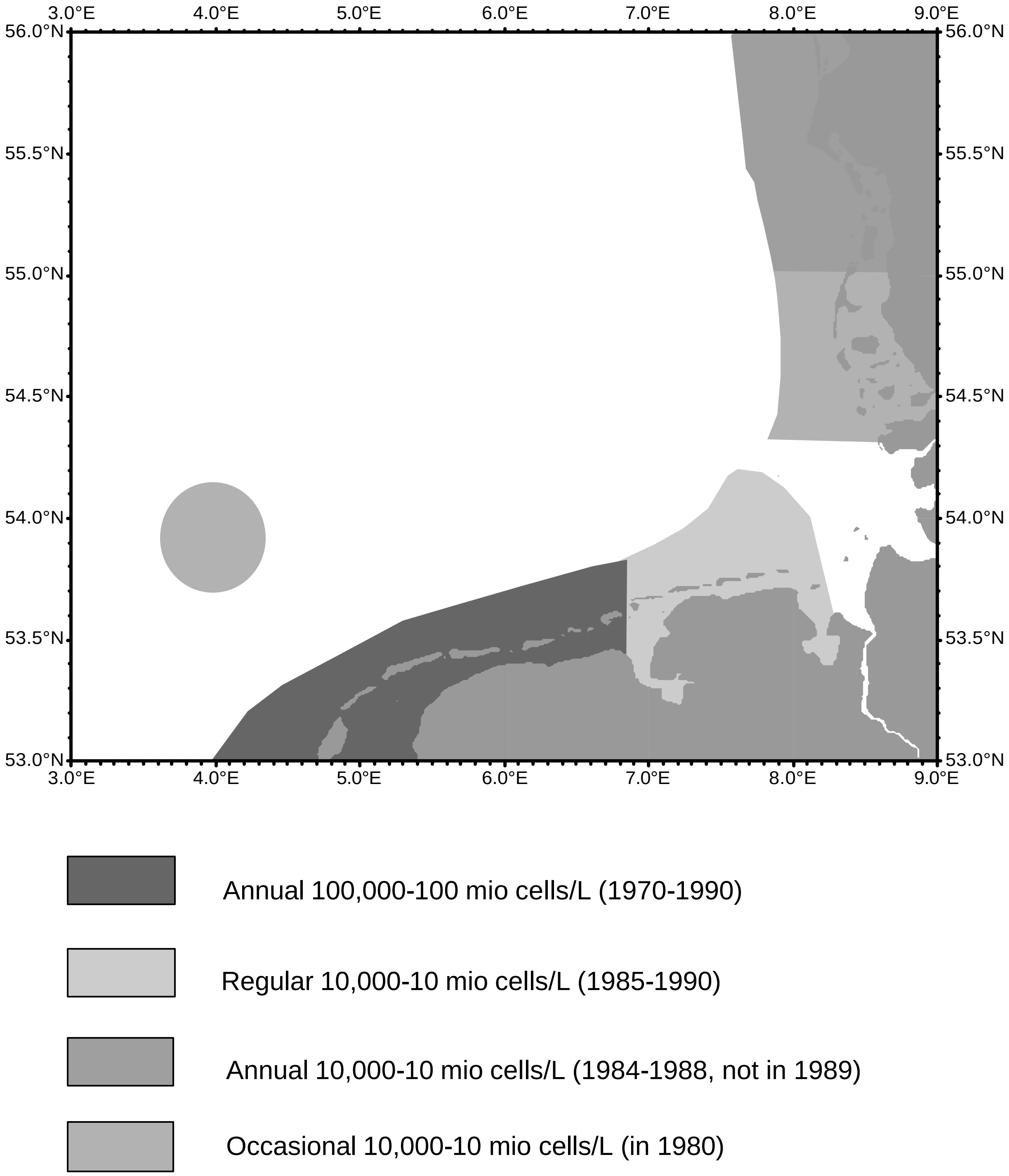


Fig. 2.5.2.6: Distribution of *Phaeocystis ssp*. in spring - summer during various years (redrawn from Zevenboom 1994)

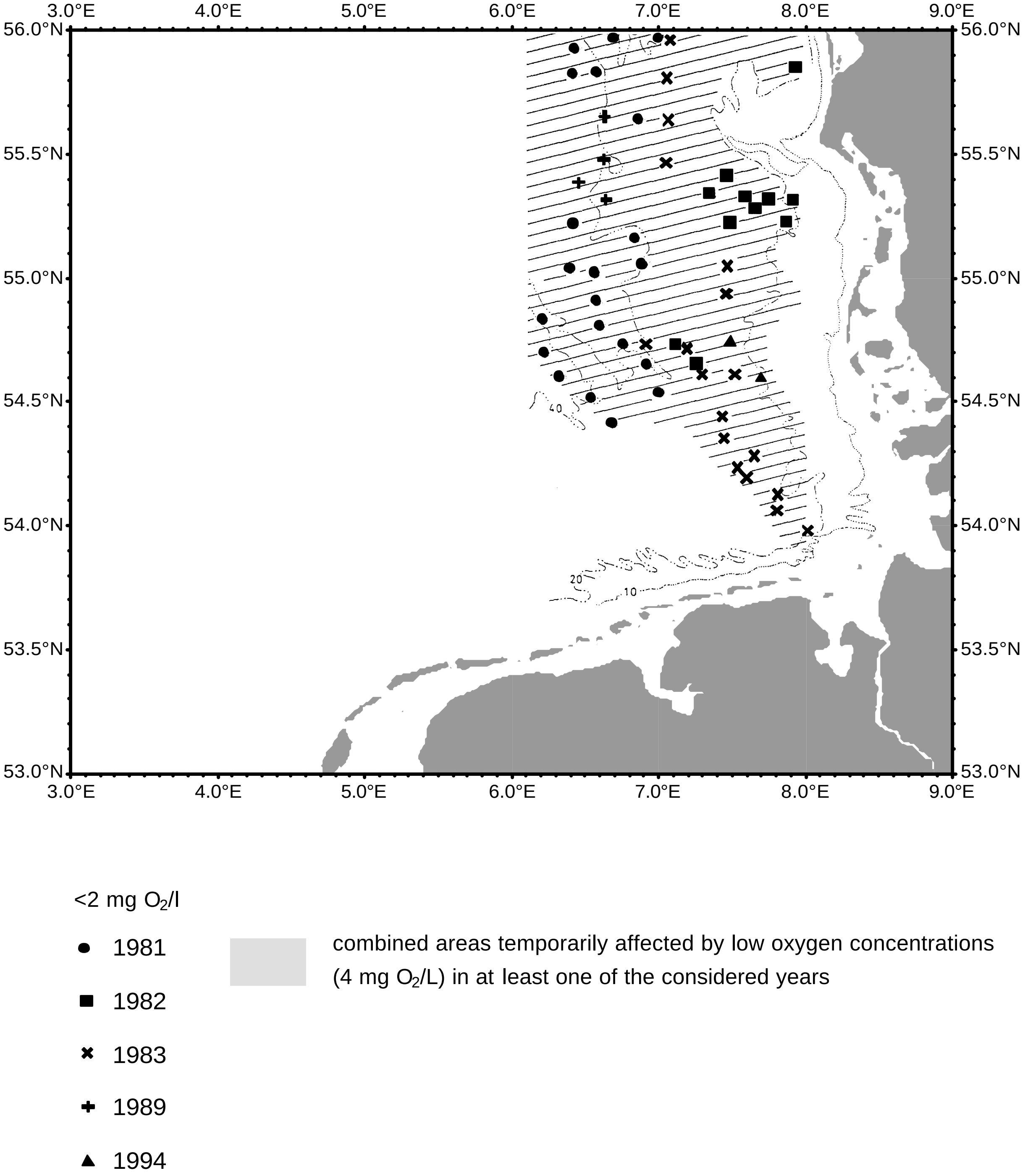


Fig. 2.7.1: Areas with oxygen deficiency in the south - eastern North Sea 1981 - 1983, 1989 and 1994 [redrawn from Westernhagen et al. 1989; Niermann 1990; data 1994 from the KUSTOS-project.

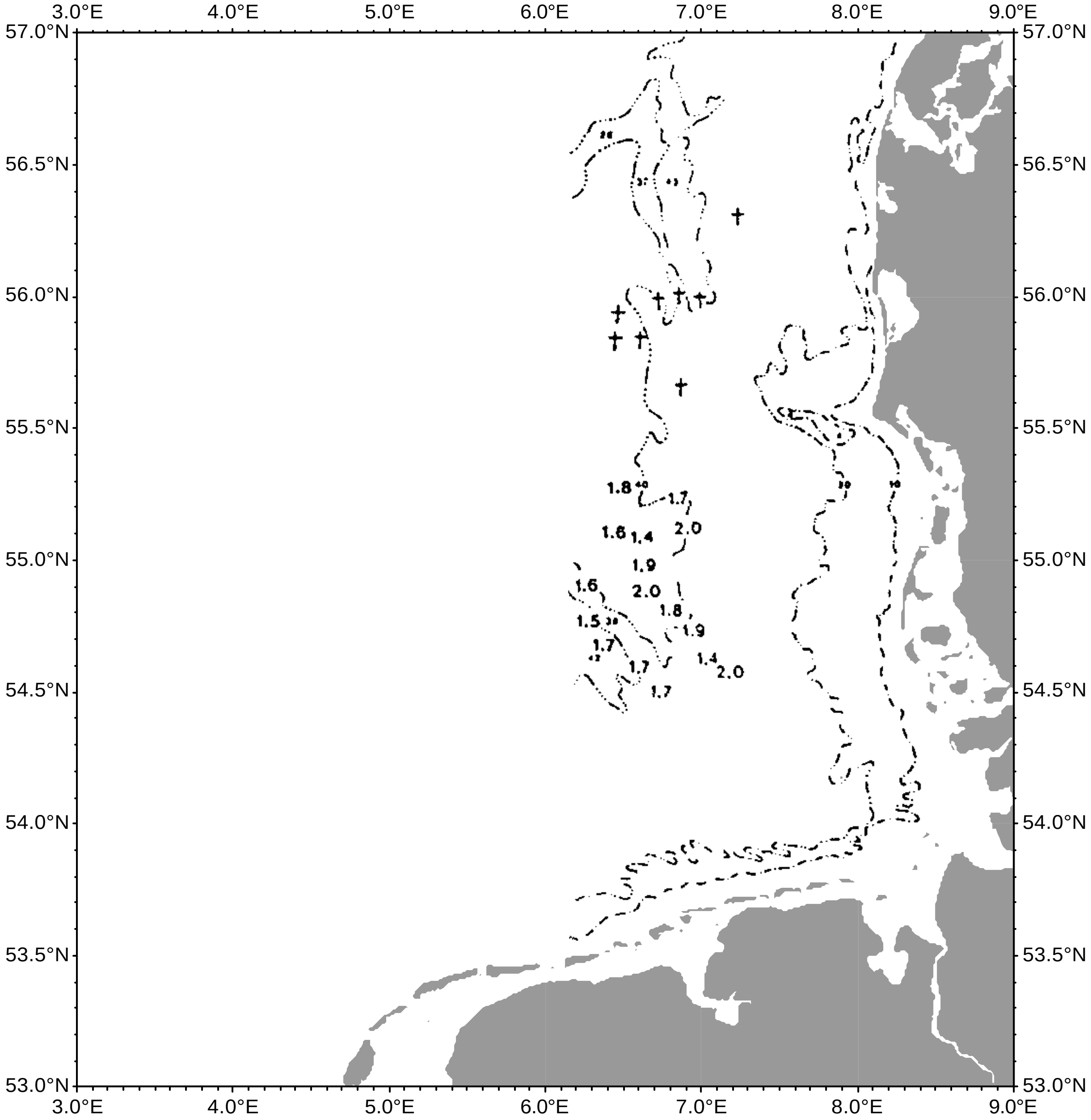


Fig. 2.7.2: Observations of bottom water with < 3 mg O_2/L (< 2ml O_2/L or 30% oxygen saturation, respectively) and fish mortality (crosses) in the German Bight in September 1981.
Dotted/broken lines = depth contours [redrawn from Gerlach 1990]

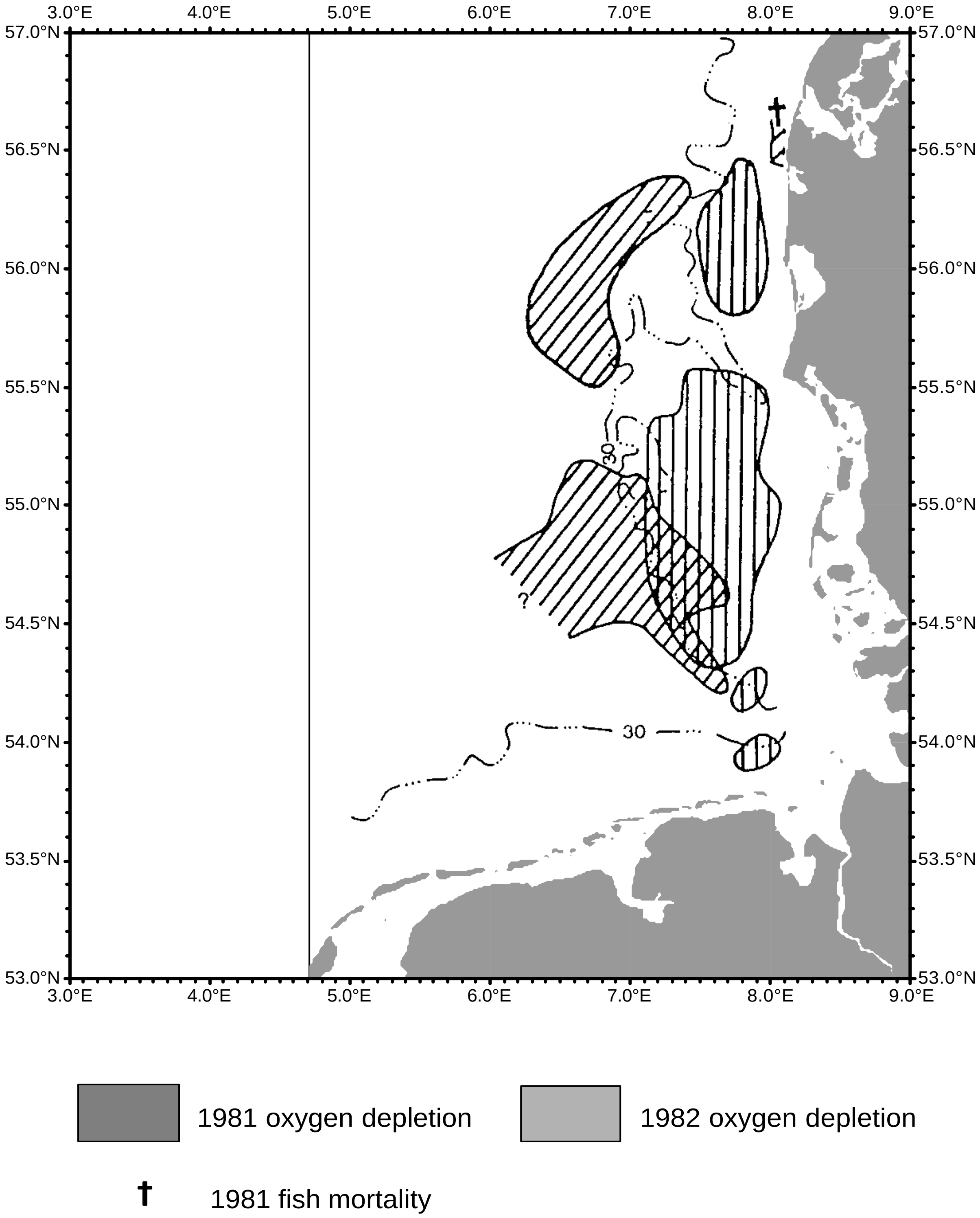


Fig. 2.7.3: Observations of bottom water with < 4 mg O_2/L or < 50% oxygen saturation in the German Bight, 1981 – 1982. [redrawn from Brockmann and Eberlein 1986; Gerlach 1990]

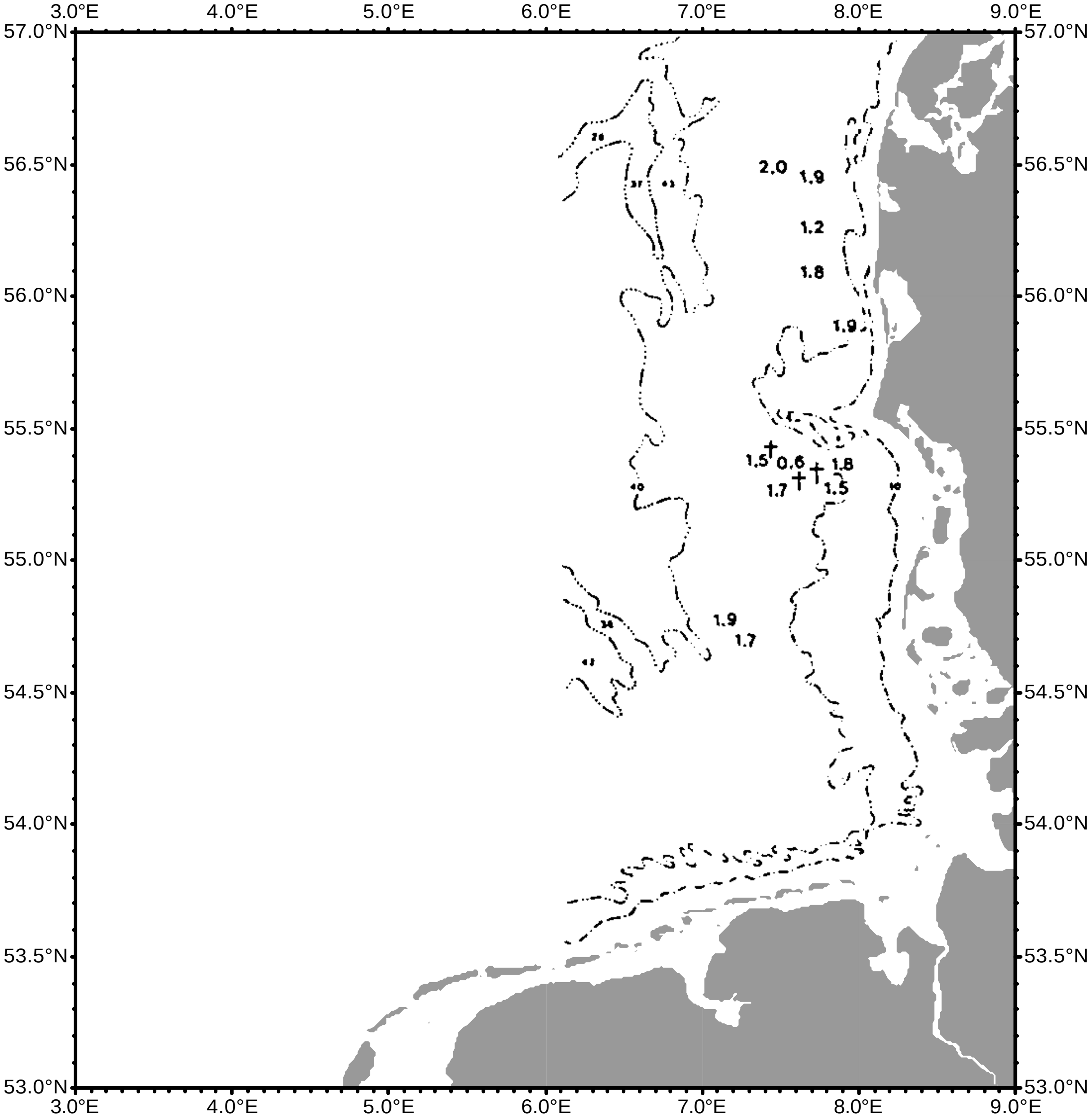


Fig. 2.7.4: Observations of bottom water with < 3 mg O_2/L (< 2 ml O_2/L or 30% oxygen saturation) and fish mortality (crosses) in the German Bight in August 1982. Dotted/broken lines = depth contours [redrawn from Gerlach 1990]

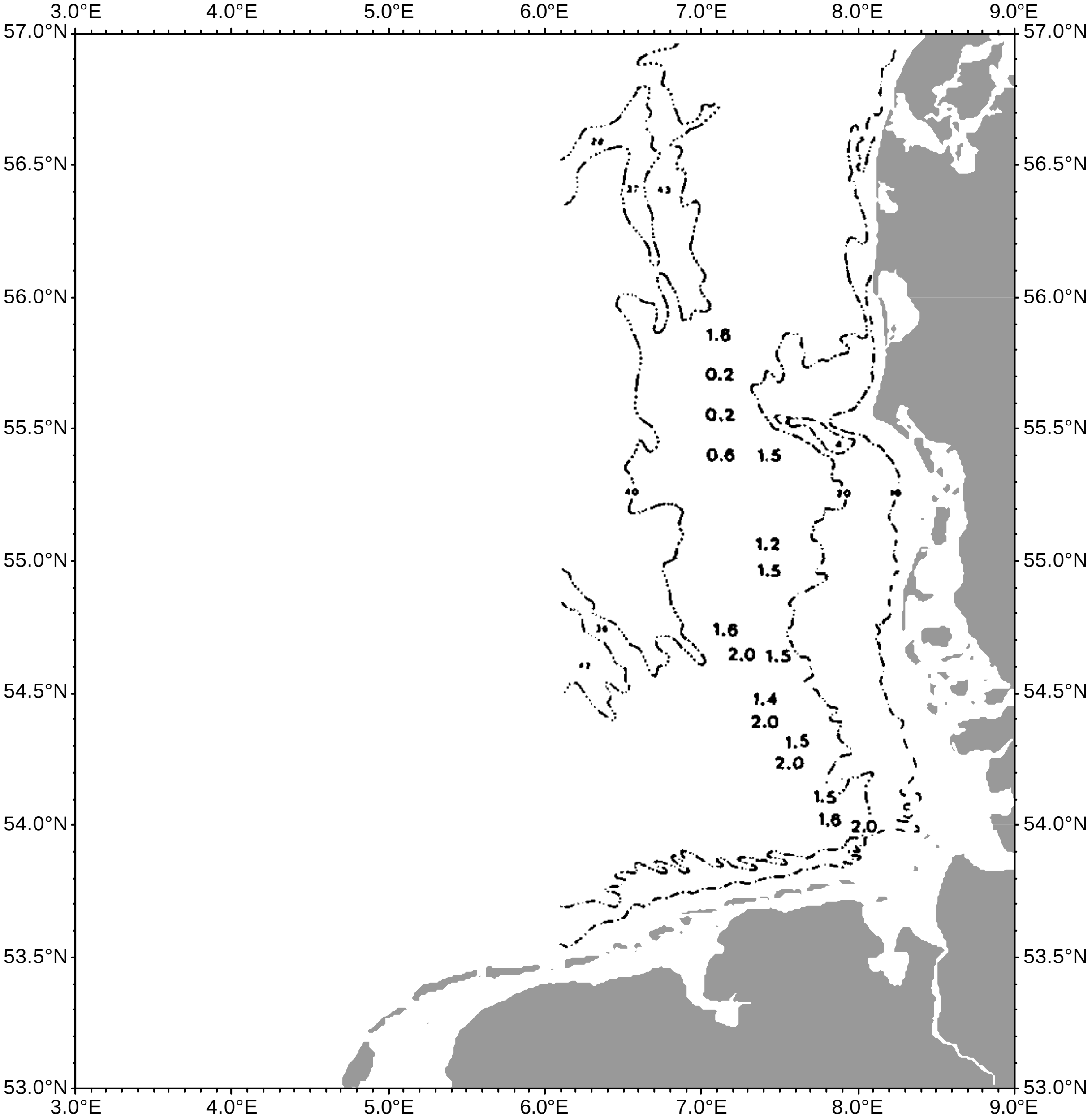


Fig. 2.7.5: Observations of bottom water with < 3 mg O_2/L (< 2 ml O_2/L or 30% oxygen saturation) in the German Bight in August 1983. Dotted/broken lines = depth contours [redrawn from Gerlach 1990]

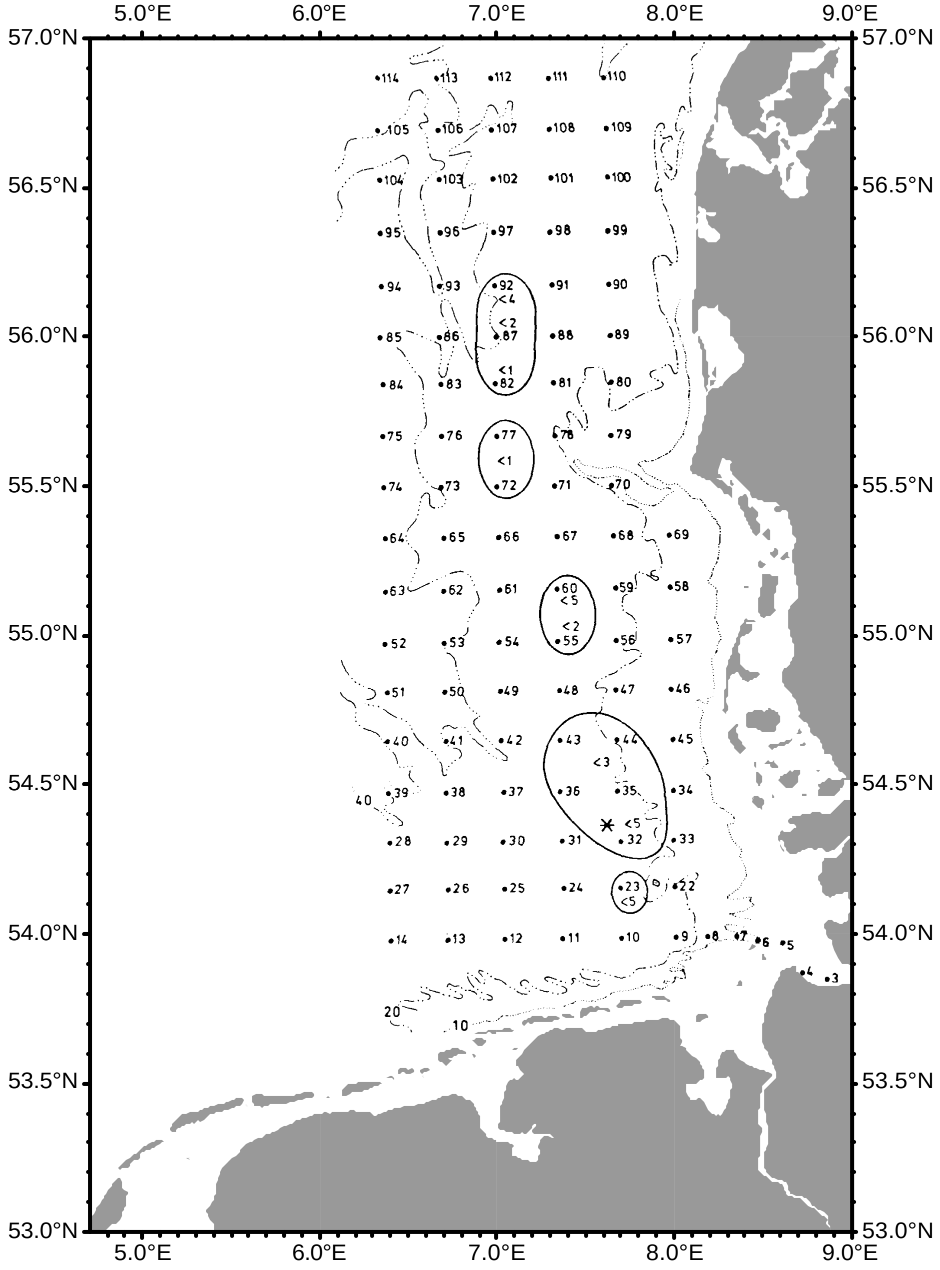


Fig. 2.7.6: Areas in which the fauna was affected to different degrees by oxygen depletion in 1983. The oxygen depletion in August 1983 of the bottom water is indicated inside area. [redrawn from Hickel et al. 1989]

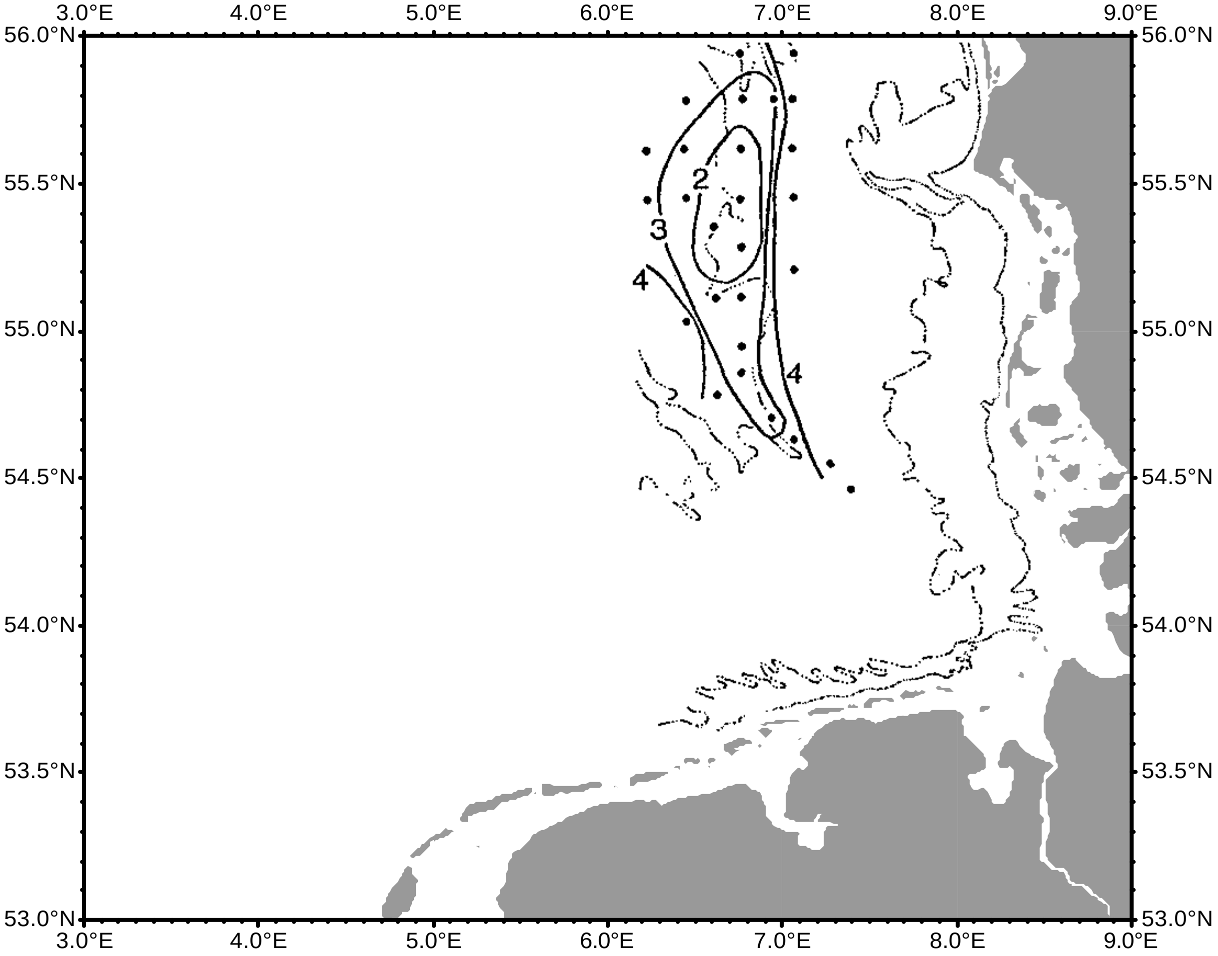
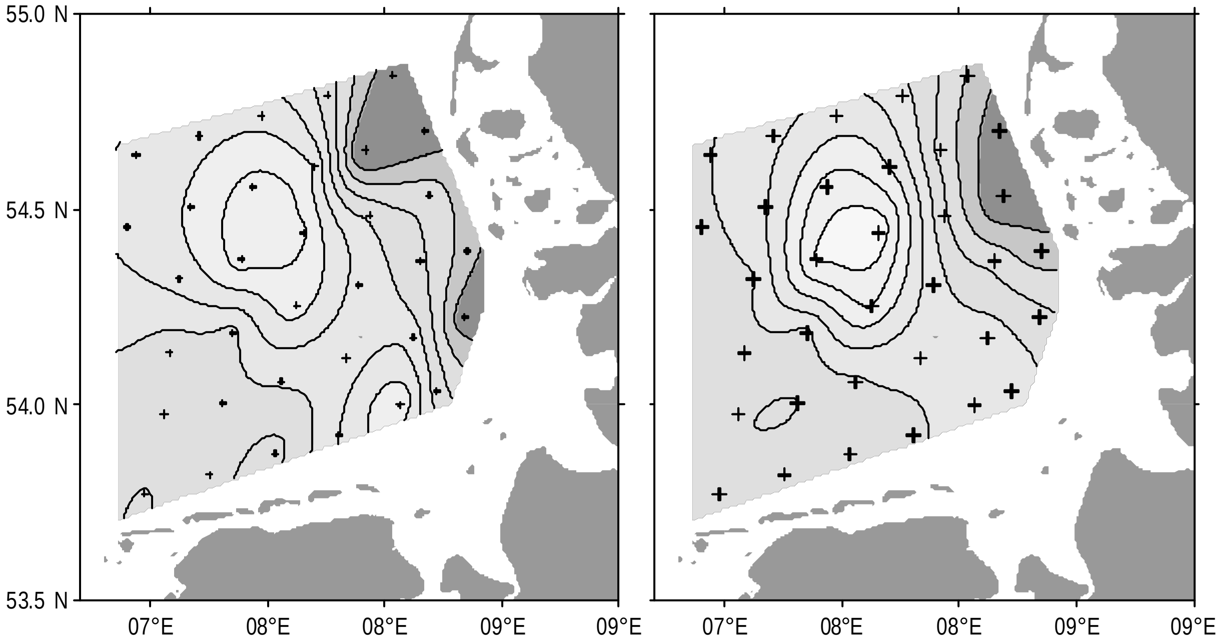
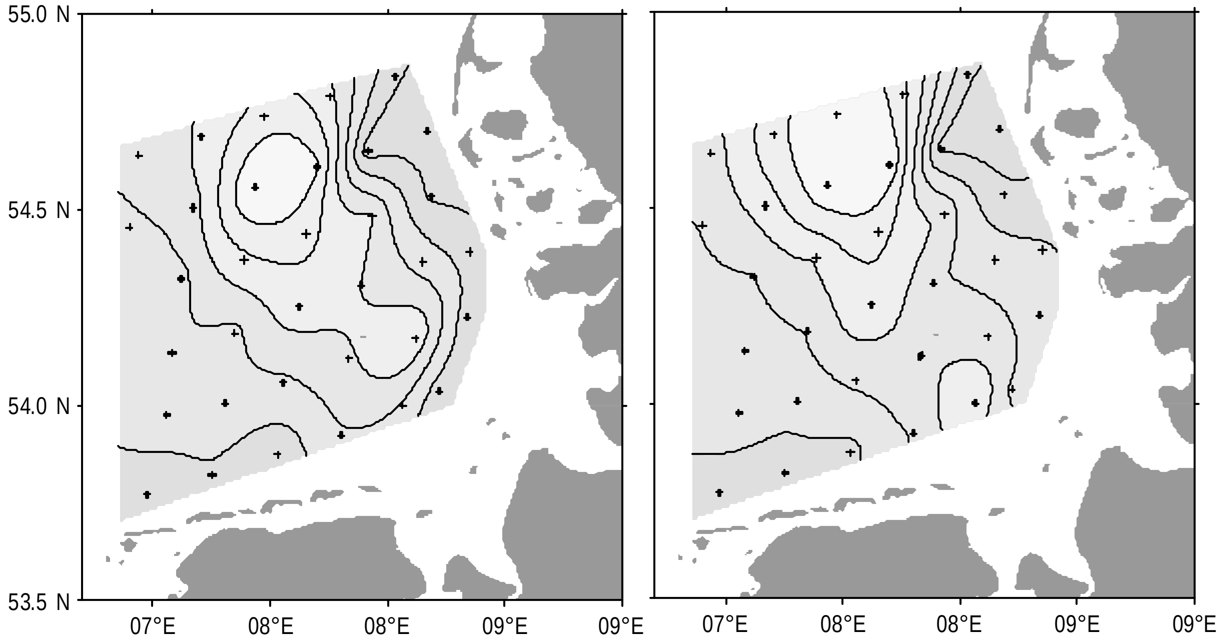


Fig. 2.7.7: Oxygen concentration (mg O_2/L) in the bottom water of the south-eastern North Sea (12. - 15.9.1989). Dotted lines: depth contours of 10, 20 and 40 m. [redrawn from Niermann 1990]



Grid 0: 23.07. - 25.07.94

Grid 1: 26.07. - 29.07.94



Grid 2: 02.08. - 04.08.94

Grid 3: 07.08. - 11.08.94

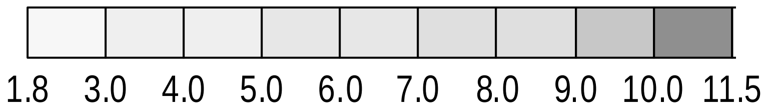


Fig. 2.7.8: Oxygen concentrations [mg/L] in the bottom water of the German Bight during 4 succeeding grids in July/August 1994

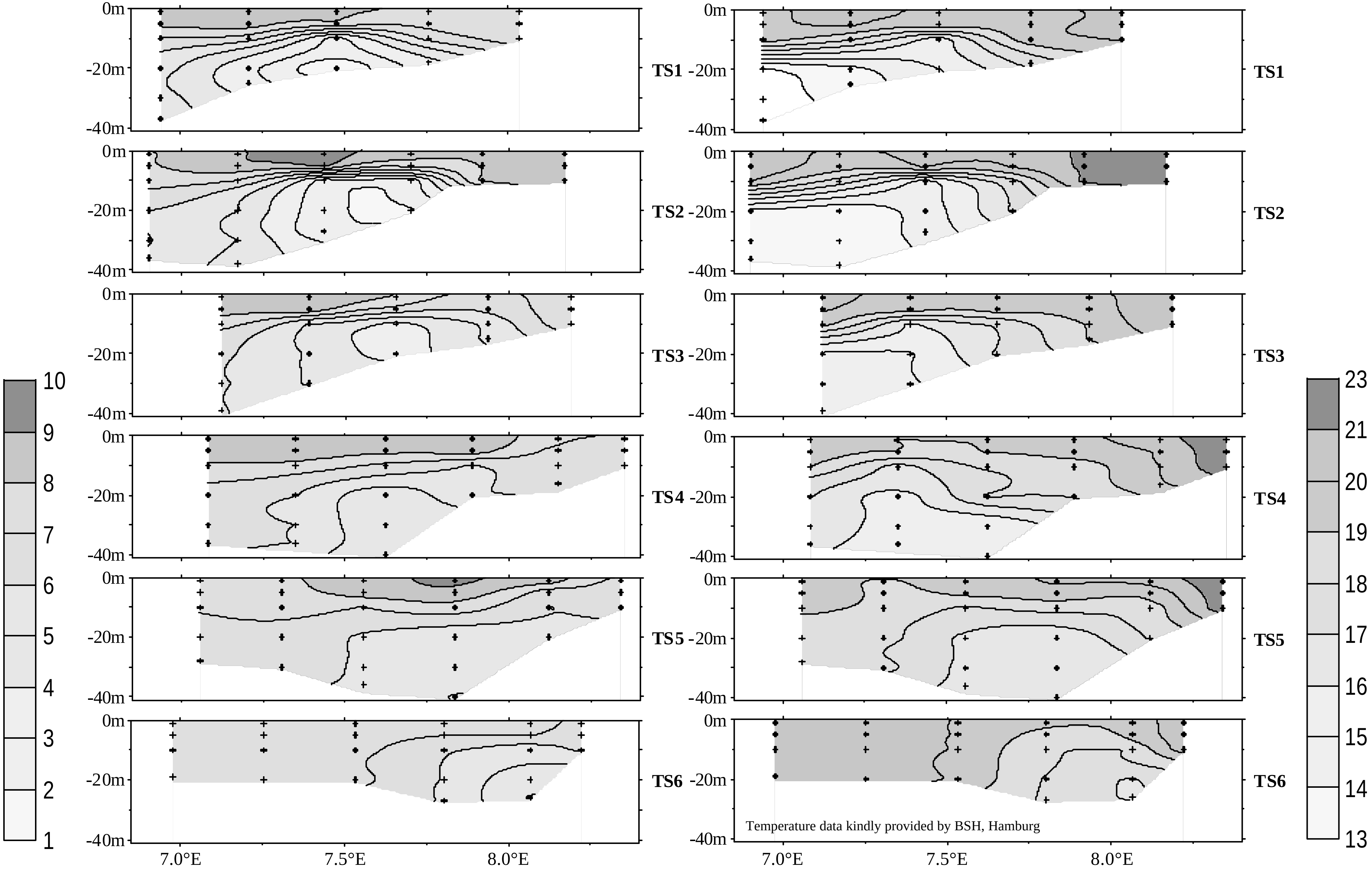


Fig. 2.7.9: Vertical profiles of oxygen concentration [mg/L], left, and temperature [°C], right, in the German Bight during grid 3 sampling (7.8. – 11.8.1994)

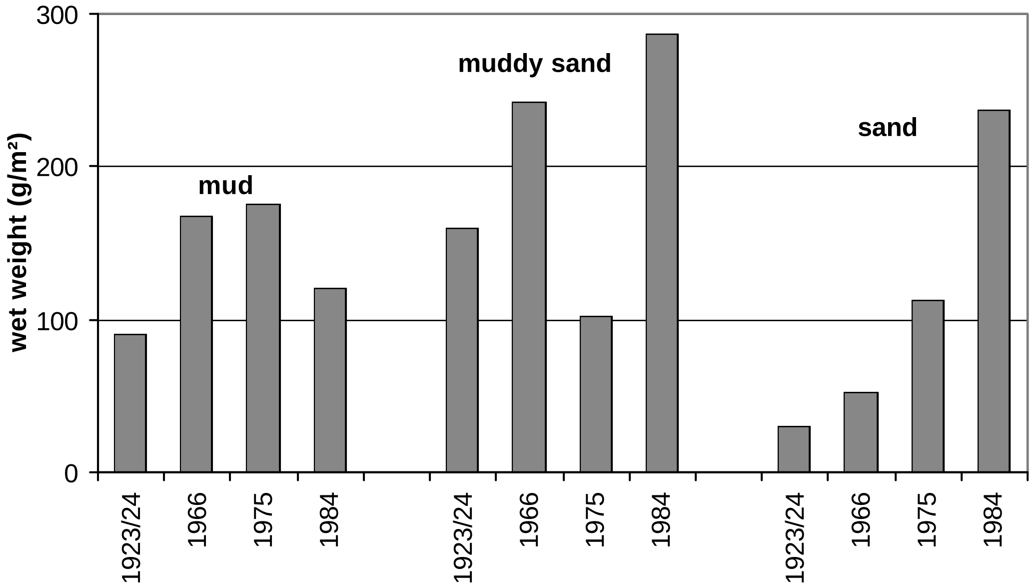


Fig. 2.8.2.2.1: Long-term comparison of macrozoobenthos biomass for different sediment types in the German Bight (included are: crustaceans, echinoderms, polychaetes, molluscs and miscellaneous)
(modified after Rachor, 1990).

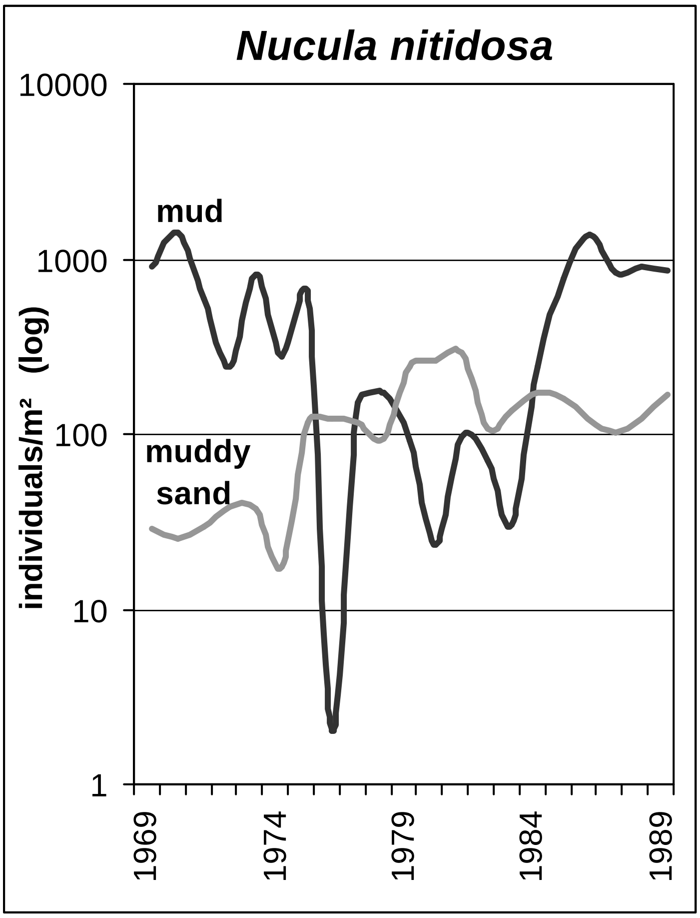
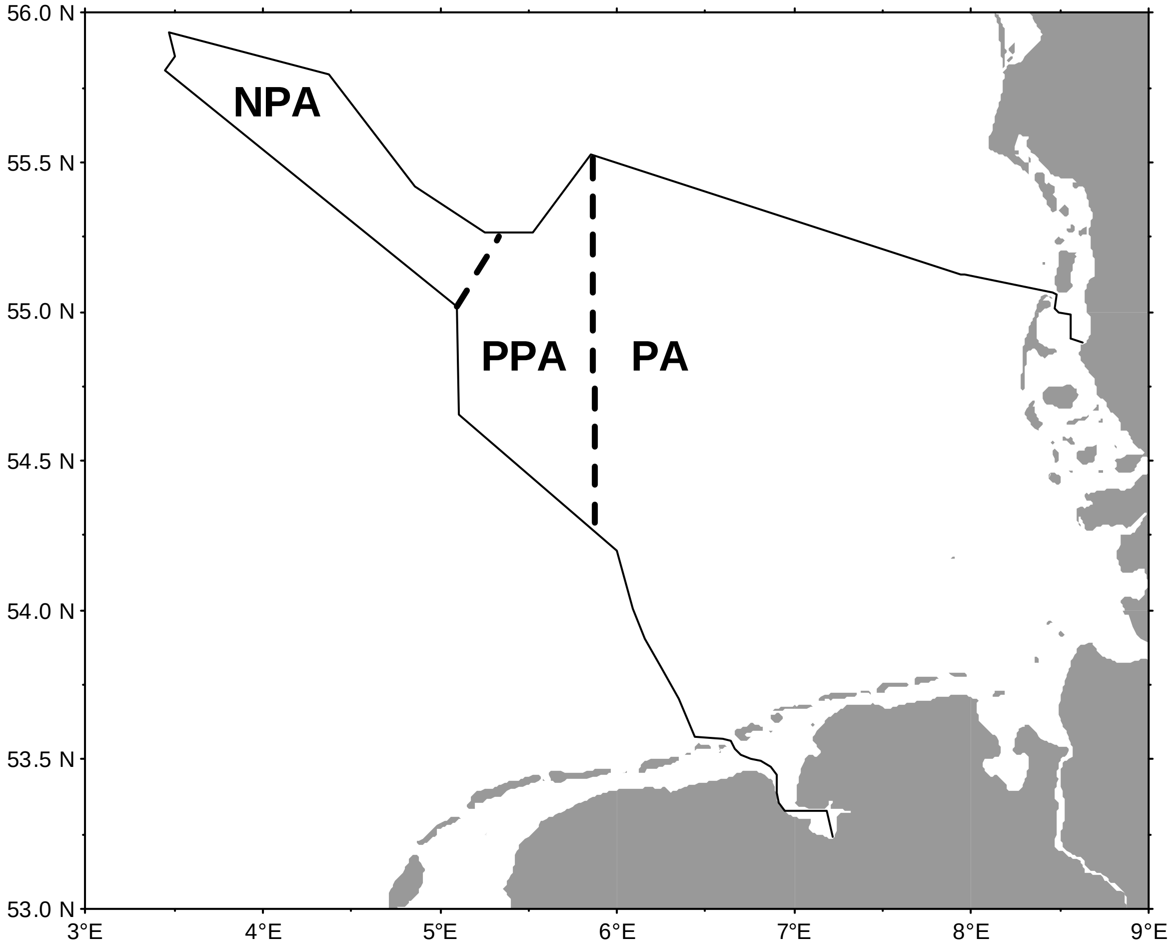


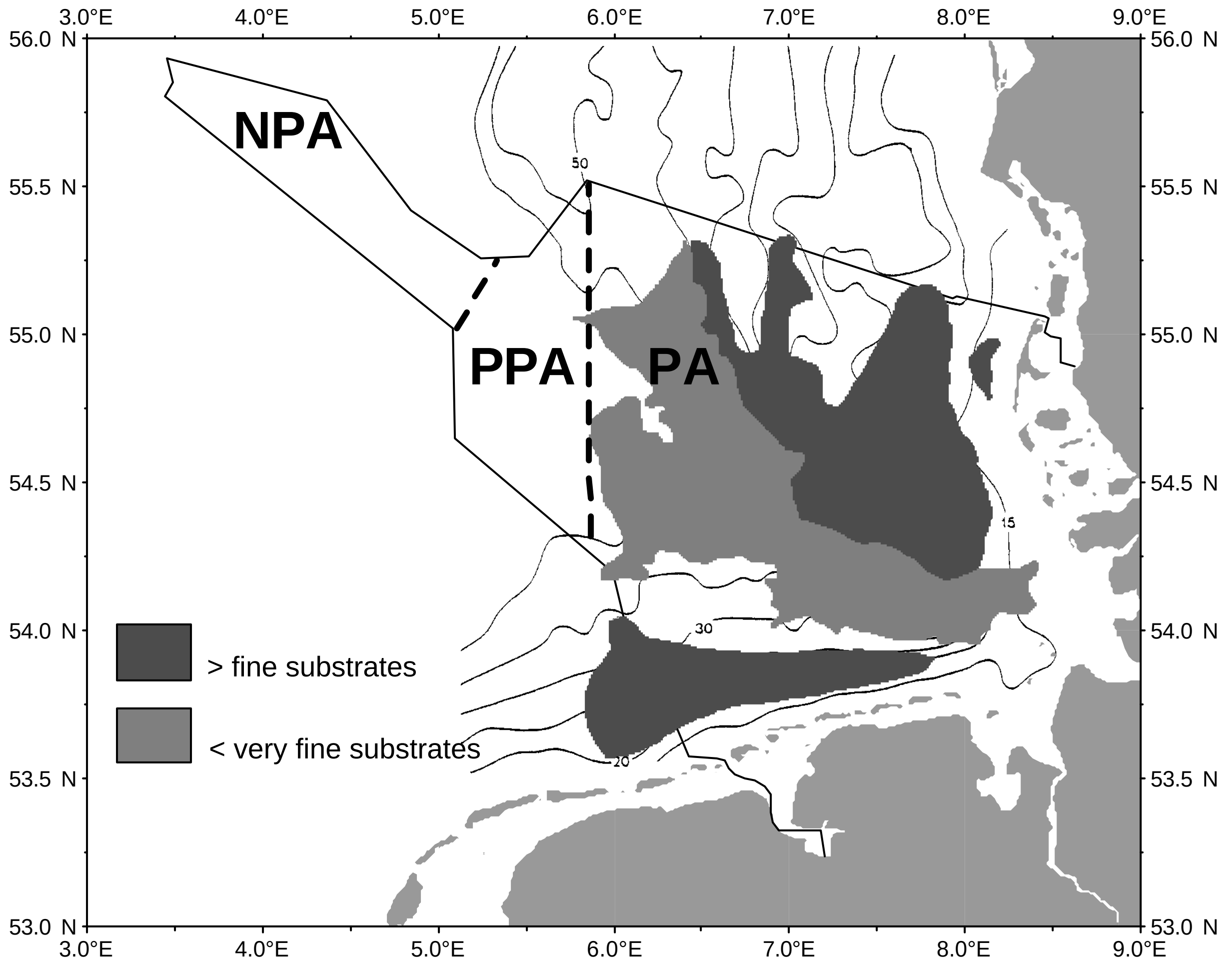
Fig. 2.8.2.2.2: Abundance of *Nucula nitidosa* at two different stations in the German Bight [redrawn from Rachor 1990, in: Lozán et al. 1990].



classification of German territorial waters and the exclusive economic zone into:

- problem areas due to eutrophication (PA)
- potential problem areas (PPA)
- non-problem areas (NPA)

Fig. 3.1.0: Classification of German territorial waters and the exclusive economic zone



classification of German territorial waters and the exclusive economic zone into:

- problem areas due to eutrophication (PA)
- potential problem areas (PPA)
- non-problem areas (NPA)

Fig. 3.1.1: Classification of German territorial waters and the exclusive economic zone, substrate gradients and depth contours (redrawn from Figge 1981).

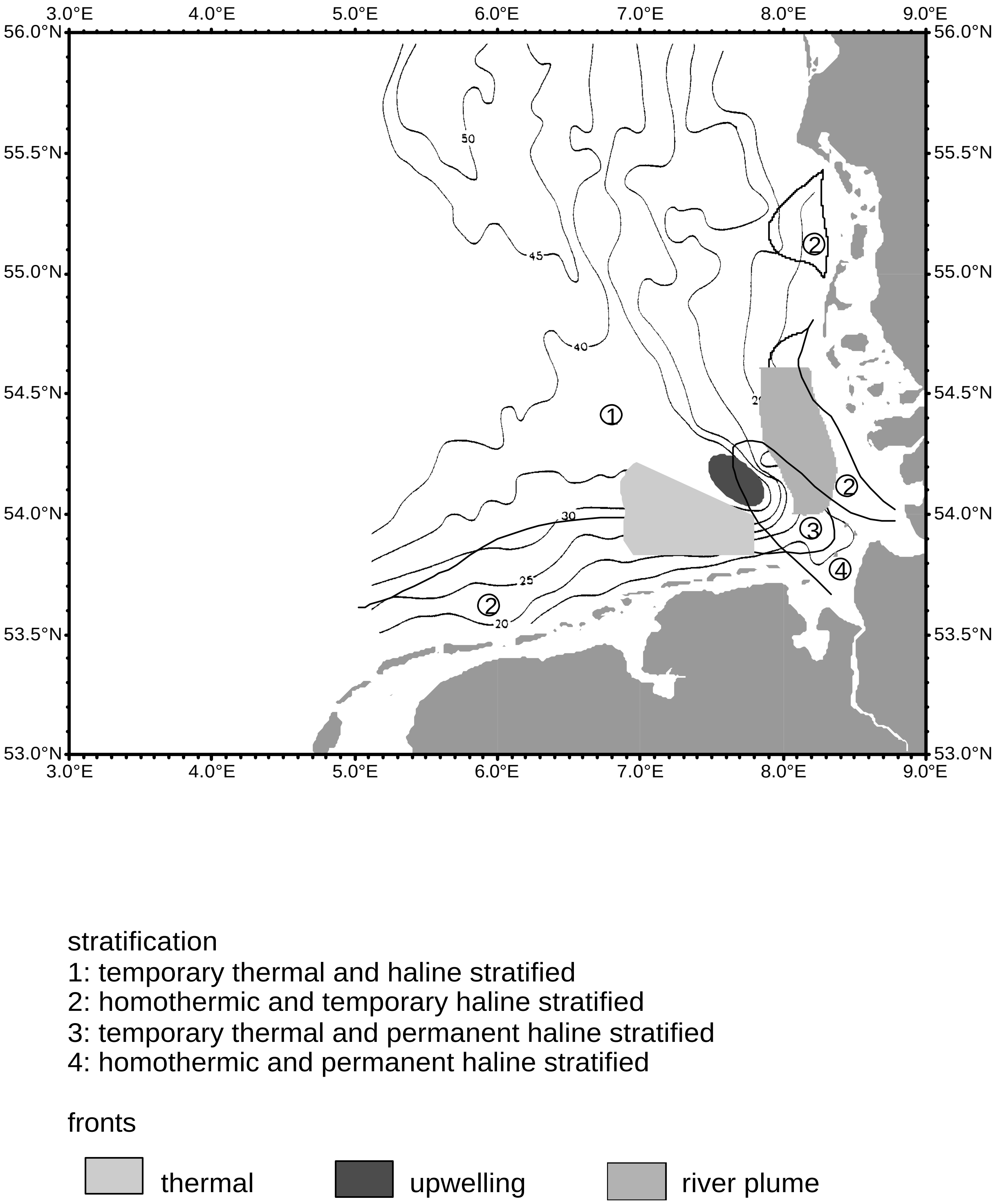
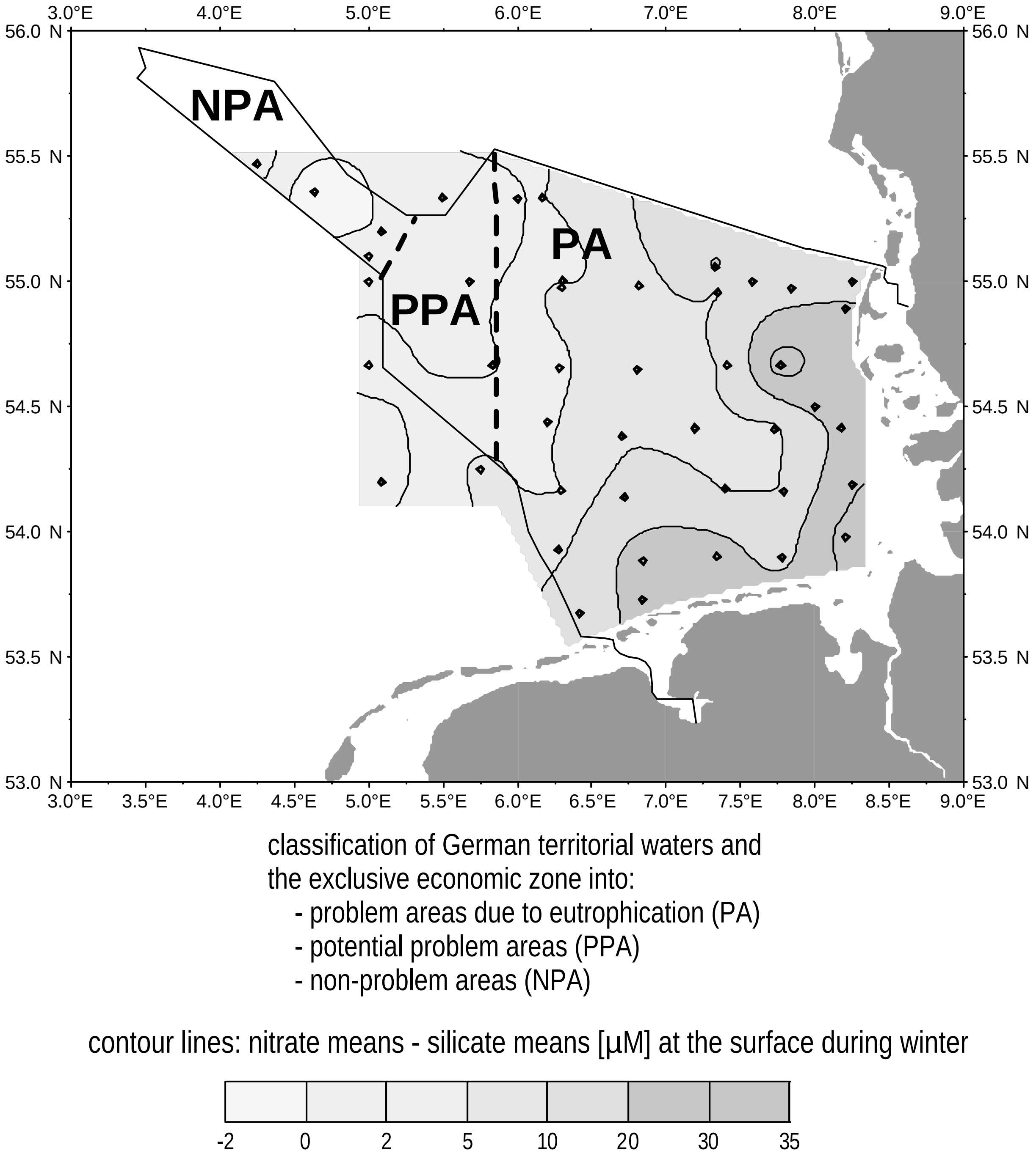


Fig. 3.1.2: The German Bight with bottom topography (depth in meters), stratification and fronts. [redrawn from Frey 1990 and Krause et al. 1986]



classification of German territorial waters and
the exclusive economic zone into:

- problem areas due to eutrophication (PA)
- potential problem areas (PPA)
- non-problem areas (NPA)

contour lines: nitrate means - silicate means [µM] at the surface during winter

Fig. 3.1.3: Classification of territorial waters and the exclusive economic zone, and differences between nitrate and silicate means during winter